Seasonal Variations in Nutrient and Total Suspended Solids Sources and Fluxes in Rivers of Northern New Jersey and Newark Bay, USA

A DISSERTATION

Submitted to the Faculty of Montclair State University in partial fulfillment

of the requirements

for the degree of Doctor of Philosophy

by

Jing Nie

Montclair State University

Montclair, NJ

May 2023

Dissertation Chair: Dr. Huan Feng

MONTCLAIR STATE UNIVERSITY

THE GRADUATE SCHOOL

DISSERTATION APPROVAL

We hereby approve the Dissertation

Seasonal Variations in Nutrient and Total Suspended Solids Sources and Fluxes in Rivers of Northern New Jersey and Newark Bay, USA

of

Jing Nie

Candidate for the Degree:

Doctor of Philosophy

Graduate Program:

Environmental Science and Management

Certified by:

[redacted]

Dr. Kenneth Sumner
Vice Provost for Research and
Dean of the Graduate School

Date:

Dissertation Committee:

[redacted]

Dr. Huan Feng
Dissertation Chair

[redacted]

Dr. Yang Deng

[redacted]

Dr. Danlin Yu

[redacted]

Dr. Shinjae Yoo

[redacted]

Dr. Benjamin Witherell

Abstract

In recent decades, aquatic systems have experienced major problems with water quality due to high nutrient concentrations from both point and non-point sources resulting from industrialization, urbanization, and population growth. While nutrient pollution due to land use change cannot be ignored, point sources such as combined sewer overflows and discharging sites have also contributed to the problem. Integrated hydrodynamic, chemical, and biological models have been developed to simulate nutrient transportation from both sources. This paper reviews and analyzes water quality data from published literature to evaluate nutrient pollution in aquatic systems and emphasizes the need for a continuously developed integrated monitoring and management plan to regulate nutrient discharges.

Two studies were conducted in northern New Jersey, USA, to examine the impact of land use change on water quality in the Passaic River and to estimate nutrient fluxes from the Passaic and Hackensack Rivers into Newark Bay. The first study used long-term water quality monitoring and land-use data to show that urban land use is a significant contributor to water quality problems in the Passaic River, while natural landscapes dominate the area. The second study collected bi-weekly total inorganic nitrogen and orthophosphate concentration data over 15 years to estimate the annual nutrient loading from both rivers, which varied seasonally due to weather conditions such as hurricane events.

Another study investigated the relationship between total suspended solids (TSS) loadings and Land Use Land Cover (LULC) type across six drainage basin areas in New Jersey, using 16 years of published monitoring data. The study found that water discharge has a strong correlation with the area of a drainage basin and that TSS concentration is positively correlated with medium and high developed LULC types and negatively impacted by forests and wetlands. The study also used the ARIMA model to forecast future TSS loading trends and fluctuations over time, indicating its effectiveness in capturing cyclic patterns, especially with seasonal variations in time series data.

ACKNOWLEDGEMENTS

I would like to express my sincere gratitude to Dr. Huan Feng, my advisor, for his guidance, encouragement, and support throughout the dissertation process. Without his invaluable advice and feedback, this work would not have been possible. I am also grateful to my committee members, Drs. Deng Yang, Danlin Yu, Ben Witherell and Shinjae Yoo for their insightful comments and suggestions, which helped to improve the quality of this dissertation.

I express my gratitude to Dr. Mark Chopping, Director of the Doctoral Program in Environmental Science and Management PhD program at Montclair State University (MSU), and Dr. Stefanie Brachfeld, former chair of the Earth and Environmental Studies (EAES) Department and Associate Dean for Academic Affairs and Research in the College of Science and Mathematics (CSAM), for their generous support and cooperation. Additionally, I am appreciative of the guidance and supervision provided by all the EAES and CSAM faculty members during my work as a teaching assistant and lecture instructor at MSU.

I express my sincerest appreciation to MSU and CSAM for providing me with a doctoral assistantship, as well as to the Graduate School for their assistance in editing and formatting my dissertation. Special thanks to Rebecca Newton for her unwavering support.

Furthermore, I would like to extend my appreciation to my fellow students and friends, Junkui Cui, Lisitai Yang, Du Xu, Qiufeng Lin, Sana Miraz, Oluwafemi Soetan, Michael Viteritto, Krishna Polius who provided me with emotional support and motivation during this challenging journey. Their constant encouragement and understanding have been crucial in helping me to stay focused and achieve my goals.

Finally, and above all, I owe an immense debt of gratitude to my remarkable parents for their unwavering love, affection, and inspiration throughout my life. I also want to thank my fiancé, who has been my strongest supporter and life saver, for his unwavering support and unwavering commitment during the ups and downs of my doctoral studies.

Dedications

To my parents Lihua Zhou and Youcheng Nie, my fiancé Ximing Wang

and

To the people who always support me

Table of Contents

List of Tables

List of Figures

Chapter 1 Summarizing report

1. Introduction

1.1 Nutrient and total suspend solids concentration impacts on riverine and Newark Bay systems.

Rivers and streams play a critical role in supporting diverse aquatic ecosystems and human communities that depend on them (Wurtsbaugh, Paerl, & Dodds, 2019). However, the health and sustainability of these systems can be significantly impacted by various environmental factors, including nutrient concentrations (Ardón et al., 2021). Nutrients, such as nitrogen and phosphorus, are essential for the growth and survival of aquatic organisms, but excessive amounts of these nutrients can lead to a range of ecological problems, including eutrophication, harmful algal blooms, and oxygen depletion (Q. Chen et al., 2020). Therefore, understanding the impacts of nutrient concentrations on river streams is crucial for managing and protecting these ecosystems (Wijesiri et al., 2019). Total suspended solids (TSS) concentration is a crucial parameter in determining the quality of river streams (Yu et al., 2019). It's suspended particles in water that include silt, clay, organic matter, and inorganic matter (Zeng, Han, & Yang, 2020). The concentration of TSS can have a significant impact on the physical, chemical, and biological properties of river streams (Tang et al., 2019). High TSS concentration can reduce water clarity, decrease the amount of light that penetrates the water, and affect aquatic plant growth (Abdul Maulud et al., 2021). It can also increase the water temperature, reduce dissolved oxygen, and affect aquatic animal habitats (Ustaoğlu, Tepe, & Taş, 2020). Therefore, understanding the effects of TSS concentration on river streams is crucial for maintaining healthy ecosystems and ensuring sustainable water resource management.

Coastal and bay areas are impacted by high levels of nutrient contamination. Studies implied that sediment resuspension by hurricanes accelerates the release of nutrient to the dissolved phase (Kalnejais, Martin, & Bothner, 2010). The results suggest that the occurrence of unnormal high nutrients concentration may be caused by hurricanes. Another study in Sweden certified the impacts of resuspension. In this study, resuspension changes the flux rates. It decreases the flux of phosphate and increases the flux of nitrate-nitrite. However, the flux of ammonia-nitrogen show no significant change (Tengberg, Almroth, & Hall, 2003). The coastal and bay areas are especially vulnerable to the impact of TSS concentration due to their proximity to land, anthropogenic activities, and hydrodynamic characteristics (Villa, Fölster, Kyllmar, & assessment, 2019). The TSS concentration in these areas is influenced by various factors such as stormwater runoff, wastewater discharge, erosion, and sedimentation (Copetti et al., 2019). Studies have shown that high TSS concentrations can have several negative impacts on coastal and bay areas, including reduced light penetration and increased turbidity, which can harm aquatic plants and animals by reducing photosynthesis, interfering with feeding, and altering habitats (Jiang et al., 2021). High TSS concentrations can also lead to sedimentation, which can smother bottom-dwelling organisms, reduce water clarity, and alter water flow dynamics (Serajuddin, Chowdhury, Haque, & Haque, 2019).

Estuarine areas are special because the polluted streams in these areas may have impacts on coastal waters. Integrated management methods are in highly needs for identifying and control the possible pollution contributes to the estuarine areas(Gaspar et al., 2017). High nutrient concentrations not only damage the aquatic ecosystem but also endanger human health. Especially for the main river supplying drinking water, nutrient concentrations should be controlled in a safe range(Chaudhary, Mishra, & Kumar, 2017).

Nutrient concentration is vital to human health, ecosystem, and environment. It is affected by both natural process and human activities(Shin, Artigas, Hobble, & Lee, 2013). During the last decades, thousands of water quality restoration projects were conducted(Ofiara, 2015). The Passaic River has been severely contaminated historically due to the industrialization in the area (Parette & Pearson, 2014). Those studies suggested that it was a practical method for researchers to conduct pattern and historical trend study to estimate the transportation of pollution in water.

1.2 Land use land cover impacts on water quality

Urbanization has had a significant impact on aquatic systems. The intensive urban development has caused a key issue in habitat health as indicated by the water quality. Surface water quality can be a good indicator to monitoring the urban development and environment quality change (Brabec, Schulte, & Richards, 2002). Water quality in the watershed is impacted by polluted non-point source runoff and point source pollutant discharge such as nutrients.

Urban development is the driving force for the United States to change the environment. Newly urbanized land can bring economic profit. In the meantime, however, it causes the damage to the ecosystem (Lathrop, Tulloch, & Hatfield, 2007). Reducing the non-point source pollution from agriculture is an essential issue to improve the nutrient concentrations in aquatic ecosystems. Agriculture may improve or negatively affect the water quality based on specific situations. Crop planting can help keep metal and materials in soil and roots. However, overdose insecticide and nutrients (e.g., phosphate and nitrogen) can result in eutrophication in the water body. Knowing the processes of how agricultural land use change affect the aquatic ecosystem will help to implement sustainable water management (B. Mehdi et al., 2015).

When land is paved, built upon, or altered in other ways (indicated by impervious surface area), it can increase the amount and speed of runoff, which can carry pollutants and sediment into nearby streams and rivers (Dutta, Rahman, Paul, & Kundu, 2021). Agriculture, forestry, and construction can all contribute to soil erosion, which can also increase sediment in waterways and reduce water quality (Mishra, Rai, Rai, & Science, 2020). Agricultural and urban land uses can both contribute to nutrient pollution, which can lead to harmful algal blooms, oxygen depletion, and other negative impacts on aquatic ecosystems (S. W. Wang, Gebru, Lamchin, Kayastha, & Lee, 2020). Industrial and urban land uses can contribute to chemical contamination of waterways, including heavy metals, pesticides, and other toxic substances. Altering land use can also result in habitat destruction for aquatic species, which can have cascading impacts on the health of aquatic ecosystems (Naikoo, Rihan, & Ishtiaque, 2020).

1.3 Seasonal patterns of water quality in river

Due to the different characteristics of riverine and estuarine areas, water quality will be different both in spatial and temporal scale, and these differences reveal the terrestrial influences (W. Zhu, Y. Q. Tian, Q. Yu, & B. L. J. R. S. o. E. Becker, 2013b). The amount and timing of rainfall can have a significant impact on river water quality. In general, increased precipitation can lead to higher flows, which can increase erosion, sedimentation, and nutrient and pollutant runoff (Shi et al., 2020). Based on the studies on spatial and temporal variations in nutrient concentrations, the researcher concluded that seasonal change with a different pattern of precipitation could affect the nutrient concentrations (S. Li et al., 2009). In addition, with high river flow, nutrient concentrations tend to decrease. And from the analysis, urbanization and agricultural area have higher nutrient concentrations (S. Li et al., 2009). Many factors can affect the nutrient concentrations in the aquatic environment. A study on coastal Louisiana freshwater lake indicated that sediment redox condition affected the release of metal and nutrients into the aquatic environment. Iron (Fe) concentration reduction in waterbody will result in an increase in phosphate concentration (Miao, DeLaune, & Jugsujinda, 2006).Studies in the Mediterranean and the Black Sea revealed that with water flux decreasing, total flux inputs of N and P from the river to the bay area increased. Associated ecosystem change happened due to the reconstructed nutrients river inputs. The results indicated the river discharge of nutrients was the major driver for ecosystem change (Ludwig, Dumont, Meybeck, & Heussner, 2009).

Water temperature can also influence water quality by affecting the solubility of nutrients and other substances, as well as the growth and metabolism of aquatic organisms (Meshesha, Wang, & Melaku, 2020). Different land uses, such as agricultural fields, urban areas, and forests, can all have different impacts on water quality depending on the season (J. Zhang, Li, Dong, Jiang, & Ni, 2019). For example, in agricultural areas, nutrient runoff may be higher during the growing season, while in urban areas, higher temperatures and increased stormwater runoff may lead to higher pollutant concentrations in the summer (Xu et al., 2019). The activity of plants and other organisms in the river can also influence water quality, with higher levels of photosynthesis and respiration leading to fluctuations in oxygen and other nutrients (Nobre et al., 2020). seasonal patterns of water quality in rivers are complex and can be influenced by a variety of factors. Monitoring water quality throughout the year and understanding the underlying drivers of variation can help to identify areas of concern and inform management and conservation efforts.

1.4 Causes and treatment of nutrient pollution in rivers and estuaries

During the last decades, anthropogenic activities contribute the major nutrient contamination source into the waterbody. The main contamination sources can be categorized as fertilizer, animal wastes, human sewage, household products, byproducts from petroleum and agriculture fields. The main fertilizer is from agriculture development, other sources includes industrial manufacturing and lawn use (Antweiler, Goolsby, & Taylor, 1996). South China had a serious nutrients contamination issue and eutrophication in river and estuary is still at high level. In Huang's study, total dissolved inorganic nitrogen and phosphate and their ratio are the major characters evaluating nutrients condition in the waterbody (Huang, Huang, & Yue, 2003). Sewage treatment plants that discharge effluent into rivers and estuaries can contribute to nutrient pollution if the treatment process does not effectively remove excess nutrients. Previous studies indicate that an increase in nutrient concentrations is mainly caused by wastewater discharges and urban or agricultural stormwater runoff (John Fillos & William R Swanson, 1975). Stormwater runoff from urban areas can contain high levels of nutrients from lawns, gardens, and other sources (Osman et al., 2019). Nitrogen can be deposited into rivers and estuaries through rainfall and atmospheric deposition from industrial emissions and transportation (Keiser, Kling, & Shapiro, 2019).

Treatment of nutrient pollution typically involves reducing the amount of nutrients entering the water body, improving water quality through various methods, and restoring affected ecosystems (Council, 2019). Strategies such as improving agricultural practices, using low-phosphorus detergents, and implementing stormwater management practices can all help to reduce nutrient inputs into waterways (Lintern et al., 2020). Wastewater from wastewater treatment plants, known discharging points are point sources of pollution. With the information from discharge location, it's much easier to identify and control the pollution. Advanced wastewater treatment processes, such as tertiary treatment or nutrient removal systems, can effectively remove excess nutrients from wastewater before discharge (Grizzetti et al., 2021).

However, non-point sources of pollution such as stormwater runoff from the agricultural and urban area is managed by other practice strategies. Restoring natural wetlands and riparian areas can help to absorb and remove excess nutrients from runoff before it reaches rivers and estuaries (Jabłońska et al., 2020). Vegetative buffer zones along the edges of rivers and streams can help filter and absorb excess nutrients, as well as sediment and other pollutants (Walton et al., 2020). Implementing strategies to reduce emissions from transportation and industry can help to reduce the amount of nitrogen deposited into waterways (Walton et al., 2020) . Overall, treating nutrient pollution in rivers and estuaries requires a combination of approaches tailored to the specific circumstances and characteristics of the affected water body.

2. Materials and Methods

2.1 Study area

The study area includes six major rivers in North Jersey, USA (Figure 1-1). Table 1-1 lists the six rivers that are the subject of this study: Passaic River, Saddle River, Hackensack River, Elizabeth River, Rahway River, and Raritan River. The study area in North New Jersey is depicted in Figure 1-1, with each river represented by a station (St). Passaic River, with a length of 120 km (Kenneth R Olson, Tharp, & Conservation, 2020), is located within a river basin of 2,135 km^2 (Oteng Mensah, Alo, & Change, 2023) and is a significant waterway in northeastern New Jersey. It flows through several counties, including Morris, Somerset, Union, Essex, Passaic, and Bergen, and has a history of industrial use (Ophori, Firor, & Soriano, 2019).

Saddle River, a tributary of Passaic River, flows through Bergen County and is approximately 40 km long (Graham, Graham, & Wilcox, 2020). The Hackensack River, another major river in the New York City metropolitan area, flows through Bergen and Hudson counties and has a length of approximately 72 km (Reinfelder & Janssen, 2019). Elizabeth River, a tributary of Newark Bay, is 6.4 km long and provides access to the Arthur Kill and New York Harbor (Bozinovic et al., 2021). It has been impacted by pollution from various sources due to its location (Wieczerak, Wolde, Lal, Witherell, & Deng, 2020).

The Rahway River, with a length of approximately 39 km, flows through Essex, Union, and Middlesex counties and empties into Arthur Kill, which separates Staten Island, New York from mainland New Jersey (Mousa, Hussein, & Kineber, 2022). It is prone to flooding, particularly in the lower reaches (Alagrabawi, 2022). The Raritan River, with a length of approximately 137 km, flows from Morris County to Raritan Bay in Middlesex County and has a watershed covering about 2845 km^2, including parts of several counties (Slattery, 2022). It has been affected by pollution from industrial and agricultural sources, but there are ongoing efforts to restore its ecological health (Y. Wang, Gong, & Di, 2022).

2.2 Data processing and analysis

Three different types of datasets were used in the study area (Figure 1-1). Water quality data came from New Jersey Harbor Discharge Group (NJHDG); Water discharge data was from United States Geological Survey (USGS) discharge gage station; Land use land cover data was from National Land Cover Database (NLCD) product provided by USGS. Figure 1-2 shows the water quality stations used in Passaic River and the Combined sewage overflow (CSO) sites along the Passaic River. Figure 1-3 shows the water quality sites and discharge sites in the Passaic River, the Hackensack River, Saddle River and Newark Bay. Data from these sites were used for calculation of nutrient fluxes into Newark Bay. Figure 1-4 shows the drainage basin of each river in the study area and their land use land cover in 2004. These datasets help to quantify the relationship between land use type and water quality.

3. Organization of the dissertation

The chapters in this study are organized and displayed as follows:

Chapter 1. Summarizing Report - This chapter presents a synopsis of the dissertation and outlines the research findings, along with suggestions for future inquiries.

Chapter 2. Causes, Assessment and Treatment of Nutrient (N and P) Pollution in Rivers, Estuaries and Coastal Waters – This chapter published in Current Pollution Reports from Springer International Publishing in June 2018. This chapter reviews and analyzes water quality data from published literature to evaluate nutrient (N and P) pollution in aquatic systems.

Chapter 3 Nitrogen and phosphorus concentrations and their potential sources in urban watersheds in Northern New Jersey, USA - This chapter examines the impact of land use change on water quality in the Passaic River, New Jersey, using 13 years of water quality monitoring and land-use data. The study distinguishes between non-point sources of nutrient pollution (land cover of the watershed) and point sources (combined sewer overflow and dry cleaner sites). Results indicate that urban land use, caused by local industrialization and urbanization, has a significant impact on water quality, while agricultural land is not dominant in the study area. The Passaic River watershed is still mostly made up of natural landscapes, such as wetlands and forests.

Chapter 4 Estimation of Nutrient (N and P) Fluxes into Newark Bay, USA - This chapter was published in Marine Pollution Bulletin from Pergamon in March 2023. This chapter estimates the nutrient (N and P) fluxes from the Passaic River, the Hackensack River, and other sources into Newark Bay and the nutrient residence time in Newark Bay in northern New Jersey, USA. Data on total inorganic nitrogen (TIN) and orthophosphate concentrations were collected bi-weekly for over 15 years (2004-2019), along with daily river discharge data. Results show that the annual TIN and ortho-P loading from the Passaic River ranged from 915×10^{3} kg y^{-1} to 251×10^{4} kg y^{-1} and 94×10^{3} kg y^{-1} to 372×10^{3} kg $^{y-1}$, respectively, while the annual TIN and ortho-P loading from the Hackensack River ranged from 3.13×10^{3} kg y^{-1} to 234×10^{3} kg y^{-1} and 0.28×10^{3} kg y^{-1} to 6.97×10^{3} kg y^{-1}, respectively. The study also finds that hurricane events increase TIN and ortho-P loading from riverine input and reduce residence time in Newark Bay.

Chapter 5 Statistical Analysis of Total Suspended Solids Loadings and Potential Relations with Land Use Land Cover Type in New Jersey – This chapter examines the relationship between total suspended solids (TSS) loadings and Land Use Land Cover (LULC) type across six drainage basin areas in New Jersey using 16 years of published monitoring data. The study finds a strong correlation between water discharge and the area of a drainage basin. Positive correlations are observed between TSS concentration and medium and high developed LULC types, while forests and wetlands have a negative impact on TSS concentration. The study also analyzes annual and seasonal variations of TSS loading, showing that hurricane and storm events have a significant impact on TSS loading, with Hurricane Irene having the greatest impact. Finally, the ARIMA model is employed to forecast future TSS loading trends and fluctuations over time, which is well-suited for capturing cyclic patterns, especially with seasonal variations in time series data.

4. Research objectives

4.1 Objective 1

¶ Summarize the literature review to gain a comprehensive understanding of the various factors contributing to water quality pollution in northern New Jersey and the potential remedies available.

4.2 Objective 2

¶ To establish a foundation for understanding the relationship between nutrient concentration and land use type through analysis of nutrient concentration data and land use land cover data.

¶ To calculate river fluxes of nutrients using discharge data and estimate the riverine nutrient input to Newark Bay.

¶ To analyze nutrient fluxes data and gain insight into seasonal patterns of nutrient levels in rivers as well as the impacts of storm events.

4.3 Objective 3

¶ To conduct statistical analysis using TSS data and land use land cover data to determine the potential relationship between land use type and TSS in rivers.

¶ To incorporate drainage area data to establish the correlation between discharge data and drainage area data.

5. Results/Findings

5.1 Causes and treatment of nutrient pollution in rivers, estuaries and coastal waters

While combined sewer overflows remain the primary source of nutrient pollution in rivers, estuaries, and coastal waters, effective methods such as wetland construction and CSO management can help reduce their impact on the aquatic environment. However, non-point sources of nutrient pollution resulting from land use and land cover changes remain a significant challenge, and quantifying and identifying storm water runoff from agricultural and urban areas continues to be difficult. Identifying and controlling point source pollution is relatively feasible when discharge locations are known. To improve water quality, multiple management strategies should be implemented, and future cost-benefit analyses should be conducted. Furthermore, an integrated hydrodynamic, chemical, and biological model should be developed to assist in identifying the transport and fate of nutrients from both point and non-point sources. Nutrients are vital indicators of water quality, essential to both human life and aquatic ecosystems.

5.2 Regional land use type and its potential impact on water quality.

After examining the critical impact of land use type on water quality in riverine-estuarine systems, the findings reveal that the upper Passaic River watershed, which features a high proportion of natural pristine forest landscape, exhibits water quality levels within the natural range. In contrast, the lower Passaic River watershed, which is predominantly urbanized, experiences significant degradation due to anthropogenic sources such as sewage discharges, industrial effluents, and stormwater runoff.

The results of this study underscore the importance of both water quality monitoring and land use management in the sustainable development of the Passaic River watershed. Continuous monitoring of water quality parameters such as nutrient concentrations, dissolved oxygen, and pH is essential for understanding the current state of the watershed and tracking the effectiveness of management strategies over time. Additionally, proactive land use management measures such as implementing green infrastructure, controlling pollutant sources, and promoting sustainable land use practices can help prevent further degradation and improve the health of the ecosystem.

Furthermore, these findings have important implications for other riverine-estuarine systems globally, particularly those in urbanized areas where anthropogenic sources are a significant threat to water quality. By implementing effective water quality monitoring and land use management practices, it is possible to mitigate the negative impacts of human activities on these ecosystems and ensure their sustainability for generations to come.

5.3 Estimation of nutrient fluxes into Newark Bay, USA

This part aimed to explore the yearly and seasonal variations of TIN and ortho-P fluxes in Newark Bay and the residence time of these nutrients. Furthermore, this part also calculated the fluxes from other sources based on riverine input and nutrient mass in Newark Bay. The findings showed that hurricane events caused a peak in nutrient loading from the riverine and reduced the residence time in Newark Bay. The Passaic River was identified as the major riverine source of TIN and ortho-P, with annual loadings (Figure 1-5) ranging from 1016×10^3 kg y^{-1} to 2864×10^3 kg y^{-1} and 94.3×10^3 kg y^{-1} to 372×10^3 kg y^{-1}.

Seasonal variations in TIN loading were observed in the Passaic River, with higher loadings occurring in winter and spring under normal and hurricane conditions. However, storm events caused higher loading in summer instead of winter. In contrast, no seasonal variation in TIN loading was detected in the Hackensack River under normal and storm conditions. Nonetheless, TIN loading increased during spring and winter under hurricane conditions, as observed in the Passaic River. Both rivers showed no seasonal variations in ortho-P loading under any weather condition.

Residence time of nutrients was substantially reduced by hurricane events. Under normal and storm conditions, ortho-P showed seasonal changes mainly due to loading from other sources. TIN loading from the Hackensack River played an essential role in TIN mass variation in Newark Bay. Under hurricane conditions, TIN mass increased slightly during spring due to higher loading from both the Passaic and Hackensack Rivers. In contrast, the inner sediments of Newark Bay were the primary source of ortho-P mass during hurricane events. Overall, these findings have significant implications for managing nutrient pollution in Newark Bay, highlighting the importance of accounting for seasonal and extreme weather conditions when designing management strategies.

5.4 Statistical Analysis of Total Suspended Solids Loadings and Potential Relations With Land Use Land Cover Type in New Jersey

In this study, the researchers investigate the relationship between TSS loadings and LULC (land use/land cover) type in six different drainage basin areas. The study also explores the annual and seasonal variations of TSS loadings to understand the pattern of seasonal change and the impact of storm and hurricane events. The researchers find that there is a strong linear relationship between the drainage basin area and the discharge, with more drainage area leading to more discharge into the river.

Using Pearson's correlation matrix, the researchers investigate the relationship between TSS concentration and LULC, finding a positive correlation between TSS concentration and the spatial difference (station number) and the level of development (low, medium, and high intensity) (Figure 1-6). The study also finds that certain LULC types, such as MI and HI, have a strong positive correlation with TSS concentration, while others, such as EForest and WWetland, have a significant negative impact on TSS concentration. The researchers report negative correlations between TSS concentration and several other LULC types, including Open Space, MForest, Barren, Crop, Water, Shrub, DForest, Ewetland, Grass, and Pasture.

The study also examines the annual and seasonal variation of TSS loading in six rivers in New Jersey under different weather conditions (Figure 1-7&Figure 1-8). The researchers find that the Raritan River had the highest TSS loading in 2007, 2018, and 2019, while the Passaic River contributed the highest TSS loading among the six rivers analyzed. The impact of hurricanes and storms on TSS loading is also investigated, with Hurricane Irene being the most significant event affecting the rivers' water quality. The study finds that Saddle River, Hackensack River, and Rahway River were significantly affected by weather conditions, with storm events having a significant impact on Saddle River and Rahway River, while hurricanes highly impacted Hackensack River. The Rahway River and Saddle River showed significant seasonal differences and were highly impacted by storm events, while Elizabeth River and Passaic River did not exhibit any significant differences in TSS loading seasons and showed no response to storm or hurricane events.

Finally, the researchers used an ARIMA model to capture the cyclic patterns that occur within the time series data and to provide more accurate and reliable forecasting of future trends and fluctuations over time. The incorporation of seasonal variations into the ARIMA model helps to account for the effects of seasonal factors such as holidays and weather patterns, making it a powerful tool for analyzing and forecasting time series data with seasonal components. Overall, this study provides valuable insights into the relationship between TSS loadings, LULC type, and weather conditions, and highlights the importance of incorporating seasonal variations into forecasting models.

6. Summary and Recommendations

Estimating nutrient fluxes in rivers is crucial in comprehending the impact of runoff and land use on water quality in riverine and bay areas. By summarizing the yearly and seasonal patterns of nutrient and TSS fluxes, a better understanding of their correlation can be obtained. This knowledge can assist in developing appropriate management strategies to mitigate water pollution. Statistical studies have linked the concentration of TSS with land use type, indicating the importance of identifying and managing land use practices that contribute to water pollution.

Moreover, the study highlights the significant influence of drainage areas on the discharge into rivers. Future research should focus on calculating nutrient fluxes of each river and analyzing their potential relationships with land use types. Such studies can provide more precise estimates of the nutrient load and help develop effective water management policies.

References

Abdul Maulud, K. N., Fitri, A., Wan Mohtar, W. H. M., Wan Mohd Jaafar, W. S., Zuhairi, N. Z., & Kamarudin, M. K. A. J. A. J. o. G. (2021). A study of spatial and water quality index during dry and rainy seasons at Kelantan River Basin, Peninsular Malaysia. *14*, 1-19.

Adimassu, Z., Langan, S., Johnston, R., Mekuria, W., & Amede, T. (2017). Impacts of Soil and Water Conservation Practices on Crop Yield, Run-off, Soil Loss and Nutrient Loss in Ethiopia: Review and Synthesis. *Environmental Management, 59*(1), 87-101.

Alagrabawi, Y. (2022). The Effects and Costs of Changes in River Flooding in New Jersey.

Alamdari, N., Claggett, P., Sample, D. J., Easton, Z. M., & Yazdi, M. N. J. J. o. C. P. (2022). Evaluating the joint effects of climate and land use change on runoff and pollutant loading in a rapidly developing watershed. *330*, 129953.

Allen, J. I., Holt, J. T., Blackford, J., & Proctor, R. (2007). Error quantification of a high-resolution coupled hydrodynamic-ecosystem coastal-ocean model: Part 2. Chlorophyll-a, nutrients and SPM. *Journal of marine systems, 68*(3), 381-404.

Amar, M., Bauter, N., Bonomo, J., Burchell, A., Dua, K., Granton, C., . . . Prioleau, D. (2014). Bringing the City of Newark's Stormwater Management System into the 21st Century.

Anderson, J. R. (1976). *A land use and land cover classification system for use with remote sensor data* (Vol. 964): US Government Printing Office.

Anderson, N. M., Germain, R. H., & Hall, M. H. (2012). An Assessment of Forest Cover and Impervious Surface Area on Family Forests in the New York City Watershed. *Northern Journal of Applied Forestry, 29*(2), 67-73. doi:10.5849/njaf.11-009

Antweiler, R. C., Goolsby, D. A., & Taylor, H. E. (1996). Nutrients in the Mississippi river. *US GEOLOGICAL SURVEY CIRCULAR USGS CIRC*, 73-86.

Appelo, C. A. J., & Postma, D. (2004). *Geochemistry, groundwater and pollution*: CRC press.

Ardón, M., Zeglin, L. H., Utz, R. M., Cooper, S. D., Dodds, W. K., Bixby, R. J., . . . Harms, T. K. J. B. R. (2021). Experimental nitrogen and phosphorus enrichment stimulates multiple trophic levels of algal and detrital based food webs: a global meta analysis from streams and rivers. *96*(2), 692-715.

Arndt, S., Lacroix, G., Gypens, N., Regnier, P., & Lancelot, C. (2011). Nutrient dynamics and phytoplankton development along an estuary–coastal zone continuum: a model study. *Journal of marine systems, 84*(3), 49-66.

Arnold Jr, C. L., & Gibbons, C. J. (1996). Impervious surface coverage: the emergence of a key environmental indicator. *Journal of the American planning Association, 62*(2), 243-258.

Association, A. P. H., Association, A. W. W., Federation, W. P. C., & Federation, W. E. (1915). *Standard methods for the examination of water and wastewater* (Vol. 2): American Public Health Association.

Bateman, I., Agarwala, M., Binner, A., Coombes, E., Day, B., Ferrini, S., . . . Posen, P. (2016). Spatially explicit integrated modeling and economic valuation of climate driven land use change and its indirect effects. *Journal of environmental management, 181*, 172-184.

Becouze-Lareure, C., Thiebaud, L., Bazin, C., Namour, P., Breil, P., & Perrodin, Y. (2016). Dynamics of toxicity within different compartments of a peri-urban river subject to combined sewer overflow discharges. *Science of the total environment, 539*, 503-514.

Bernal, S., Butturini, A., & Sabater, F. J. B. (2005). Seasonal variations of dissolved nitrogen and DOC: DON ratios in an intermittent Mediterranean stream. *75*, 351-372.

Blumberg, A. F., Ali Khan, L., & St. John, J. P. (1999). Three-dimensional hydrodynamic model of New York Harbor region. *Journal of Hydraulic Engineering, 125*(8), 799 - 816.

Bonnet, M., & Wessen, K. (2001). ELMO, a 3-D water quality model for nutrients and chlorophyll: first application on a lacustrine ecosystem. *Ecological Modelling, 141*(1), 19-33.

Bozinovic, G., Shea, D., Feng, Z., Hinton, D., Sit, T., & Oleksiak, M. F. J. P. o. (2021). PAH-pollution effects on sensitive and resistant embryos: Integrating structure and function with gene expression. *16*(4), e0249432.

Brabec, E., Schulte, S., & Richards, P. L. (2002). Impervious surfaces and water quality: a review of current literature and its implications for watershed planning. *Journal of planning literature, 16*(4), 499-514.

Bristow, L. A., Callbeck, C. M., Larsen, M., Altabet, M. A., Dekaezemacker, J., Forth, M., . . . Lavik, G. J. N. G. (2017). N2 production rates limited by nitrite availability in the Bay of Bengal oxygen minimum zone. *10*(1), 24-29.

Carswell, L. (1976). Appraisal of water resources in the Hackensack River basin, New Jersey. *J Water-Resources Investigations Report, 76*, 74.

Chaudhary, M., Mishra, S., & Kumar, A. (2017). Estimation of water pollution and probability of health risk due to imbalanced nutrients in River Ganga, India. *International Journal of River Basin Management, 15*(1), 53-60.

Chen, Q., Shi, W., Huisman, J., Maberly, S. C., Zhang, J., Yu, J., . . . Yi, Q. J. N. S. R. (2020). Hydropower reservoirs on the upper Mekong River modify nutrient bioavailability downstream. *7*(9), 1449-1457.

Chen, W., Wang, J., Cao, X., Ran, H., Teng, D., Chen, J., . . . Zheng, X. J. E. R. (2021). Possibility of using multiscale normalized difference vegetation index data for the assessment of total suspended solids (TSS) concentrations in surface water: A specific case of scale issues in remote sensing. *194*, 110636.

Chesworth, W. (2008). Encyclopedia of soil science.

Cole, H. V., Lamarca, M. G., Connolly, J. J., & Anguelovski, I. J. J. E. C. H. (2017). Are green cities healthy and equitable? Unpacking the relationship between health, green space and gentrification. *71*(11), 1118-1121.

Cooper, S. R. (1995). Chesapeake Bay watershed historical land use: impact on water quality and diatom communities. *Ecological applications*, 703-723.

Copetti, D., Marziali, L., Viviano, G., Valsecchi, L., Guzzella, L., Capodaglio, A., . . . Mezzanotte, V. J. W. S. (2019). Intensive monitoring of conventional and surrogate quality parameters in a highly urbanized river affected by multiple combined sewer overflows. *19*(3), 953-966.

Council, C. B. N. (2019). Nutrient Pollution in Casco Bay (Fact Sheet).

Crawford, D., Bonnevie, N., Gillis, C., & Wenning, R. (1994). Historical changes in the ecological health of the Newark Bay Estuary, New Jersey. *Ecotoxicology and Environmental Safety*, *29*(3), 276-303.

Crawford, D. W., Bonnevie, N. L., & Wenning, R. J. (1995). Sources of pollution and sediment contamination in Newark Bay, New Jersey. *Ecotoxicology and Environmental Safety*, *30*(1), 85-100.

Dean, J., Webb, J., Jacobsen, G., Chisari, R., & Dresel, P. (2014). Biomass uptake and fire as controls on groundwater solute evolution on a southeast Australian granite: aboriginal land management hypothesis. *Biogeosciences*, *11*(15), 4099-4114.

Del Giudice, D., Zhou, Y., Sinha, E., & Michalak, A. M. (2018). Long-term phosphorus loading and springtime temperatures explain interannual variability of hypoxia in a large temperate lake. *52*(4), 2046-2054.

Downing, J., McClain, M., Twilley, R., Melack, J., Elser, J., Rabalais, N., . . . Soto, D. (1999). The impact of accelerating land-use change on the N-cycle of tropical aquatic ecosystems: current conditions and projected changes. *Biogeochemistry*, *46*(1-3), 109-148.

Downing, J. A., McCauley, E. J. L., & Oceanography. (1992). The nitrogen: phosphorus relationship in lakes. *37*(5), 936-945.

Du, J., & Shen, J. (2016). Water residence time in Chesapeake Bay for 1980–2012. *Journal of Marine Systems, 164*, 101-111.

Dutta, D., Rahman, A., Paul, S., & Kundu, A. J. U. C. (2021). Impervious surface growth and its inter-relationship with vegetation cover and land surface temperature in peri-urban areas of Delhi. *37*, 100799.

Eklund, B. M., Simon, M. A. J. J. o. t. A., & Association, W. M. (2007). Concentration of tetrachloroethylene in indoor air at a former dry cleaner facility as a function of subsurface contamination: a case study. *57*(6), 753-760.

EPA. (2012, March 06, 2012). Phosphorus.

Farnham, D. J., Gibson, R. A., Hsueh, D. Y., McGillis, W. R., Culligan, P. J., Zain, N., & Buchanan, R. (2017). Citizen science-based water quality monitoring: Constructing a large database to characterize the impacts of combined sewer overflow in New York City. *Science of the total environment, 580*, 168-177.

Fillos, J., & Swanson, W. R. (1975). The release rate of nutrients from river and lake sediments. *Journal (Water Pollution Control Federation), 47*(5), 1032-1042.

Fillos, J., & Swanson, W. R. (1975). The release rate of nutrients from river and lake sediments. *Journal (Water Pollution Control Federation)*, 1032-1042.

Fisher, J., Lowther, J., & Shene, C. (2004). Curve and surface interpolation and approximation: knowledge unit and software tool. *ACM SIGCSE Bulletin, 36*(3), 146-150.

Fowler, T., & Dockter, C. J. R. J. (2010). Optimizing remediation strategies for a former dry cleaner site. *20*(4), 83-104.

Foy, R. H. J. P. a., & environment, t. (2005). The return of the phosphorus paradigm: agricultural phosphorus and eutrophication. (phosphorusagric), 911-939.

Friedman, C. L., & Lohmann, R. (2014). Comparing sediment equilibrium partitioning and passive sampling techniques to estimate benthic biota PCDD/F concentrations in Newark Bay, New Jersey (USA). *Environmental Pollution, 186*, 172-179.

Garcia, A. M., Alexander, R. B., Arnold, J. G., Norfleet, L., White, M. J., Robertson, D. M., & Schwarz, G. (2016). Regional effects of agricultural conservation practices on nutrient transport in the Upper Mississippi River Basin. *Environmental science & technology, 50*(13), 6991-7000.

Gaspar, R., Marques, L., Pinto, L., Baeta, A., Pereira, L., Martins, I., . . . Neto, J. M. (2017). Origin here, impact there—The need of integrated management for river basins and coastal areas. *Ecological Indicators, 72*, 794-802.

Gedefaw, M. G., Geli, H. M., & Abera, T. A. J. R. S. (2021). Assessment of rangeland degradation in New Mexico using time series segmentation and residual trend analysis (tss-restrend). *13*(9), 1618.

GESAMP. (1987). *Land/Sea Boundary Flux of Contaminants: Contributions from Rivers*. Retrieved from New York:

Geymen, A. (2016). DETERMINATION OF THE IMPACTS OF LAND USE CHANGE ON WATER BASIN USING REMOTE SENSING AND GEOGRAPHICAL INFORMATION SYSTEM. *ANADOLU UNIVERSITY JOURNAL OF SCIENCE AND TECHNOLOGYï A Applied Sciences and Engineering*, *17*(3), 484-496.

Gong, J., Wang, Y., & Joseph, H. (2022). *Assessing and mitigating transportation infrastructure vulnerability to coastal storm events with the convergence of advanced spatial analysis, infrastructure modeling, and storm surge simulations*.

Graham, L. E., Graham, J. M., & Wilcox, L. W. (2020). *Plant biology*: Upper Saddle River, NJ: Prentice Hall, c2003.

Grizzetti, B., Vigiak, O., Udias, A., Aloe, A., Zanni, M., Bouraoui, F., . . . De Roo, A. J. G. E. C. (2021). How EU policies could reduce nutrient pollution in European inland and coastal waters. *69*, 102281.

Group, N. J. H. D. (2008). *Nutrients Reduction Cost Estimation Study Summary Report*.

Gump, D., Klemm, A., van Westendorp, C., Wood, D., & Doroba, J. (2017). *Hurricane Maria, 2017 Hydrographic Response for Puerto Rico and US Virgin Islands by the NOAA Ship Thomas Jefferson.* Paper presented at the AGU Fall Meeting Abstracts.

Guo, Q., & Lordi, G. P. (2000). Method for quantifying freshwater input and flushing time in estuaries. *Journal of Environmental Engineering, 126*(7), 675-683.

Hallström, E., Röös, E., & Börjesson, P. (2014). Sustainable meat consumption: A quantitative analysis of nutritional intake, greenhouse gas emissions and land use from a Swedish perspective. *Food Policy, 47*, 81-90.

Harianja, A. H., Suoth, A. E., Nazir, E., Saragih, G. S., Fauzi, R., & Hidayat, M. Y. (2019). *Impact of land use and socio-economic changes in water catchment area on total suspended solid (TSS) in Lake Toba* Paper presented at the IOP Conference Series: Earth and Environmental Science.

Hecky, R., Campbell, P., Hendzel, L. J. L., & Oceanography. (1993). The stoichiometry of carbon, nitrogen, and phosphorus in particulate matter of lakes and oceans. *38*(4), 709-724.

Helminen, H., Juntura, E., Koponen, J., Laihonen, P., & Ylinen, H. (1998). Assessing of long-distance background nutrient loading to the Archipelago Sea, northern Baltic, with a hydrodynamic model. *Environmental Modelling & Software, 13*(5), 511-518.

Hopkins, K. G., Bhaskar, A. S., Woznicki, S. A., & Fanelli, R. M. J. H. p. (2020). Changes in event based streamflow magnitude and timing after suburban development with infiltration based stormwater management. *34*(2), 387-403.

Huang, X., Huang, L., & Yue, W. (2003). The characteristics of nutrients and eutrophication in the Pearl River estuary, South China. *Marine Pollution Bulletin, 47*(1), 30-36.

Huo, S., He, Z., Ma, C., Zhang, H., Xi, B., Xia, X., . . . Wu, F. (2019). Stricter nutrient criteria are required to mitigate the impact of climate change on harmful cyanobacterial blooms. *Journal of Hydrology, 569*, 698-704.

Hussain, S., Mubeen, M., Ahmad, A., Akram, W., Hammad, H. M., Ali, M., . . . Research, P. (2020). Using GIS tools to detect the land use/land cover changes during forty years in Lodhran District of Pakistan. *27*, 39676-39692.

Inc, T. S. (2013). *Combined Sewer Overflow/Stormwater Outfall Investigation Quality Assurance Project Plan*.

Inc., T. S. (2013). *CSO/SWO Investigation Quality Assurance Project Plan*. Retrieved from

Irby, I. D., & Friedrichs, M. (2019). Evaluating confidence in the impact of regulatory nutrient reduction on Chesapeake Bay water quality. *Journal of Estuaries Coasts, 42*(1), 16-32.

Jabłońska, E., Wiśniewska, M., Marcinkowski, P., Grygoruk, M., Walton, C. R., Zak, D., . . . Kotowski, W. J. W. (2020). Catchment-scale analysis reveals high cost-effectiveness of wetland buffer zones as a remedy to non-point nutrient pollution in north-eastern Poland. *12*(3), 629.

Jacobs, S., Weeser, B., Breuer, L., Butterbach-Bahl, K., & Rufino, M. (2016). *Identifying the impacts of land use on water and nutrient cycling in the South-West Mau, Kenya.* Paper presented at the EGU General Assembly Conference Abstracts.

Jeong, H., Choi, J. Y., Lee, J., Lim, J., & Ra, K. J. E. P. (2020). Heavy metal pollution by road-deposited sediments and its contribution to total suspended solids in rainfall runoff from intensive industrial areas. *265*, 115028.

Ji, Z.-G. (2017). *Hydrodynamics and water quality: modeling rivers, lakes, and estuaries*: John Wiley & Sons.

Jiang, D., Matsushita, B., Pahlevan, N., Gurlin, D., Lehmann, M. K., Fichot, C. G., . . . Zhang, Y. J. R. s. o. e. (2021). Remotely estimating total suspended solids concentration in clear to extremely turbid waters using a novel semi-analytical method. *258*, 112386.

Jin, S., Homer, C., Yang, L., Danielson, P., Dewitz, J., Li, C., . . . Howard, D. J. R. S. (2019). Overall methodology design for the United States national land cover database 2016 products. *11*(24), 2971.

Johnes, P. J. (1996). Evaluation and management of the impact of land use change on the nitrogen and phosphorus load delivered to surface waters: the export coefficient modelling approach. *Journal of hydrology, 183*(3-4), 323-349.

Jung, H. B. (2017). Nutrients and heavy metals contamination in an urban estuary of Northern New Jersey. *J Geosciences, 7*(4), 108.

Jung, H. B. (2020). Geochemical and hydrological study of coastal groundwater discharging to an urban estuary in northern New Jersey. *J Environmental Earth Sciences, 79*(6), 1-14.

Jung, H. B., Richards, J., & Fitzgerald, A. (2020). Temporal and spatial variations of water quality in the Newark Bay Estuary. *Regional Studies in Marine Science*, 101589.

Kadir, A., Ahmed, Z., Uddin, M., Xie, Z., & Kumar, P. J. W. (2022). Integrated Approach to Quantify the Impact of Land Use and Land Cover Changes on Water Quality of Surma River, Sylhet, Bangladesh. *14*(1), 17.

Kalnejais, L. H., Martin, W. R., & Bothner, M. H. (2010). The release of dissolved nutrients and metals from coastal sediments due to resuspension. *Marine Chemistry, 121*(1), 224-235.

Keiser, D. A., Kling, C. L., & Shapiro, J. S. J. P. o. t. N. A. o. S. (2019). The low but uncertain measured benefits of US water quality policy. *116*(12), 5262-5269.

Khoi, D. N., Loi, P. T., Trang, N. T. T., Vuong, N. D., Fang, S., Nhi, P. T. T. J. E. S., & Research, P. (2022). The effects of climate variability and land-use change on streamflow and nutrient loadings in the Sesan, Sekong, and Srepok (3S) River Basin of the Lower Mekong Basin. *29*(5), 7117-7126.

Kim, K., Kim, B., Knorr, K. H., Eum, J., Choi, Y., Jung, S., & Peiffer, S. (2016). Potential effects of sediment processes on water quality of an artificial reservoir in the Asian monsoon region. *Inland Waters, 6*(3), 423-435.

Kling, C. L., Panagopoulos, Y., Valcu, A. M., Gassman, P. W., Rabotyagov, S. S., Campbell, T., . . . Jha, M. K. (2014). Land Use Model Integrating Agriculture and the Environment (LUMINATE): Linkages between agricultural land use, local water quality and hypoxic concerns in the Gulf of Mexico Basin.

Krauskopf, K. B. (1979). *Introduction to geochemistry*: McGraw-Hill.

Lathrop, R. G., Tulloch, D. L., & Hatfield, C. (2007). Consequences of land use change in the New York–New Jersey Highlands, USA: Landscape indicators of forest and watershed integrity. *Landscape and Urban Planning, 79*(2), 150-159. doi:10.1016/j.landurbplan.2006.02.008

León, L. F., Imberger, J., Smith, R. E., Hecky, R. E., Lam, D. C., & Schertzer, W. M. (2005). Modeling as a tool for nutrient management in Lake Erie: a hydrodynamics study. *Journal of Great Lakes Research, 31*, 309-318.

Li, M., Lee, Y. J., Testa, J. M., Li, Y., Ni, W., Kemp, W. M., & Di Toro, D. M. (2016). What drives interannual variability of hypoxia in Chesapeake Bay: Climate forcing versus nutrient loading? *Geophysical Research Letters, 43*(5), 2127-2134.

Li, S., Liu, W., Gu, S., Cheng, X., Xu, Z., & Zhang, Q. (2009). Spatio-temporal dynamics of nutrients in the upper Han River basin, China. *Journal of hazardous materials, 162*(2), 1340-1346.

Li, Y.-n., Li, J., Shao, Z., Duan, Z., Xie, Y., Cui, Z., . . . Li, S. J. W. S. (2020). Quantitative analysis of 209 polychlorinated biphenyl (PCB) congeners in sediments of the Raritan River estuary, USA. *20*(6), 2400-2414.

Li, Y., Feng, H., Zhang, H., Sun, J., Yuan, D., Guo, L., . . . Du, J. (2019). Hydrodynamics and water circulation in the New York/New Jersey Harbor: A study from the perspective of water age. *Journal of Marine Systems, 199*, 103219.

Li, Z., Wang, S., Song, S., Wang, Y., & Musakwa, W. J. J. o. A. E. (2021). Detecting land degradation in Southern Africa using time series segment and residual trend (TSS-RESTREND). *184*, 104314.

Lintern, A., McPhillips, L., Winfrey, B., Duncan, J., Grady, C. J. E. S., & Technology. (2020). Best management practices for diffuse nutrient pollution: wicked problems across urban and agricultural watersheds. *54*(15), 9159-9174.

Loehr, R. C. J. J. (1974). Characteristics and comparative magnitude of non-point sources. 1849-1872.

Lucash, M. S., Scheller, R. M., Kretchun, A. M., Clark, K. L., & Hom, J. (2014). Impacts of fire and climate change on long-term nitrogen availability and forest productivity in the New Jersey Pine Barrens. *Canadian journal of forest research, 44*(5), 404-412.

Ludwig, W., Dumont, E., Meybeck, M., & Heussner, S. (2009). River discharges of water and nutrients to the Mediterranean and Black Sea: major drivers for ecosystem changes during past and future decades? *Progress in Oceanography, 80*(3), 199-217.

Lung, W.-S. (1986). Assessing phosphorus control in the James River Basin. *Journal of Environmental Engineering, 112*(1), 44-60.

Lung, W.-S., & Testerman, N. (1989). Modeling Fate and Transport of Nutrients on the James Estuary. *Journal of Environmental Engineering, 115*(5), 978-991.

Ly, K., Metternicht, G., & Marshall, L. (2020). Linking Changes in Land Cover and Land Use of the Lower Mekong Basin to Instream Nitrate and Total Suspended Solids Variations. *Sustainability, 12*(7). doi:10.3390/su12072992

Malagó, A., Bouraoui, F., Vigiak, O., Grizzetti, B., & Pastori, M. (2017). Modelling water and nutrient fluxes in the Danube River Basin with SWAT. *J Science of The Total Environment, 603*, 196-218.

Malherbe, H., Le Maitre, D., Le Roux, J., Pauleit, S., & Lorz, C. (2019). A simplified method to assess the impact of sediment and nutrient inputs on river water quality in two regions of the southern coast of South Africa. *J Environmental management, 63*(5), 658-672.

Masi, F., Rizzo, A., Bresciani, R., & Conte, G. (2017). Constructed wetlands for combined sewer overflow treatment: Ecosystem services at Gorla Maggiore, Italy. *Ecological Engineering, 98*, 427-438.

Mauriello, M. N. (2009). New Jersey Nutrient Criteria Enhancement Plan.

Mehdi, B., Lehner, B., Gombault, C., Michaud, A., Beaudin, I., Sottile, M. F., & Blondlot, A. (2015). Simulated impacts of climate change and agricultural land use change on surface water quality with and without adaptation management strategies. *Agriculture, Ecosystems & Environment, 213*, 47-60. doi:10.1016/j.agee.2015.07.019

Mehdi, B., Schürz, C., Grath, B., & Schulz, K. J. S. o. T. T. E. (2021). Storm event impacts on in-stream nitrate concentration and discharge dynamics: A comparison of high resolution in-situ measured data with model simulations. *755*, 143406.

Melton, F., Xiong, J., Wang, W., Milesi, C., Li, S., Quackenbush, A., . . . Nemani, R. (2016). Potential Impacts of Climate and Land Use Change on Ecosystem Processes in the Great Northern and Appalachian Landscape Conservation Cooperatives. In *Climate Change in Wildlands* (pp. 119-150): Springer.

Meshesha, T. W., Wang, J., & Melaku, N. D. J. J. o. H. (2020). Modelling spatiotemporal patterns of water quality and its impacts on aquatic ecosystem in the cold climate region of Alberta, Canada. *587*, 124952.

Miao, S., DeLaune, R., & Jugsujinda, A. (2006). Influence of sediment redox conditions on release/solubility of metals and nutrients in a Louisiana Mississippi River deltaic plain freshwater lake. *Science of the Total Environment, 371*(1), 334-343.

Mishra, P. K., Rai, A., Rai, S. C. J. T. E. J. o. R. S., & Science, S. (2020). Land use and land cover change detection using geospatial techniques in the Sikkim Himalaya, India. *23*(2), 133-143.

Monsen, N. E., Cloern, J. E., Lucas, L. V., & Monismith, S. (2002). A comment on the use of flushing time, residence time, and age as transport time scales. *Limnology & Oceanography, 47*(5), 1545-1553.

Morgan, D., Xiao, L., & McNabola, A. (2017). Evaluation of combined sewer overflow assessment methods: case study of Cork City, Ireland. *Water and Environment Journal*.

Moriarty, J., Harris, C. K., Fennel, K., Xu, K., Rabouille, C., & Friedrichs, M. A. (2017). A Model Archive for a Coupled Hydrodynamic-Sediment Transport-Biogeochemistry Model for the Rhône River Sub-aqueous Delta, France.

Mousa, A. A., Hussein, M., & Kineber, A. F. J. T. R. R. (2022). Value-Engineering Methodology for the Selection of an Optimal Bridge System. *2676*(4), 483-498.

Moustafa, M., & Hamrick, J. (2000). Calibration of the wetland hydrodynamic model to the Everglades Nutrient Removal Project. *Water Quality and Ecosystems Modeling, 1*(1), 141-167.

Mulholland, P. J. (1992). Regulation of nutrient concentrations in a temperate forest stream: roles of upland, riparian, and instream processes. *Limnology and Oceanography, 37*(7), 1512-1526.

Myklestad, S. J. J. o. E. M. B., & Ecology. (1977). Production of carbohydrates by marine planktonic diatoms. II. Influence of the NP ratio in the growth medium on the assimilation ratio, growth rate, and production of cellular and extracellular carbohydrates by Chaetoceros affinis var. willei (Gran) Hustedt and Skeletonema costatum (Grev.) Cleve. *29*(2), 161-179.

Naikoo, M. W., Rihan, M., & Ishtiaque, M. J. J. o. U. M. (2020). Analyses of land use land cover (LULC) change and built-up expansion in the suburb of a metropolitan city: Spatio-temporal analysis of Delhi NCR using landsat datasets. *9*(3), 347-359.

Nauman, E. B. (2008). Residence time theory. *Industrial Engineering Chemistry Research, 47*(10), 3752-3766.

Nkwanda, I., Feyisa, G., Zewge, F., Makwinja, R. J. I. J. o. E., & Resources, W. (2021). Impact of land-use/land-cover dynamics on water quality in the Upper Lilongwe River basin, Malawi. *5*, 193-204.

Nobre, R. L. G., Caliman, A., Cabral, C. R., de Carvalho Araújo, F., Guerin, J., Dantas, F. d. C. C., . . . Amado, A. M. J. S. o. t. T. E. (2020). Precipitation, landscape properties and land use interactively affect water quality of tropical freshwaters. *716*, 137044.

Ofiara, D. D. (2015). The New York Bight 25years later: Use impairments and policy challenges. *Marine Pollution Bulletin, 90*(1), 281-298.

Olson, K. R., & Tharp, M. (2020). How did the Passaic River, a Superfund site near Newark, New Jersey, become an Agent Orange dioxin TCDD hotspot? *Journal of Soil and Water Conservation, 75*(2), 33A-37A. doi:10.2489/jswc.75.2.33A

Olson, K. R., Tharp, M. J. J. o. S., & Conservation, W. (2020). How did the Passaic River, a Superfund site near Newark, New Jersey, become an Agent Orange dioxin TCDD hotspot? , *75*(2), 33A-37A.

Ophori, D., Firor, C., & Soriano, P. J. E. E. S. (2019). Impact of road deicing salts on the Upper Passaic River Basin, New Jersey: a geochemical analysis of the major ions in groundwater. *78*, 1-13.

Osman, M., Wan Yusof, K., Takaijudin, H., Goh, H. W., Abdul Malek, M., Azizan, N. A., . . . Sa'id Abdurrasheed, A. J. S. (2019). A review of nitrogen removal for urban stormwater runoff in bioretention system. *11*(19), 5415.

Oteng Mensah, F., & Alo, C. A. (2023). Modeling monthly actual evapotranspiration: an application of geographically weighted regression technique in the Passaic River Basin. *Journal of Water and Climate Change, 14*(1), 17-37. doi:10.2166/wcc.2022.111

Oteng Mensah, F., Alo, C. A. J. J. o. W., & Change, C. (2023). Modeling monthly actual evapotranspiration: an application of geographically weighted regression technique in the Passaic River Basin. *14*(1), 17-37.

Pálfy, T., Molle, P., Langergraber, G., Troesch, S., Gourdon, R., & Meyer, D. (2016). Simulation of constructed wetlands treating combined sewer overflow using HYDRUS/CW2D. *Ecological Engineering, 87*, 340-347.

Parette, R., & Pearson, W. N. (2014). 2, 4, 6, 8-Tetrachlorodibenzothiophene in the Newark Bay Estuary: The likely source and reaction pathways. *Chemosphere, 111*, 157-163.

Park, K., Jung, H.-S., Kim, H.-S., & Ahn, S.-M. (2005). Three-dimensional hydrodynamic-eutrophication model (HEM-3D): application to Kwang-Yang Bay, Korea. *Marine environmental research, 60*(2), 171-193.

Perillo, G. M. E., & Piccolo, M. C. (1991). An interpolation method for estuarine and oceanographic data. *Computers & Geosciences, 17*(6), 813-820.

Proctor, R., Holt, J. T., Allen, J. I., & Blackford, J. (2003). Nutrient fluxe
s and budgets for the North West European Shelf from a three-dimensional model. *Science of the total environment, 314*, 769-785.

Protection, N. J. D. o. E. (2000). *CSO communites cost*. http://www.wnjpin.net/OneStopCareerCenter/LaborMarketInformation/lmi25/pub/index.html.

Qualls, R. G., & Haines, B. L. (1991). Geochemistry of Dissolved Organic Nutrients in Water Percolating through a Forest Ecosystem. *Soil Science Society of America Journal, 55*(4), 1112-1123. doi:10.2136/sssaj1991.03615995005500040036x

Quijano, J. C., Zhu, Z., Morales, V., Landry, B. J., & Garcia, M. H. (2017). Three-dimensional model to capture the fate and transport of combined sewer overflow discharges: A case study in the Chicago Area Waterway System. *Science of the total environment, 576*, 362-373.

Rattan, K., Corriveau, J., Brua, R., Culp, J., Yates, A., & Chambers, P. (2017). Quantifying seasonal variation in total phosphorus and nitrogen from prairie streams in the Red River Basin, Manitoba Canada. *Science of The Total Environment, 575*, 649-659.

Reemtsma, T., Gnirß, R., & Jekel, M. (2000). Infiltration of combined sewer overflow and tertiary municipal wastewater: an integrated laboratory and field study on nutrients and dissolved organics. *Water research, 34*(4), 1179-1186.

Reinfelder, J. R., & Janssen, S. E. J. J. o. h. m. (2019). Tracking legacy mercury in the Hackensack River estuary using mercury stable isotopes. *375*, 121-129.

Rio, M., Salles, C., Cernesson, F., Marchand, P., & Tournoud, M.-G. J. J. o. H. (2020). An original urban land cover representation and its effects on rain event-based runoff and TSS modelling. *586*, 124865.

Romero, J., Antenucci, J., & Imberger, J. (2004). One-and three-dimensional biogeochemical simulations of two differing reservoirs. *Ecological Modelling, 174*(1), 143-160.

Ruan, Z., Kuang, Y., He, Y., Zhen, W., & Ding, S. J. R. S. (2020). Detecting Vegetation Change in the Pearl River Delta Region Based on Time Series Segmentation and Residual Trend Analysis (TSS-RESTREND) and MODIS NDVI. *12*(24), 4049.

Saba, T., & Su, S. (2013). Tracking polychlorinated biphenyls (PCBs) congener patterns in Newark Bay surface sediment using principal component analysis (PCA) and positive matrix factorization (PMF). *Journal of hazardous materials, 260*, 634-643.

Saleh, F., Ramaswamy, V., Wang, Y., Georgas, N., Blumberg, A., & Pullen, J. (2017). A multi-scale ensemble-based framework for forecasting compound coastal-riverine flooding: The Hackensack-Passaic watershed and Newark Bay. *Advances in Water Resources, 110*, 371-386.

Schreiber, J. S., House, S., Prohonic, E., Smead, G., Hudson, C., Styk, M., & Lauber, J. J. R. A. (1993). An investigation of indoor air contamination in residences above dry cleaners. *13*(3), 335-344.

Serajuddin, M., Chowdhury, A., Haque, M. M., & Haque, M. E. (2019). *Using turbidity to determine total suspended solids in an urban stream: a case study.* Paper presented at the

Proceedings of the 2nd International Conference on Water and Environmental Engineering, Dhaka.

Shi, Y., Zhang, L., Li, Y., Zhou, L., Zhou, Y., Zhang, Y., . . . Zhu, G. J. S. o. t. T. E. (2020). Influence of land use and rainfall on the optical properties of dissolved organic matter in a key drinking water reservoir in China. *699*, 134301.

Shin, J. Y., Artigas, F., Hobble, C., & Lee, Y.-S. (2013). Assessment of anthropogenic influences on surface water quality in urban estuary, northern New Jersey: multivariate approach. *Environmental monitoring and assessment, 185*(3), 2777-2794.

Shin, J. Y., Artigas, F., Hobble, C., Lee, Y.-S., & assessment. (2013). Assessment of anthropogenic influences on surface water quality in urban estuary, northern New Jersey: multivariate approach. *J Environmental monitoring, 185*(3), 2777-2794.

Slattery, K. (2022). The Raritan River.

Stineman, R. W. (1980). A consistently well-behaved method of interpolation. *Creative Computing, 6*, 54-57.

Subalusky, A. L., Dutton, C. L., Njoroge, L., Rosi, E. J., & Post, D. M. (2018). Organic matter and nutrient inputs from large wildlife influence ecosystem function in the Mara River, Africa. *Ecology, 99*(11), 2558-2574.

Sunardi, S., Nursamsi, I., Dede, M., Paramitha, A., Arief, M. C. W., Ariyani, M., . . . Indonesia, T. (2022). Assessing the influence of land-use changes on water quality using remote sensing and GIS: A Study in Cirata Reservoir, Indonesia. *7*(1), 106-114.

Tan, J. F., Anastasi, A., & Chandra, S. J. C. O. i. E. (2022). Electrochemical detection of nitrate, nitrite and ammonium for on-site water quality monitoring. *32*, 100926.

Tang, J., Li, X., Cao, C., Lin, M., Qiu, Q., Xu, Y., & Ren, Y. J. E. I. (2019). Compositional variety of dissolved organic matter and its correlation with water quality in peri-urban and urban river watersheds. *104*, 459-469.

Teklu, B. M., Hailu, A., Wiegant, D. A., Scholten, B. S., & Van den Brink, P. J. (2018). Impacts of nutrients and pesticides from small-and large-scale agriculture on the water quality of Lake Ziway, Ethiopia. *Environmental Science & Pollution Research, 25*(14), 13207-13216.

Tengberg, A., Almroth, E., & Hall, P. (2003). Resuspension and its effects on organic carbon recycling and nutrient exchange in coastal sediments: in situ measurements using new experimental technology. *Journal of Experimental Marine Biology and Ecology, 285*, 119-142.

Testa, J. M., Li, Y., Lee, Y. J., Li, M., Brady, D. C., Di Toro, D. M., . . . Fitzpatrick, J. J. (2014). Quantifying the effects of nutrient loading on dissolved O 2 cycling and hypoxia in Chesapeake Bay using a coupled hydrodynamic–biogeochemical model. *Journal of marine systems, 139*, 139-158.

Trang, N. T. T., Shrestha, S., Shrestha, M., Datta, A., & Kawasaki, A. (2017). Evaluating the impacts of climate and land-use change on the hydrology and nutrient yield in a transboundary river basin: A case study in the 3S River Basin (Sekong, Sesan, and Srepok). *Science of the total environment, 576*, 586-598.

Tsegaye, B. J. I. J. o. A. S., & Technology, F. (2019). Effect of land use and land cover changes on soil erosion in Ethiopia. *5*(1), 26-34.

USACE. (2010). *US Army Corps of Engineers: Newark Bay*. Retrieved from

Ustaoğlu, F., Tepe, Y., & Taş, B. J. E. i. (2020). Assessment of stream quality and health risk in a subtropical Turkey river system: A combined approach using statistical analysis and water quality index. *113*, 105815.

Varol, M. (2020). Use of water quality index and multivariate statistical methods for the evaluation of water quality of a stream affected by multiple stressors: A case study. *Environ Pollut, 266*(Pt 3), 115417. doi:10.1016/j.envpol.2020.115417

Villa, A., Fölster, J., Kyllmar, K. J. E. m., & assessment. (2019). Determining suspended solids and total phosphorus from turbidity: comparison of high-frequency sampling with conventional monitoring methods. *191*, 1-16.

Vitousek, P. M., Aber, J. D., Howarth, R. W., Likens, G. E., Matson, P. A., Schindler, D. W., . . . Tilman, D. G. (1997). Human alteration of the global nitrogen cycle: sources and consequences. *Ecological applications, 7*(3), 737-750.

Walker, W. J., McNutt, R. P., & Maslanka, C. K. (1999). The potential contribution of urban runoff to surface sediments of the Passaic River: sources and chemical characteristics. *Chemosphere, 38*(2), 363-377.

Walton, C. R., Zak, D., Audet, J., Petersen, R. J., Lange, J., Oehmke, C., . . . Jabłońska, E. J. S. o. t. T. E. (2020). Wetland buffer zones for nitrogen and phosphorus retention: Impacts of soil type, hydrology and vegetation. *727*, 138709.

Wang, S. W., Gebru, B. M., Lamchin, M., Kayastha, R. B., & Lee, W.-K. J. S. (2020). Land use and land cover change detection and prediction in the Kathmandu district of Nepal using remote sensing and GIS. *12*(9), 3925.

Wang, Y., Gong, J., & Di, C. J. A. P. (2022). A building-scale hydrodynamic model for extreme urban flash flooding simulation: A Confluence Area in Raritan River Basin during Hurricane Ida.

Weigel, D., Vilhena, L., Woods, P., Tonina, D., Tranmer, A., Benjankar, R., . . . Goodwin, P. (2017). Aquatic habitat response to climate-driven hydrologic regimes and water operations in a montane reservoir in the Pacific Northwest, USA. *Aquatic Sciences*, 1-14.

Wickham, J. D., Stehman, S. V., Gass, L., Dewitz, J., Fry, J. A., & Wade, T. G. (2013). Accuracy assessment of NLCD 2006 land cover and impervious surface. *Remote Sensing of Environment*, *130*, 294-304.

Wieczerak, T., Wolde, B., Lal, P., Witherell, B., & Deng, Y. J. M. S. G. (2020). Public Perception and willingness to pay for green infrastructure improvements in northern New Jersey. *53*, 32-42.

Wijesiri, B., Liu, A., Deilami, K., He, B., Hong, N., Yang, B., . . . Goonetilleke, A. J. E. P. (2019). Nutrients and metals interactions between water and sediment phases: An urban river case study. *251*, 354-362.

Winfield, C. H. (1923). *"Saddle River, New Jersey." History of Bergen County, New Jersey, 1630-1923*.

Wu, G., & Xu, Z. (2011). Prediction of algal blooming using EFDC model: Case study in the Daoxiang Lake. *Ecological Modelling*, *222*(6), 1245-1252.

Wurtsbaugh, W. A., Paerl, H. W., & Dodds, W. K. J. W. I. R. W. (2019). Nutrients, eutrophication and harmful algal blooms along the freshwater to marine continuum. *6*(5), e1373.

Xu, G., Li, P., Lu, K., Tantai, Z., Zhang, J., Ren, Z., . . . Cheng, Y. J. C. (2019). Seasonal changes in water quality and its main influencing factors in the Dan River basin. *173*, 131-140.

Yang, S., Büttner, O., Kumar, R., Jäger, C., Jawitz, J. W., Rao, P., & Borchardt, D. (2019). Spatial patterns of water quality impairments from point source nutrient loads in Germany's largest national River Basin (Weser River). *Science of The Total Environment*, *697*, 134145.

Yu, X., Lee, Z., Shen, F., Wang, M., Wei, J., Jiang, L., & Shang, Z. J. R. S. o. E. (2019). An empirical algorithm to seamlessly retrieve the concentration of suspended particulate matter from water color across ocean to turbid river mouths. *235*, 111491.

Zeng, J., Han, G., & Yang, K. J. J. o. C. P. (2020). Assessment and sources of heavy metals in suspended particulate matter in a tropical catchment, northeast Thailand. *265*, 121898.

Zhang, J., Li, S., Dong, R., Jiang, C., & Ni, M. J. J. o. C. P. (2019). Influences of land use metrics at multi-spatial scales on seasonal water quality: a case study of river systems in the Three Gorges Reservoir Area, China. *206*, 76-85.

Zhang, W. G., Wilkin, J. L., & Schofield, O. M. E. (2010). Simulation of Water Age and Residence Time in New York Bight. *Journal of Physical Oceanography, 40*(5), 965-982. doi:10.1175/2009jpo4249.1

Zhao, T., Poe, G. L., & Boisvert, R. N. (2015). *Management Areas and Fixed Costs in the Economics of Water Quality Trading*.

Zhu, W., Tian, Y. Q., Yu, Q., & Becker, B. L. (2013a). Using Hyperion imagery to monitor the spatial and temporal distribution of colored dissolved organic matter in estuarine and coastal regions. *Remote Sensing of Environment*, *134*, 342-354.

Zhu, W., Tian, Y. Q., Yu, Q., & Becker, B. L. J. R. S. o. E. (2013b). Using Hyperion imagery to monitor the spatial and temporal distribution of colored dissolved organic matter in estuarine and coastal regions. *134*, 342-354.

Figures

Figure 1-1 The major rivers and their water quality stations in the study area.

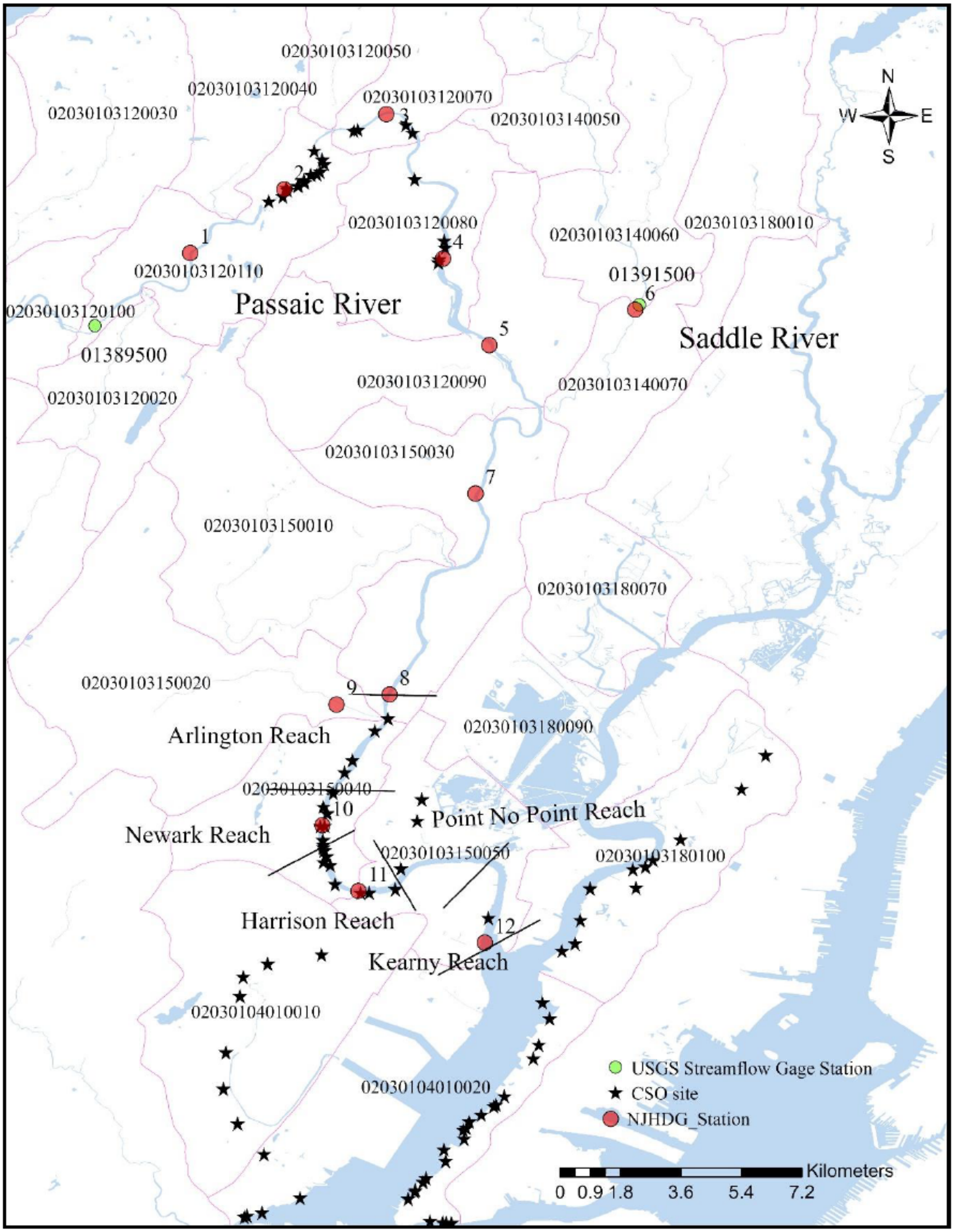


Figure 1-2 Study area map in Passaic River with NJHDG water quality monitoring stations and USGS flow sites and CSOs

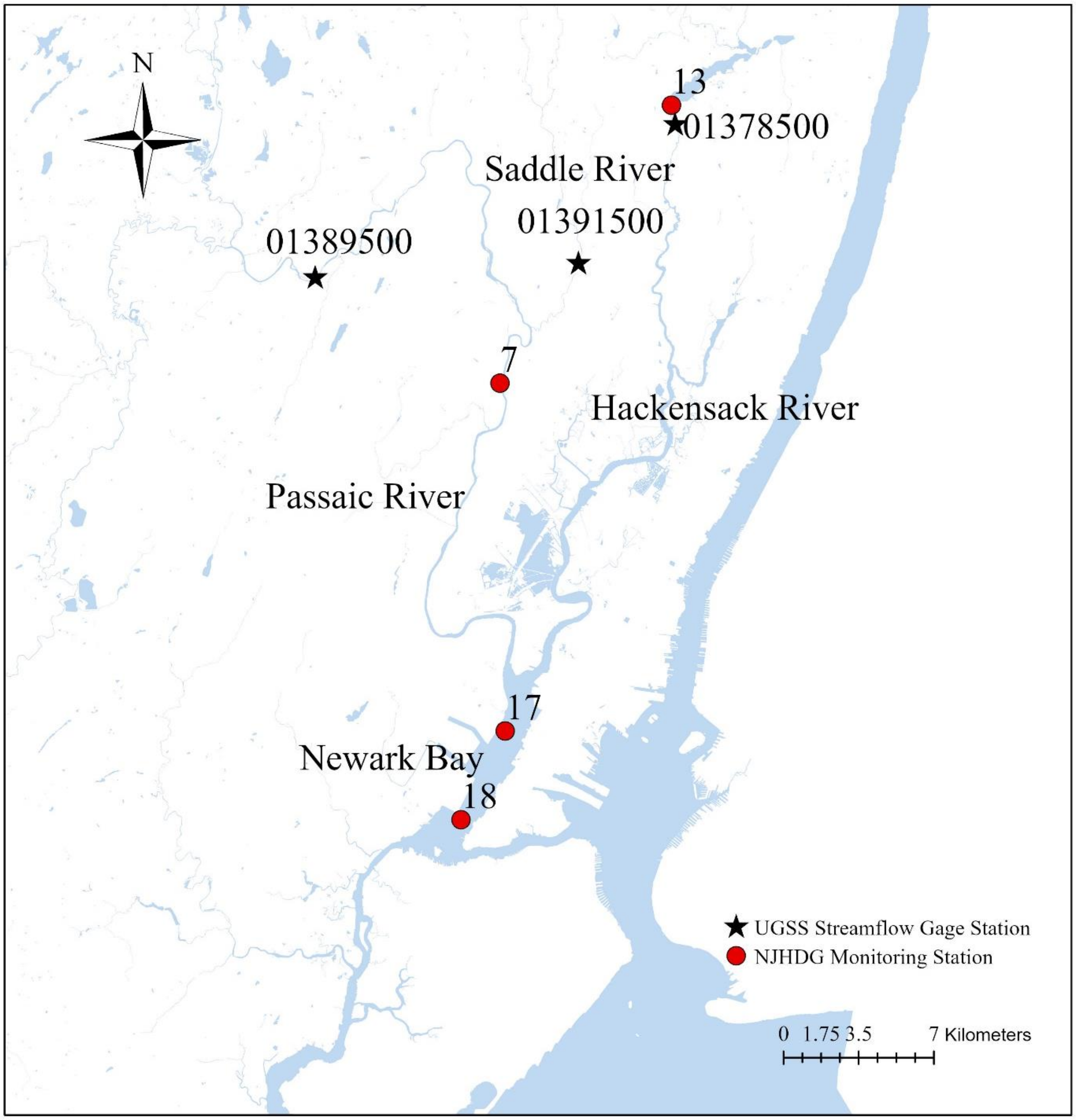
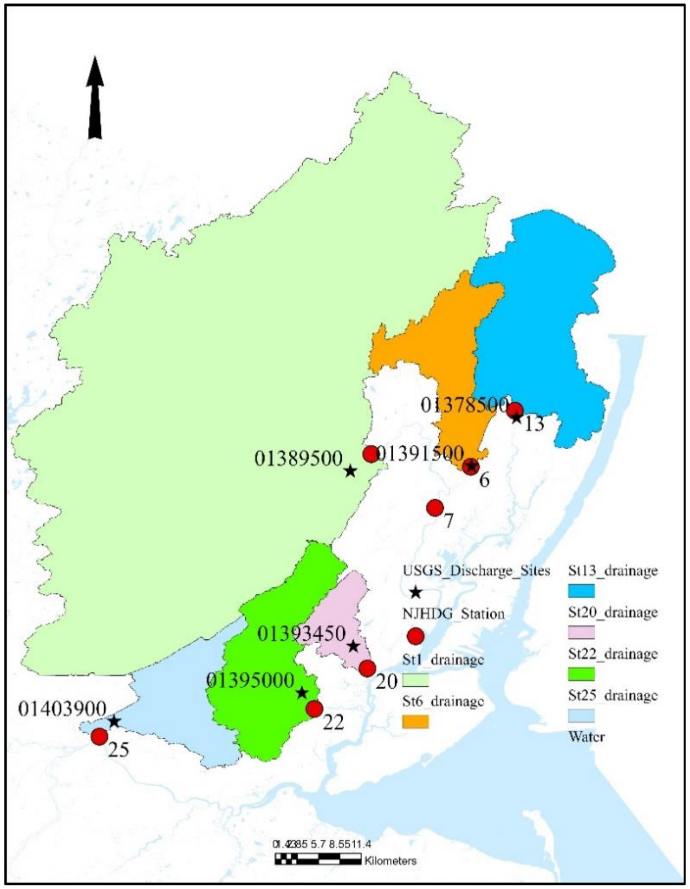
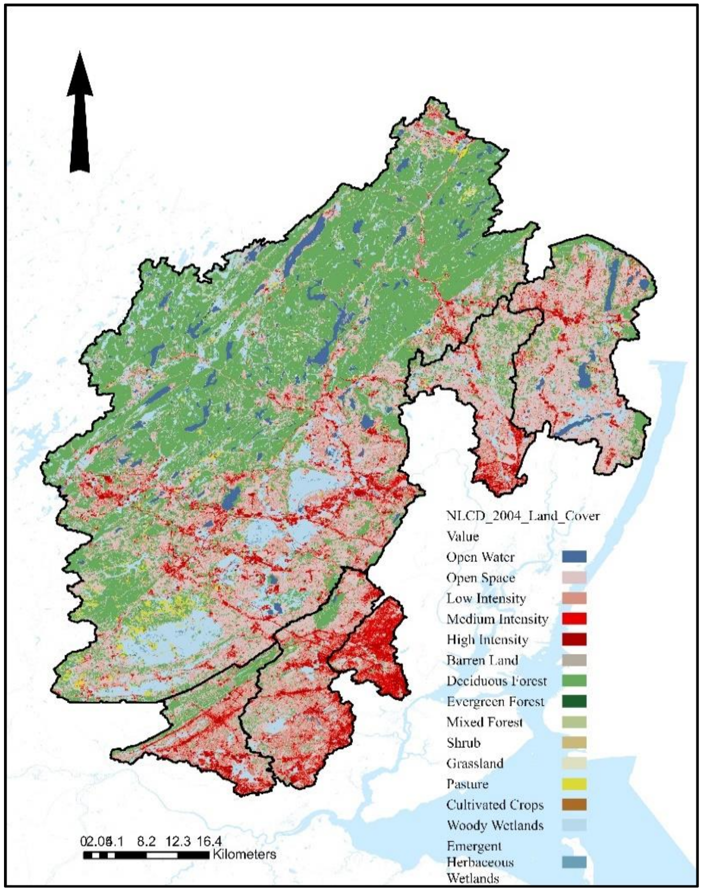


Figure 1-3 Map showing NJHDG monitoring stations and USGS streamflow gaging stations in the Passaic River, the Hackensack River, Saddle River, and Newark Bay. Round dots represent the NJHDG water monitoring stations. Star dots indicated the United States Geological Survey (USGS) gage stations.



a



b

Figure 1-4 The study area of North New Jersey displaying (a) NJHDG water quality and USGS discharge stations as well as drainage basin of each water quality stations, (b) a 2004 land use/land cover (LULC) map of drainage basin.

Figure 1-5 Annual-averaged nutrient (TIN and ortho-P) loadings from the Passaic River measured at St 7 and the Hackensack River measured at St 13 (nutrient load in log10 scale) from 2004 to 2019.

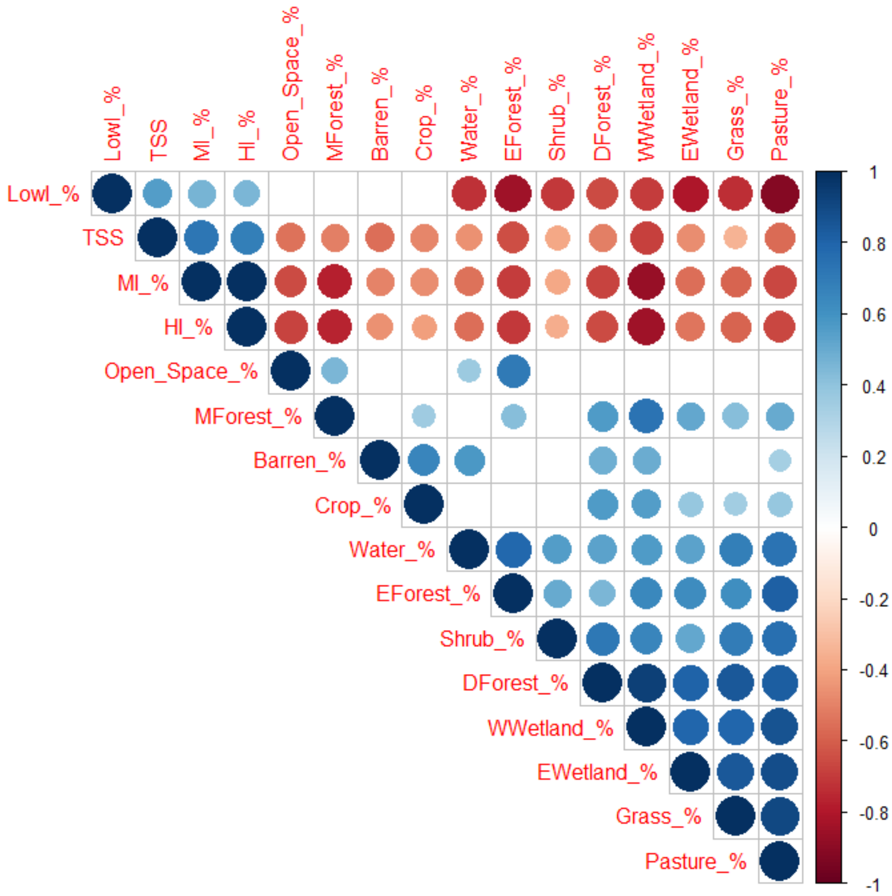


Figure 1-6 Pearson's correlation matrix among different land use type and water quality indicators with significance (p)<0.05.

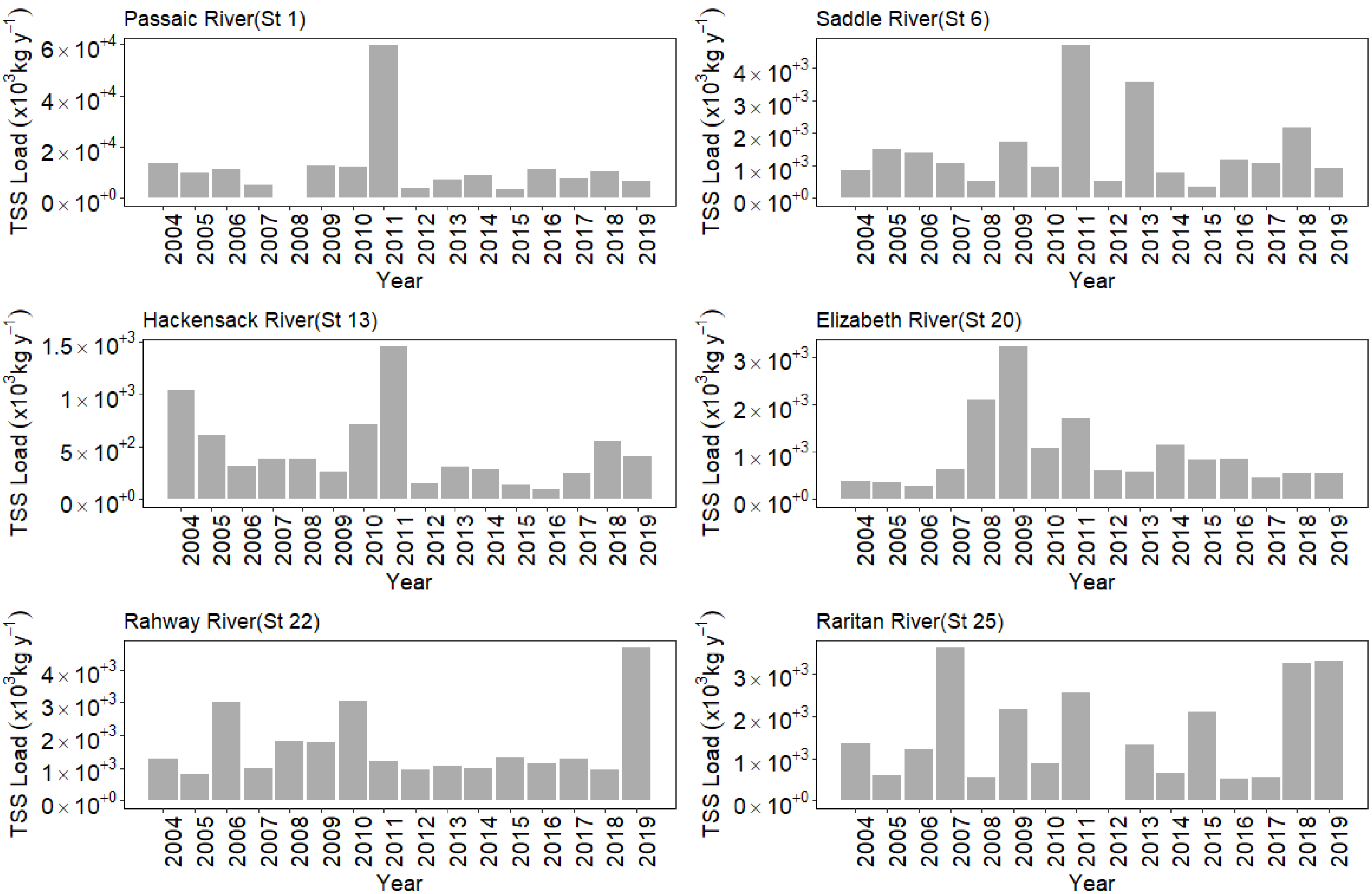


Figure 1-7 Annual-averaged TSS loadings from the six rivers from 2004 to 2019.

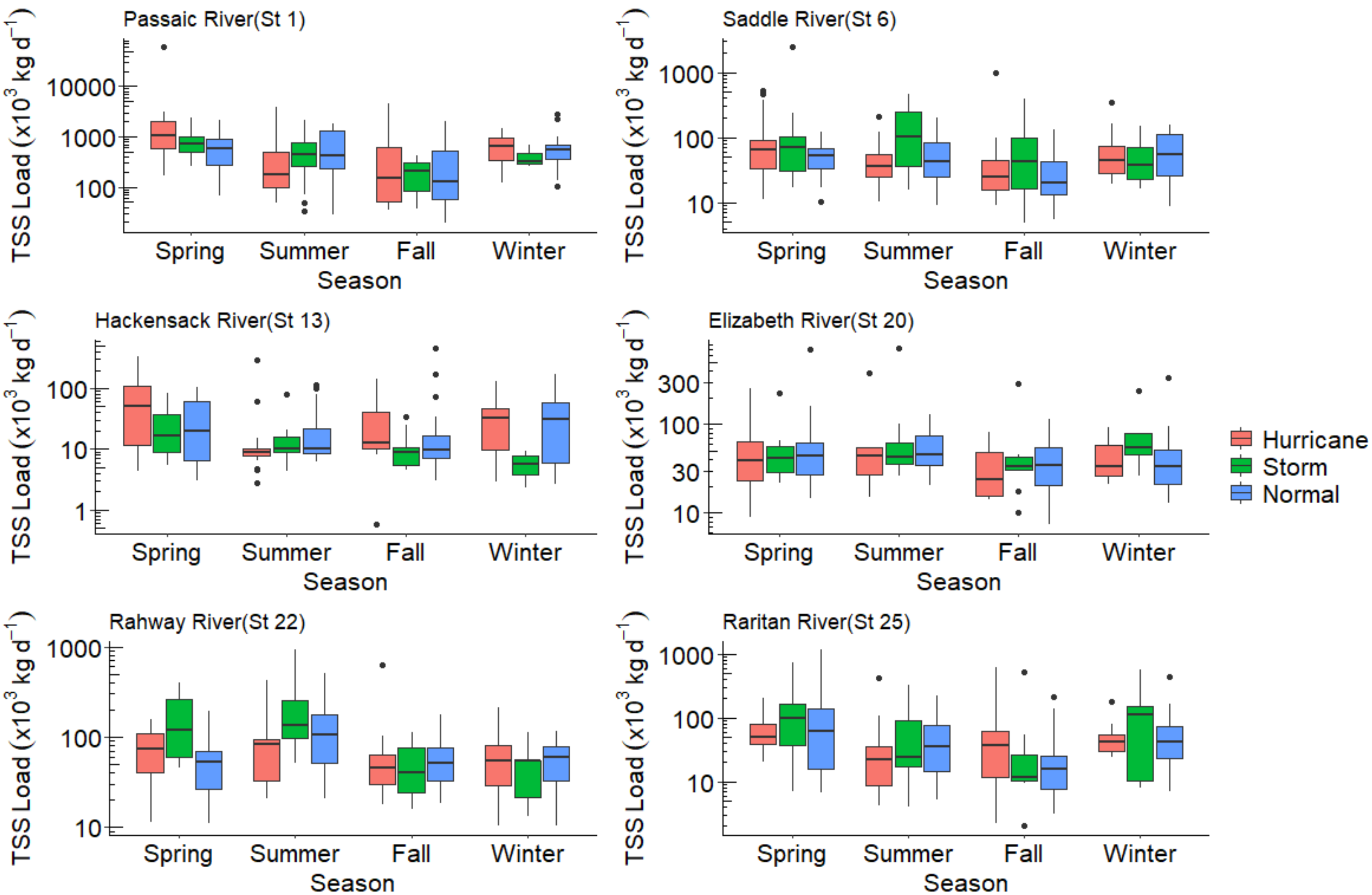


Figure 1-8 Seasonal variations of TSS loadings under hurricane, storm and normal weather conditions in six rivers from 2004 to 2019. Note: Normal weather are years without any hurricane or storm event and hurricane conditions are hurricane occurrence year.

Tables

Table 1-1 Water quality stations, discharge sites and river area for this study

River Name	Water Quality Monitoring Station No.	Flow Stations (USGS sites)
Passaic River	1	01389500
Saddle River	6	01391500
Hackensack River	13	01378500
Elizabeth River	20	01393450
Rahway River	22	01395000
Raritan River	25	01403900

Chapter 2 Causes, Assessment and Treatment of Nutrient (N and P) Pollution in Rivers, Estuaries and Coastal Waters

Jing Nie [1], Huan Feng [1, *], Benjamin B. Witherell [2], Marzooq Alebus [2], Manoj D. Mahajan [3], Weiguo Zhang [4], Lizhong Yu [4]

1. Department of Earth and Environmental Studies, Montclair State University, Montclair, New Jersey 07043, USA
2. New Jersey Department of Environmental Protection, Trenton, New Jersey 08625, USA
3. Department of Technology and Society, Stony Brook University, Stony Brook, New York 11794, USA
4. State Key Laboratory of Estuarine and Coastal Research, East China Normal University, Shanghai 200062, PRC

*Corresponding author, fengh@montclair.edu

Abstract

As a consequence of industrialization, urbanization, and population growth in the past decades, high nutrient concentrations from point and non-point sources in aquatic systems have caused major problems to the water quality in rivers, estuaries and coastal waters Although the nutrient pollution due to land use change cannot be ignored, the combined sewer overflows and discharging sites have been important point sources of nutrient pollution. Integrated hydrodynamic, chemical and biological models developed in recent years, which simulate the nutrient transportation from both point and non-point sources, are useful tools to assist in identifying the transport and fate of nutrients from both point and non-point sources. In this paper, water quality data from published literature were reviewed and analyzed to evaluate nutrient (N and P) pollution in aquatic systems. An integrated monitoring and management plan should be continuously developed in future to monitor and regulate nutrient discharges from point and non-point sources.

Keywords: nutrient pollution; nitrogen; phosphorus; combined sewer overflows

1. Introduction

Nutrient contamination in waterbodies and waterways is a serious environmental problem in many countries because water quality is vital to human health, ecosystems, and environment, and can be affected by nutrient (N and P) concentrations due to both natural processes and anthropogenic activities (Shin, Artigas, Hobble, & Lee, 2013). Although newly urbanized land can bring economic profits to businesses and government, it can also cause damage to the ecosystem (Lathrop et al., 2007). Urbanization and agricultural land use usually cause high nutrient concentrations in the water body (S. Li et al., 2009). Thus, nutrient (N and P) concentrations are highly impacted by the rapid land use change and expansion in urban coastal areas, which cause non-point source nutrient pollution. In the United States, urban development both impacts and causes environmental changes. Early studies indicate that urban or agricultural storm water runoff and wastewater discharges can cause an increase in nutrient concentrations (John Fillos & William R Swanson, 1975). For example, the lower Passaic River and Newark Bay in the USA have suffered from severe chemical, metal, and nutrient pollution for decades (Parette & Pearson, 2014). Most of the nutrients were from publicly owned treatment works (POTWs) and combined sewer overflows (CSOs) (D. W. Crawford, Bonnevie, & Wenning, 1995). Both water and sediment quality data showed that the biodiversity and natural resource abundance in these areas were reduced significantly, and thus result in water quality degradation (D. Crawford, Bonnevie, Gillis, & Wenning, 1994). Therefore, thousands of water quality restoration projects have been conducted for the purpose to deal with water quality issue (Ofiara, 2015).

In coastal areas, estuaries are a transition zone between river environments and maritime environments, home to unique plant and animal communities, and vulnerable to nutrient pollution when the polluted stream water passes through these areas into coastal waters. In nitrogen cycle among atmosphere, land, sea and sediments, atmospheric deposition of inorganic nitrogen (NO, NO_2, and NH_3) is an important source to the ocean, although the input rate is still uncertain (Appelo & Postma, 2004). In contrast, riverine input has an insignificant impact on the ocean water during the denitrification process in the coastal zone (Krauskopf, 1979). It was reported that the marine biota only contains less than 0.05% of reactive nitrogen while land biota contains 5% of the terrestrial nitrogen that is present largely in the soil (Chesworth, 2008). However, human alterations have doubled reactive nitrogen input into the terrestrial nitrogen cycle, increased N_2O emission, caused losses of soil nutrients, and greatly increased the transfer of nitrogen through rivers to estuaries and coastal oceans (Vitousek et al., 1997). Anthropogenic influence, such as combustion of fossil fuels routinely used in agricultural and industrial practices, can affect nitrogen cycle and introduce a large quantity of reactive nitrogen (Nr) into water, air and land, which causes health risks to human beings. Many factors such as sediment redox condition change and iron (Fe) reduction in the waterbody can affect the nutrient concentrations in aquatic environment (Miao et al., 2006). Previous studies show that seasonal change with a different precipitation pattern can also affect nutrient concentrations (Friedman & Lohmann, 2014; Parette & Pearson, 2014; Saba & Su, 2013). Due to its unique environmental settings and characteristics in different riverine and estuarine systems, water quality can be naturally different in both spatial and temporal scales to reflect the terrestrial influences (W. Zhu, Y. Q. Tian, Q. Yu, & B. L. Becker, 2013a). Abnormal high nutrient concentrations can occur in certain circumstances, such as hurricanes that can cause sediment resuspension and accelerate the release of nutrients to the dissolved phase (Kalnejais et al., 2010). A study conducted in Sweden shows the impact of sediment resuspension on changes of the nutrient flux rates with a decrease in phosphate, an increase in nitrate and nitrite, and no significant change in ammonia (Tengberg et al., 2003). In the Mediterranean and the Black Sea area, a study conducted by a group of researchers (Ludwig et al., 2009) reported that freshwater discharge in Mediterranean rivers was reduced at least by 20% between 1960 and 2000, N and P fluxes in Mediterranean rivers were strongly influenced by human activities, and riverine nutrient discharges were the major sources causing nutrient pollution and worsening the local ecosystem.

High nutrient concentrations can not only damage the aquatic ecosystem but also endanger human health. This is especially of concern for the main rivers supplying drinking water where nutrient concentrations should be controlled in a safe range (Chaudhary et al., 2017). Therefore, an effort to reduce nutrient input into the rivers, estuaries and coastal waters has to be enforced globally and regionally. In the United States, the U.S. government has established a number of water monitoring stations to monitor and ensure the water quality. Since 1987, the United States Environmental Protection Agency(USEPA) has developed total maximum daily loading (TMDL) to regulate non-point sources of nutrient input (Group, 2008). Subsequently, many TMDL models are developed by the government for different local areas. Some regional efforts have also been made, including New York/New Jersey Harbor Estuary (NY/NJ Harbor), Delaware Estuary, Nearshore Ocean, and Shallow Coastal Bays (including Barnegat Bay) (Mauriello, 2009). The New York-New Jersey Harbor Estuary Program (HEP) is one of the projects established to study the nutrient TMDLs throughout the NY/NJ harbor. The program established the detailed criterion for nutrient discharge from different treatment plants and treatment processes to reduce nitrogen and phosphorus loadings. The program also categorized the total nitrogen (TN) removal levels into low (TN 10-12 mg L^{-1}), medium (TN 6-10 mg L^{-1}) and high (TN 4-5 mg L^{-1}) levels according to different treatment processes(Group, 2008). The

US New Jersey Department of Environmental Protection published a new nutrient requirement in 2009 for high water-quality needs in our life. For instance, total phosphorus (TP) concentration in the effluent has been changed from 1.0 mg L^{-1} to 0.1 mg L^{-1} (Mauriello, 2009).

A number of studies have shown that increased nutrient (N and P) concentrations and fluxes are strongly impacted by anthropogenic activities. Therefore, integrated management methods are in high demand for identification and control of the possible pollution sources that impact on the environment (Gaspar et al., 2017). Some studies use the ratio between total dissolved inorganic nitrogen and phosphate as a measure to evaluate nutrient eutrophication conditions in the waterbody (Huang et al., 2003). In this paper, water quality data and modeling techniques from published literature were reviewed and analyzed to evaluate nutrient (N and P) pollution in aquatic systems. In future, environmental management and ecosystem restoration should be a focal area in this regard.

2. Point and non-point sources of nutrients

Intensive urban development has caused a serious issue in habitat health as indicated by water quality. During the last few decades, anthropogenic activities contributed major nutrient contamination to the waterbody. The main contaminant sources include fertilizers, animal waste, human sewage, household products, and byproducts from petroleum production and agricultural fields. Other sources include industrial manufacturing and lawn use (Antweiler et al., 1996). Input of nutrient contaminants are categorized into point and non-point sources (Table 2-1). Land use, land cover change, and combined sewage overflow are considered as significant non-point and point sources, causing nutrient pollution. Impervious surface coverage is a quantifiable land-use change indicator. The causes and corresponding treatment methods of nutrient pollution are summarized in Table 2-1. Strategies for landscape design should be made by administration to address the environmental problems in a community (Wickham et al., 2013).

3. Non-point sources due to land use and land cover changes

Because different land use types can determine soil type, land use and land cover changes can cause changes in land geology and geomorphology. This can affect biological community, soil and sediment stability, and water runoff rate. Therefore, these are important factors affecting aquatic systems. Nutrient (N and P) concentrations in surface water are considered to be mainly controlled by water-rock interactions (i.e. weathering). A study in the Asian monsoon region shows that sediment processes have potential impacts on water quality because inorganic nutrients are mostly from storm runoff during a monsoon and can be transported to a relatively long distance(Kim et al., 2016). As exemplified by a deciduous forest stream, inorganic N and P sink in upper soil horizons while the parent dolomite weathering is the major source of inorganic P into the stream. In a riparian zone, when dissolved oxygen (DO) is high, inorganic P sinks. When DO is low, however, the riparian zone is a potential source of NH_4^+ and PO_4^{3-}. In most cases, nutrients (N and P) are good indicators of land use change impact because they are used to evaluate relationships between land use change and nutrients loading change. Specifically, land use change coupling with climate change can accelerate soil erosion and result in an increase in nutrient loading and discharge in the wet season, and a decrease in the dry season (Trang, Shrestha, Shrestha, Datta, & Kawasaki, 2017).

When studying N and P concentrations in stream water, terrestrial and instream processes are important. It has been reported that land use change can affect freshwater discharge and nutrient flux (J. Downing et al., 1999), alter nutrient biogeochemical cycle, and introduce high nutrient concentrations into the water body (S. Li et al., 2009). Thus, it can have a significant environmental impact on the local ecosystems and a potential to change biogeochemistry of the aquatic system. These include the impact on microorganisms in aquatic systems and populations of communities in an ecosystem. However, ecosystem functions, such as regulation of water flows, soil retention, habitat, and biodiversity maintenance, can better support the ecosystems and protect the environment (Melton et al., 2016). In granite and silicate terrain landscape with low precipitation and high transpiration biomes, the uptake of N and P through vegetation has more significant influence than water-rock interaction in controlling nutrient concentrations (Dean, Webb, Jacobsen, Chisari, & Dresel, 2014). Previous studies also indicate that forest vegetation can control sediment loads and sufficiently ensure water quality in the aquatic system, which can then ensure the conservation of the species in aquatic ecosystems (N. M. Anderson, Germain, & Hall, 2012). In a forest ecosystem, organic matter is a major carrier of N and P. Spatial distribution and loss of N and P depends on organic matter content and its interactions with soils. Soil content is important because storm water runoff can wash out the available nutrients into streams and rivers, resulting in a high level of nutrient concentrations. Most of dissolved organic nitrogen (DON) and dissolved organic phosphate (DOP) have functional groups associated with humid, hydrophilic acid and hydrophilic neutral fractions which have little impact on the behavior of most of dissolved organic matter (DOM). The carboxylic and phenolic functional groups of DOM are very important in governing the behavior of nitrogen (Qualls & Haines, 1991).

Although physical soil and water conservation practice can reduce storm water runoff, soil erosion, and nutrient depletion, it also decreases the crop yield due to the loss of cultivable area. However, if physical soil can be changed to an agronomic soil practice, then the crop yield can be increased with a reduction in runoff and soil erosion (Adimassu, Langan, Johnston, Mekuria, & Amede, 2017). Plant uptake of nutrients is also an important mechanism to deplete nutrients in surface water. Immobilization of inorganic N and P is found to be taken up by microbes on decomposing leaves and algae (Mulholland, 1992). Agriculture may positively improve or negatively affect the water quality based on specific situations. For example, crop planting can help keep nutritional materials in soil and roots, but overdoses of insecticide and nutrients (e.g., phosphate and nitrogen) can result in eutrophication in the waterbody. Knowing the processes of how agricultural land use change affects the aquatic ecosystems will help to protect the water quality and implement sustainable water management (B. Mehdi et al., 2015). Anthropogenic nutrients are mainly from agricultural fertilizer use. In order to quantify agricultural impacts on water quality, the conservation intensity is used to represent the implementation impacts of conservation practices that indicate the agricultural land use impacts on water quality. Sufficient evidence supports that conservation practice in the Upper Mississippi River Basin has a detectable larger impact on nitrogen loading than phosphate loading (Garcia et al., 2016). Another study in aquatic ecosystems indicates that the changes in land use pattern can result in changes in biological community structure and cause the diversity of the community to decline (Cooper, 1995). Overall, reduction of non-point agricultural source pollution is essential to improve the water quality in aquatic ecosystems.

4. Point sources associated with combined sewer overflow

With development of water treatment technology, waste treatment systems have been used to improve water quality by decreasing the nutrient discharge into aquatic systems (D. W. Crawford, Bonnevie, & Wenning, 1995). However, combined sewer overflows (CSOs) and industrial waste discharge are still the major sources of nutrient pollution. The combined sewer overflows are used to assemble water from point and non-point pollution sources together and then discharge nutrients as a point source into rivers, streams, estuaries and coastal waters. Studies in the late 1990s showed that major mass loading of nutrient pollution was from publicly owned treatment works and combined sewer overflows (D. W. Crawford et al., 1995). According to laboratory and field analyses, water samples from combined sewer overflows exhibit higher nutrient concentrations (N, 24±10 mg. L^{-1}; and P, 1.8± 0.5 mg. L^{-1}) than from publicly owned treatment works (Reemtsma, Gnirß, & Jekel, 2000). Thus, water discharged from CSOs causes relatively high nutrient concentrations (Reemtsma et al., 2000), which makes combined sewer overflows an important point source of nutrient pollution to aquatic systems. In other words, discharge of untreated nutrients and other chemicals from combined sewer overflows can place high risks on aquatic environment and human health. In order to evaluate the combined sewer overflows in a less expensive way, subjective assessment criteria are proposed by some studies (Morgan, Xiao, & McNabola, 2017). Knowing the dynamics and toxicity of nutrients discharged from combined sewer overflows can enhance the management of CSO accidents. Based on the evaluation and characterization of sediment and downstream water quality and flow dynamics information, recommendations can be made to optimize management methods (Becouze-Lareure et al., 2016).

Since the combined sewer overflows can have a significant impact on water quality, evaluation of combined sewer overflows is of great importance to ensure a better-quality ecosystem. In order to improve the water quality, the government at different levels has made a concerted effort to enact the new regulations, manage the combined sewer overflows events, and evaluate the cost of nutrient reduction in each area (Protection, 2000), which has challenged the treatment process of facilities located upstream of lakes, ponds, or reservoirs. To address storm flooding and associated combined sewer overflows, both the New York Department of Environmental Conservation and the New Jersey Department of Environmental Protection in the United States have proposed several solutions such as conducting a green infrastructure plan to control storm water runoff, upgrading the control of combined sewer overflow outfalls, and reducing the overall amount of sewage flow (Amar et al., 2014). Management of combined sewer overflows is conducted by several methods including but not limited to model evaluation, wetland construction, multiple management methods. In the meantime, it is necessary to construct a large database to evaluate the impacts due to combined sewer overflows. In the United States, New York City is seeking a citizen science-based water quality monitoring program coupled with efficiency and cost analysis, which is focused on establishing a more efficient, time and cost-saving system to monitor combined sewer overflow impacts (Farnham et al., 2017). So far, diverse methods have been developed to treat combined sewer overflows. One of the methods is to construct wetlands. A case study in Italy demonstrated the monitoring of combined sewer overflows' quality and quantity at different sites (Masi, Rizzo, Bresciani, & Conte, 2017). The results show that wetland treatment can reduce nitrogen concentration by 93%, which implies a significant success (Masi et al., 2017).

5. Modeling approach in nutrient study

In order to better estimate the relationship between land use change and nutrient concentration, modeling approaches have been applied to estimate the total nitrogen and phosphate loading from different sources (Johnes, 1996). Various modeling approaches have been developed to evaluate the sources of nutrient pollution and further the fate and mass transfer of nutrients (Brabec et al., 2002; Ji, 2017). This is now a commonly used method to predict the impacts of land use changes on water quality over decades. The results can help management teams to evaluate water pollution and make strategies to control nutrient input (Johnes, 1996). For example, a research group in Kenya used the modeling approach to find the relationships between land use and nutrient cycling in applicable areas (Jacobs, Weeser, Breuer, Butterbach-Bahl, & Rufino, 2016). Their results indicate that different types of land use can impact the nitrate concentrations in streams with seasonal alteration between wet and dry seasons (Jacobs et al., 2016).

Previous studies on hydrodynamic models have shown their importance in evaluating the sources of nutrients. For example, the Everglades Wetland Hydrodynamic Model (EWHM) was originally designated to be used in wetlands (Moustafa & Hamrick, 2000). Then it was turned into a nutrient's removal model with the proper calibration. The model prediction results showed a significant correlation to the observed data (Moustafa & Hamrick, 2000). Another three-dimensional hydrodynamic model based on a 4-year data calibration was developed to estimate the amount of net nutrient inflow from the Baltic proper (Helminen, Juntura, Koponen, Laihonen, & Ylinen, 1998). The dynamic balance of mass loading calculation was used in the model and indicated the importance of background nutrients loading from Balti proper (Helminen et al., 1998). Hydrodynamic models have been developed over time from one-dimensional to three-dimensional models which are commonly used to evaluate nutrient transportation in rivers and estuaries (Testa et al., 2014). In a sense, hydrodynamic modeling is a combination of computer simulations with a consideration of physical and biological processes in surface water systems. Nutrient cycle, water flow, oxygen demand and other chemical and biological indices can be the components in the one-dimensional models. With the information compiled from different nutrient concentrations, organic matter content, and biological components, the models can be adjusted to any kind of lakes and reservoirs (Moriarty et al., 2017). In order to further evaluate the water quality in different aquatic environments, integrated models are also used in the nutrient study. Table 2-2 shows some examples of different models for nutrient study. For example, a model used by the Chicago Area Waterway System (CAWS) in the United States can capture the fate and transport of combined sewer overflow discharges (Quijano, Zhu, Morales, Landry, & Garcia, 2017). The hydrodynamic model simulates the transportation of combined sewer overflows. The results indicate that due to large water dilution impact there is no significant combined sewer overflows impact in water quality within the system boundaries (Quijano et al., 2017).

Because water quality is a major global issue today, the water quality model development has been attracting significant attention. Since a one-dimensional model has its drawbacks in determination of the hydrodynamic and ecological response, a three-dimensional model was introduced into lacustrine ecosystems. For example, ELMO (an Ecological Model) is a three-dimensional water quality model for nutrient study, which can show a quick ecosystem response to hydrodynamic influences (Bonnet & Wessen, 2001). Chemical parameters are often used in the water quality model. In a study assessing phosphorus control in the James River Estuary in Virginia, USA, parameters that can reflect the water quality, such as carbonaceous biochemical oxygen demand (CBOD), dissolved oxygen, nitrate, nitrite and other chemical parameters, are used for model simulation of chemical reaction kinetic processes (Lung, 1986). The modeling results suggest that the wastewater treatment plant can reduce a massive phosphorus loading and control phytoplankton biomass to a reasonable level (Lung, 1986). The same modeling process is also used for nitrogen estimation (Lung & Testerman, 1989). The results show that phosphorus control in the upper estuary can provide the lower estuary with more nitrogen, but the additional nitrogen has no significant impact on algae growth (Lung & Testerman, 1989). Other than just using a simple hydrodynamic model, an integrated hydrodynamic model with water quality components is more practically useful. For example, a combined physical-biological model was used as a tool in a study to estimate the impact of the nutrient cycling on zebra mussels in a lake system (León et al., 2005). Algal blooming is also an important indicator of eutrophication of water body. In a nutrient study conducted in the Daoxiang Lake, Beijing, China, a biological model named EFDC (Environmental Fluid Dynamics Code) was used to predict algal blooming (Wu & Xu, 2011). The results from this model showed that the simulation matched the observed results reasonably well with an accuracy of 63.4% for algal bloom prediction (Wu & Xu, 2011). To ensure the efficiency of a wetland construction, various models have been developed to simulate the performance of the treatment. For example, a biokinetic model evaluates transformation and degradation processes of combined sewer overflows with or without constructed wetlands with pollution loading and transportation estimated (Pálfy et al., 2016). Ammonia nitrogen and COD are a good fit for this model (Pálfy et al., 2016).

Usually, a model for nutrient studies has to be modified before it can be applied to different local areas, in order to ensure its applicability and the accuracy of the application (Wu & Xu, 2011). For example, due to limited biological data in a two-year simulation study, a three-dimensional hydrodynamic model (ELCOM) coupling with a one-dimensional aquatic ecosystem dynamic model (CAEDYM) was used to improve the accuracy of biogeochemical simulation in two different reservoirs (Romero, Antenucci, & Imberger, 2004; Weigel et al., 2017). In a study on the North West European Shelf, a three-dimensional ecosystem model was applied to estimate nutrient fluxes and budgets based on a seasonal cycle (Proctor, Holt, Allen, & Blackford, 2003). The US Environmental Protection Agency developed a three-dimensional hydrodynamic-eutrophication model (HEM-3D), which was tested in Korea, as a tool to estimate total maximum daily load (TMDL) (Park, Jung, Kim, & Ahn, 2005). The results showed that organic wastes degraded the water quality along Korea coastal areas especially in Kwang-Yang Bay (Park et al., 2005). In construction of a biological model, it is usually difficult to quantify the bio-transformations of nutrients. Because of the complexity in different water zones, ecological models have to be applied differently for each purpose. In order to overcome this problem, a two-dimensional hydrodynamic model coupling with the biogeochemical MIRO model was developed to quantify the biogeochemical transformations and fluxes of nutrients in coastal zones (Arndt, Lacroix, Gypens, Regnier, & Lancelot, 2011). In the meantime, another biogeochemical model (CONTRASTE), which combined with a hydrodynamic model, was used to evaluate nutrient concentrations in estuarine water (Arndt et al., 2011). The results indicate that both nutrient input and physical constraints are important factors that control phytoplankton blooms in coastal zone. The interface between estuary and coastal zone plays a central role in the continuum of water body (Arndt et al., 2011). Since marine and coastal systems are so complicated, hydrodynamic-ecosystem models should have error quantification by using analytical methods to ensure the model performance and prediction accuracy, which include correlations, model bias, and efficiency (Allen, Holt, Blackford, & Proctor, 2007).

Overall the integrated one-dimensional, two-dimensional and three-dimensional models, which include hydrodynamic model, biological model and water quality parameters, can be used to identify the major factors controlling the nutrients loading from rivers and streams into estuaries and coastal waters and estimate the nutrient budgets in the aquatic systems. Besides combining hydrodynamic and biological models together with water quality models to evaluate combined sewer overflows impact, an approach with a geographical information system (GIS) model as a supplementary method is also cost effective in evaluation of both chemical and ecological factors (Morgan et al., 2017).

6. Conclusion

In summary, combined sewer overflows (CSOs) are still the major point sources of nutrient pollution in rivers, estuaries and coastal waters. However, wetland construction and CSO management methods are effective methods to reduce the CSO impacts on the aquatic environment. On the other hand, land use and land cover changes are the major non-point sources of nutrient pollution. Although the modeling methods can correlate landscape use change to water quality, non-point source pollution such as storm water runoff from agricultural and urban areas is still difficult to quantify and identify. It is relatively feasible to identify and control the point source pollution when the information of discharge locations is given. As nutrients are important indicators of water quality that are vital to human life and aquatic ecosystems, multiple management strategies should be enforced to improve water quality. From economic aspects, in the meantime, a cost benefit analysis should also be conducted in the future. It was reported that a mixed-integer management method could achieve more than 13% in cost savings (Zhao, Poe, & Boisvert, 2015). In the future, an integrated hydrodynamic, chemical and biological model should be further developed to assist in identifying the transport and fate of nutrients from both point and non-point sources.

Acknowledgments

This work was supported in part by Montclair State University's Graduate Assistantship (JN), Montclair State University's Faculty Scholarship Program (HF), and the State Key Laboratory of Estuarine and Coastal Research Open Research Fund (SKLEC- KF201607). The authors would like to thank Dr. Pengfei Zhang, Section Editor of Current Pollution Reports, Ms.

Lauren Greaves, Associate Editor of Current Pollution Reports, and two anonymous reviewers whose comments and suggestions have improved the quality of an early version of this manuscript.

Additional Information

Competing financial interests: The authors declare no competing financial interests.

Reference

Adimassu, Z., Langan, S., Johnston, R., Mekuria, W., & Amede, T. (2017). Impacts of Soil and Water Conservation Practices on Crop Yield, Run-off, Soil Loss and Nutrient Loss in Ethiopia: Review and Synthesis. *Environmental Management, 59*(1), 87-101.

Allen, J. I., Holt, J. T., Blackford, J., & Proctor, R. (2007). Error quantification of a high-resolution coupled hydrodynamic-ecosystem coastal-ocean model: Part 2. Chlorophyll-a, nutrients and SPM. *Journal of marine systems, 68*(3), 381-404.

Amar, M., Bauter, N., Bonomo, J., Burchell, A., Dua, K., Granton, C., . . . Prioleau, D. (2014). Bringing the City of Newark's Stormwater Management System into the 21st Century.

Anderson, N. M., Germain, R. H., & Hall, M. H. (2012). An Assessment of Forest Cover and Impervious Surface Area on Family Forests in the New York City Watershed. *Northern Journal of Applied Forestry, 29*(2), 67-73. doi:10.5849/njaf.11-009

Antweiler, R. C., Goolsby, D. A., & Taylor, H. E. (1996). Nutrients in the Mississippi river. *US GEOLOGICAL SURVEY CIRCULAR USGS CIRC*, 73-86.

Appelo, C. A. J., & Postma, D. (2004). *Geochemistry, groundwater and pollution*: CRC press.

Arndt, S., Lacroix, G., Gypens, N., Regnier, P., & Lancelot, C. (2011). Nutrient dynamics and phytoplankton development along an estuary–coastal zone continuum: a model study. *Journal of marine systems, 84*(3), 49-66.

Becouze-Lareure, C., Thiebaud, L., Bazin, C., Namour, P., Breil, P., & Perrodin, Y. (2016). Dynamics of toxicity within different compartments of a peri-urban river subject to combined sewer overflow discharges. *Science of the total environment, 539*, 503-514.

Bonnet, M., & Wessen, K. (2001). ELMO, a 3-D water quality model for nutrients and chlorophyll: first application on a lacustrine ecosystem. *Ecological Modelling, 141*(1), 19-33.

Brabec, E., Schulte, S., & Richards, P. L. (2002). Impervious surfaces and water quality: a review of current literature and its implications for watershed planning. *Journal of planning literature, 16*(4), 499-514.

Chaudhary, M., Mishra, S., & Kumar, A. (2017). Estimation of water pollution and probability of health risk due to imbalanced nutrients in River Ganga, India. *International Journal of River Basin Management, 15*(1), 53-60.

Chesworth, W. (2008). Encyclopedia of soil science.

Cooper, S. R. (1995). Chesapeake Bay watershed historical land use: impact on water quality and diatom communities. *Ecological applications*, 703-723.

Crawford, D., Bonnevie, N., Gillis, C., & Wenning, R. (1994). Historical changes in the ecological health of the Newark Bay Estuary, New Jersey. *Ecotoxicology and Environmental Safety, 29*(3), 276-303.

Crawford, D. W., Bonnevie, N. L., & Wenning, R. J. (1995). Sources of pollution and sediment contamination in Newark Bay, New Jersey. *Ecotoxicology and Environmental Safety, 30*(1), 85-100.

Cerco, C. F., & Cole, T. (1993). Three-dimensional eutrophication model of Chesapeake Bay. *Journal of Environmental Engineering, 119*(6), 1006-1025.

Dean, J., Webb, J., Jacobsen, G., Chisari, R., & Dresel, P. (2014). Biomass uptake and fire as controls on groundwater solute evolution on a southeast Australian granite: aboriginal land management hypothesis. *Biogeosciences, 11*(15), 4099-4114.

Downing, J., McClain, M., Twilley, R., Melack, J., Elser, J., Rabalais, N., . . . Soto, D. (1999). The impact of accelerating land-use change on the N-cycle of tropical aquatic ecosystems: current conditions and projected changes. *Biogeochemistry, 46*(1-3), 109-148.

Eberl, H., Picioreanu, C., Heijnen, J., & Van Loosdrecht, M. (2000). A three-dimensional numerical study on the correlation of spatial structure, hydrodynamic conditions, and mass transfer and conversion in biofilms. *Chemical Engineering Science, 55*(24), 6209-6222.

Farnham, D. J., Gibson, R. A., Hsueh, D. Y., McGillis, W. R., Culligan, P. J., Zain, N., & Buchanan, R. (2017). Citizen science-based water quality monitoring: Constructing a large database to characterize the impacts of combined sewer overflow in New York City. *Science of the total environment, 580,* 168-177.

Fillos, J., & Swanson, W. R. (1975). The release rate of nutrients from river and lake sediments. *Journal (Water Pollution Control Federation),* 1032-1042.

Friedman, C. L., & Lohmann, R. (2014). Comparing sediment equilibrium partitioning and passive sampling techniques to estimate benthic biota PCDD/F concentrations in Newark Bay, New Jersey (USA). *Environmental Pollution, 186,* 172-179.

Garcia, A. M., Alexander, R. B., Arnold, J. G., Norfleet, L., White, M. J., Robertson, D. M., & Schwarz, G. (2016). Regional effects of agricultural conservation practices on nutrient transport in the Upper Mississippi River Basin. *Environmental science & technology, 50*(13), 6991-7000.

Gaspar, R., Marques, L., Pinto, L., Baeta, A., Pereira, L., Martins, I., . . . Neto, J. M. (2017). Origin here, impact there—The need of integrated management for river basins and coastal areas. *Ecological Indicators, 72,* 794-802.

Group, N. J. H. D. (2008). *Nutrients Reduction Cost Estimation Study Summary Report*.

Hamilton, D. P., & Schladow, S. G. (1997). Prediction of water quality in lakes and reservoirs. Part I—Model description. *Ecological Modelling, 96*(1-3), 91-110.

Helminen, H., Juntura, E., Koponen, J., Laihonen, P., & Ylinen, H. (1998). Assessing of long-distance background nutrient loading to the Archipelago Sea, northern Baltic, with a hydrodynamic model. *Environmental Modelling & Software, 13*(5), 511-518.

Huang, X., Huang, L., & Yue, W. (2003). The characteristics of nutrients and eutrophication in the Pearl River estuary, South China. *Marine Pollution Bulletin, 47*(1), 30-36.

Jacobs, S., Weeser, B., Breuer, L., Butterbach-Bahl, K., & Rufino, M. (2016). *Identifying the impacts of land use on water and nutrient cycling in the South-West Mau, Kenya.* Paper presented at the EGU General Assembly Conference Abstracts.

Jin, Y., Wang, Y., Wang, W., Shang, Q., Cao, C., & Erwin, D. (2000). Pattern of marine mass extinction near the Permian-Triassic boundary in South China. *Science, 289*(5478), 432-436.

Ji, Z.-G. (2017). *Hydrodynamics and water quality: modeling rivers, lakes, and estuaries*: John Wiley & Sons.

Johnes, P. J. (1996). Evaluation and management of the impact of land use change on the nitrogen and phosphorus load delivered to surface waters: the export coefficient modelling approach. *Journal of hydrology, 183*(3-4), 323-349.

Kalnejais, L. H., Martin, W. R., & Bothner, M. H. (2010). The release of dissolved nutrients and metals from coastal sediments due to resuspension. *Marine Chemistry, 121*(1), 224-235.

Kim, K., Kim, B., Knorr, K. H., Eum, J., Choi, Y., Jung, S., & Peiffer, S. (2016). Potential effects of sediment processes on water quality of an artificial reservoir in the Asian monsoon region. *Inland Waters, 6*(3), 423-435.

Krauskopf, K. B. (1979). *Introduction to geochemistry*: McGraw-Hill.

Lathrop, R. G., Tulloch, D. L., & Hatfield, C. (2007). Consequences of land use change in the New York–New Jersey Highlands, USA: Landscape indicators of forest and watershed integrity. *Landscape and Urban Planning, 79*(2), 150-159. doi:10.1016/j.landurbplan.2006.02.008

León, L. F., Imberger, J., Smith, R. E., Hecky, R. E., Lam, D. C., & Schertzer, W. M. (2005). Modeling as a tool for nutrient management in Lake Erie: a hydrodynamics study. *Journal of Great Lakes Research, 31*, 309-318.

Li, S., Liu, W., Gu, S., Cheng, X., Xu, Z., & Zhang, Q. (2009). Spatio-temporal dynamics of nutrients in the upper Han River basin, China. *Journal of hazardous materials, 162*(2), 1340-1346.

Ludwig, W., Dumont, E., Meybeck, M., & Heussner, S. (2009). River discharges of water and nutrients to the Mediterranean and Black Sea: major drivers for ecosystem changes during past and future decades? *Progress in Oceanography, 80*(3), 199-217.

Lung, W.-S. (1986). Assessing phosphorus control in the James River Basin. *Journal of Environmental Engineering, 112*(1), 44-60.

Lung, W.-S., & Testerman, N. (1989). Modeling Fate and Transport of Nutrients on the James Estuary. *Journal of Environmental Engineering, 115*(5), 978-991.

Masi, F., Rizzo, A., Bresciani, R., & Conte, G. (2017). Constructed wetlands for combined sewer overflow treatment: Ecosystem services at Gorla Maggiore, Italy. *Ecological Engineering, 98*, 427-438.

Mauriello, M. N. (2009). New Jersey Nutrient Criteria Enhancement Plan.

Mehdi, B., Lehner, B., Gombault, C., Michaud, A., Beaudin, I., Sottile, M. F., & Blondlot, A. (2015). Simulated impacts of climate change and agricultural land use change on surface water quality with and without adaptation management strategies. *Agriculture, Ecosystems & Environment, 213*, 47-60. doi:10.1016/j.agee.2015.07.019

Melton, F., Xiong, J., Wang, W., Milesi, C., Li, S., Quackenbush, A., . . . Nemani, R. (2016). Potential Impacts of Climate and Land Use Change on Ecosystem Processes in the Great Northern and Appalachian Landscape Conservation Cooperatives *Climate Change in Wildlands* (pp. 119-150): Springer.

Miao, S., DeLaune, R., & Jugsujinda, A. (2006). Influence of sediment redox conditions on release/solubility of metals and nutrients in a Louisiana Mississippi River deltaic plain freshwater lake. *Science of the total environment, 371*(1), 334-343.

Morgan, D., Xiao, L., & McNabola, A. (2017). Evaluation of combined sewer overflow assessment methods: case study of Cork City, Ireland. *Water and Environment Journal*.

Moriarty, J., Harris, C. K., Fennel, K., Xu, K., Rabouille, C., & Friedrichs, M. A. (2017). A Model Archive for a Coupled Hydrodynamic-Sediment Transport-Biogeochemistry Model for the Rhône River Sub-aqueous Delta, France.

Moustafa, M., & Hamrick, J. (2000). Calibration of the wetland hydrodynamic model to the Everglades Nutrient Removal Project. *Water Quality and Ecosystems Modeling, 1*(1), 141-167.

Mulholland, P. J. (1992). Regulation of nutrient concentrations in a temperate forest stream: roles of upland, riparian, and instream processes. *Limnology and Oceanography, 37*(7), 1512-1526.

Ofiara, D. D. (2015). The New York Bight 25years later: Use impairments and policy challenges. *Marine Pollution Bulletin, 90*(1), 281-298.

Pálfy, T., Molle, P., Langergraber, G., Troesch, S., Gourdon, R., & Meyer, D. (2016). Simulation of constructed wetlands treating combined sewer overflow using HYDRUS/CW2D. *Ecological Engineering, 87*, 340-347.

Parette, R., & Pearson, W. N. (2014). 2, 4, 6, 8-Tetrachlorodibenzothiophene in the Newark Bay Estuary: The likely source and reaction pathways. *Chemosphere, 111*, 157-163.

Park, K., Jung, H.-S., Kim, H.-S., & Ahn, S.-M. (2005). Three-dimensional hydrodynamic-eutrophication model (HEM-3D): application to Kwang-Yang Bay, Korea. *Marine environmental research, 60*(2), 171-193.

Proctor, R., Holt, J. T., Allen, J. I., & Blackford, J. (2003). Nutrient fluxes and budgets for the North West European Shelf from a three-dimensional model. *Science of the total environment, 314*, 769-785.

Protection, N. J. D. o. E. (2000). *CSO communites cost*. http://www.wnjpin.net/OneStopCareerCenter/LaborMarketInformation/lmi25/pub/index.html.

Qualls, R. G., & Haines, B. L. (1991). Geochemistry of Dissolved Organic Nutrients in Water Percolating through a Forest Ecosystem. *Soil Science Society of America Journal, 55*(4), 1112-1123. doi:10.2136/sssaj1991.03615995005500040036x

Quijano, J. C., Zhu, Z., Morales, V., Landry, B. J., & Garcia, M. H. (2017). Three-dimensional model to capture the fate and transport of combined sewer overflow discharges: A case study in the Chicago Area Waterway System. *Science of the total environment, 576*, 362-373.

Reemtsma, T., Gnirß, R., & Jekel, M. (2000). Infiltration of combined sewer overflow and tertiary municipal wastewater: an integrated laboratory and field study on nutrients and dissolved organics. *Water research, 34*(4), 1179-1186.

Romero, J., Antenucci, J., & Imberger, J. (2004). One-and three-dimensional biogeochemical simulations of two differing reservoirs. *Ecological Modelling, 174*(1), 143-160.

Saba, T., & Su, S. (2013). Tracking polychlorinated biphenyls (PCBs) congener patterns in Newark Bay surface sediment using principal component analysis (PCA) and positive matrix factorization (PMF). *Journal of hazardous materials, 260*, 634-643.

Shin, J. Y., Artigas, F., Hobble, C., & Lee, Y.-S. (2013). Assessment of anthropogenic influences on surface water quality in urban estuary, northern New Jersey: multivariate approach. *Environmental monitoring and assessment, 185*(3), 2777-2794.

Tengberg, A., Almroth, E., & Hall, P. (2003). Resuspension and its effects on organic carbon recycling and nutrient exchange in coastal sediments: in situ measurements using new experimental technology. *Journal of Experimental Marine Biology and Ecology, 285*, 119-142.

Testa, J. M., Li, Y., Lee, Y. J., Li, M., Brady, D. C., Di Toro, D. M., . . . Fitzpatrick, J. J. (2014). Quantifying the effects of nutrient loading on dissolved O 2 cycling and hypoxia in Chesapeake Bay using a coupled hydrodynamic–biogeochemical model. *Journal of marine systems, 139*, 139-158.

Trang, N. T. T., Shrestha, S., Shrestha, M., Datta, A., & Kawasaki, A. (2017). Evaluating the impacts of climate and land-use change on the hydrology and nutrient yield in a transboundary river basin: A case study in the 3S River Basin (Sekong, Sesan, and Srepok). *Science of the total environment, 576*, 586-598.

Vitousek, P. M., Aber, J. D., Howarth, R. W., Likens, G. E., Matson, P. A., Schindler, D. W., . . . Tilman, D. G. (1997). Human alteration of the global nitrogen cycle: sources and consequences. *Ecological applications, 7*(3), 737-750.

Weigel, D., Vilhena, L., Woods, P., Tonina, D., Tranmer, A., Benjankar, R., . . . Goodwin, P. (2017). Aquatic habitat response to climate-driven hydrologic regimes and water operations in a montane reservoir in the Pacific Northwest, USA. *Aquatic Sciences*, 1-14.

Wickham, J. D., Stehman, S. V., Gass, L., Dewitz, J., Fry, J. A., & Wade, T. G. (2013). Accuracy assessment of NLCD 2006 land cover and impervious surface. *Remote Sensing of Environment, 130*, 294-304.

Wu, G., & Xu, Z. (2011). Prediction of algal blooming using EFDC model: Case study in the Daoxiang Lake. *Ecological Modelling, 222*(6), 1245-1252.

Xu, F.-L., Dawson, R. W., Tao, S., Cao, J., & Li, B.-G. (2001). A method for lake ecosystem health assessment: an Ecological Modeling Method (EMM) and its application. *Hydrobiologia, 443*(1-3), 159-175.

Zou, Z.-H., Yi, Y., & Sun, J.-N. (2006). Entropy method for determination of weight of evaluating indicators in fuzzy synthetic evaluation for water quality assessment. *Journal of Environmental Sciences, 18*(5), 1020-1023.

Zhao, T., Poe, G. L., & Boisvert, R. N. (2015). *Management Areas and Fixed Costs in the Economics of Water Quality Trading*. Retrieved from

Zhu, W., Tian, Y. Q., Yu, Q., & Becker, B. L. (2013). Using Hyperion imagery to monitor the spatial and temporal distribution of colored dissolved organic matter in estuarine and coastal regions. *Remote Sensing of Environment*, *134*, 342-354.

Tables

Table 2-1 The causes and corresponding treatment methods of nutrient pollution.

Category of nutrient pollution	Causes of nutrient pollution	Treatment methods	References
Point source pollution	Combined sewage overflow (animal waste, human sewage and household products)	Conduct a green infrastructure plan to control storm runoff, upgrade the control of combined sewer overflow outfalls, and reduce the overall amount of sewage flow; Use combined hydrodynamic model to evaluate the CSOs impact; Establish a science-based water quality monitoring program; Construct wetlands	Reemtsma et al., 2000; Amar et al., 2014; Morgan et al., 2017; Farnham et al., 2017; Pálfy et al., 2016; Masi et al., 2017
	Publicly owned treatment work	Regulation and control of discharging loading	D. W. Crawford et al., 1995;
	Industrial manufacturing discharge	Regulation and control of discharging loading; Reevaluate the treatment process.	D. W. Crawford et al., 1995; Protection, 2000
Non-point source pollution	Land use and land cover change coupling with climate change, soil type and sediment processes	Change Physical soil to an agronomic soil practice	Adimassu et al., 2017; Trang et al., 2017
	Dissolved organic matter	Plant uptake, forest vegetation	Dean et al., 2014
	Agricultural fertilizer, lawn use	Conservation practice	Garcia et al., 2016; Mehdi et al., 2015

Table 2-2 Examples of selected one, two and three-dimensional models and their associated parameters, categories and functionalities in nutrient studies.

	Model type	Parameters	Suitable estimation area	References
Hydrodynamic and water quality model	1-D hydrodynamic model	Nutrient concentration, organic matter component, biological environment	Lakes and reservoirs	Hamilton & Schladow, 1997
	2-D hydrodynamic model	Nutrient concentration, organic matter component	Stream; Water quality	Tao, Li, Falconer, & Lin, 2001
	3-D hydrodynamic model; Everglades Wetland Hydrodynamic Model; ELMO	Bathymetry, rainfall, humidity, solar radiation, wind velocity inflow, and outflow, water surface elevation, horizontal velocities, and temperature.	Estuaries and coastal area; Tide flow	Jin, Hamrick, & Tisdale, 2000
Physical and biological model	3-D ELCOM	Biological data, rainfall, humidity, solar radiation, wind velocity inflow and so on	Lake; Nutrient cycle, fate and transport of nutrients	León et al., 2005
	3-D numerical model	Navier–Stokes equations and mass transfer with nonlinear reactions in the biofilm	Porous, heterogeneous system	Eberl, Picioreanu, Heijnen, & Van Loosdrecht, 2000
	CE-QUAL-ICM model	Multiple forms of algae, carbon, nitrogen, phosphorus, and silica; and dissolved oxygen	Time-variable, eutrophication process, nutrient runoff	Cerco & Cole, 1993
	EFDC model	Algae blooming	Swamp	Wu & Xu, 2011; Zou, Carter, Shoemaker, Parker, & Henry, 2006
	CAEDYM+ELCOM	Flow and adjective transport	Reservoir	Romero et al., 2004
	HEM-3D	Total maximum daily load	Bay	Park et al., 2005
	2-D+MIRO	Nutrients	Coastal	Arndt et al., 2011
	2-D+CONTRASTE	Nutrients	Estuary	Arndt et al., 2011

Chapter 3 Investigation of Regional Land Use Type and Its Potential Impact on Water Quality in Northern New Jersey, USA

Jing Nie[1], Sana Mirza[1], Michael Viteritto[1], Benjamin Witherell[2], Yang Deng[1], Huan Feng[1,*]

1 Department of Earth and Environmental Studies, Montclair State University, Montclair, New Jersey 07043, USA

Abstract

In this study, long-term (13 years) water quality monitoring and land-use data were used to examine the impact of land use change on the water quality in the Passaic River, New Jersey. Land cover of a watershed is considered a non-point source of nutrient pollution, while combined sewer overflow and dry cleaner sites are considered point sources of nutrient pollution. This study shows that the type of land use in the Passaic River watershed is still dominated by natural landscapes (i.e., wetland and forest area) and, to a less extent, urban landscapes (i.e., urban and barren areas). Agricultural land is not dominant in the study area. The results from this study indicate that urban land use can impact the water quality and local industrialization and urbanization are the main causes of urban environmental problems.

Keywords: nutrient pollution; nitrogen; phosphorus; land use land cover

1. Introduction

In a highly developed urban area, land use and land cover change are significant. Alterations to land use can result in differences in soil type and the amount impervious surface area. Regional differences in land use are an important factor that can affect nutrients loading. Surface water quality is related to both urban and agriculture land use change. Agriculture is the major nonpoint source in the riverine ecosystem among different land use pattern (Bateman et al., 2016). It has also been reported that regional changes to land use and land cover could affect temporal and spatial patterns of runoff and can eventually lead to further changes in nutrients loading. It has been reported that land use changes can have several effects on the region's environmental health. One study conducted in Sweden showed that changes in land use can have an affect the water quality, such as changing the nutrient concentration in the streams (Hallström, Röös, & Börjesson, 2014). Another Study reported that the land use change could even affect precipitation change and direct runoff change in a water basin(Geymen, 2016).Since New Jersey has diverse land use types, local differences in land use can result in different nutrients concentrations in watershed streams (Kling et al., 2014; Lucash, Scheller, Kretchun, Clark, & Hom, 2014).Variations in land use can reflect different economic statuses and population densities, as well as affect the nutrient loadings. For example, large grass lawns around a property will generally use more fertilizer which can be transported to local streams resulting in higher nutrients loading.

Combined Sewage Overflows (CSOs) are another factor that cause high nutrient pollution in rivers. CSOs place high risks on human health because they release untreated effluent directly into surface waters. In the past, geographical information systems (GIS) model have been used as a supplement to the criteria method to evaluate both chemical and ecological status (Morgan et al., 2017). Previous studies suggest that it is necessary to construct a large database to assess the impacts of CSOs. An example of a city implementing such plans was when New York City sought to establish a citizen science-based water quality monitoring system, the total effects of which were focused on establishing a more efficient, time saving, and cost effective system for monitoring CSO’s impacts (Farnham et al., 2017).

With a growing demand of human services, dry cleaner sites became an inevitable pollution source. Chlorinated solvents and chemicals in detergent established a high loading output to river and streams (Eklund, Simon, & Association, 2007). Residual chemicals in dry cleaner sites can cause a high health risk through the vapor intrusion, groundwater use and direct contact (Fowler & Dockter, 2010). In addition, these dry cleaner sites cause an extra input of phosphorus into CSOs and rivers(Schreiber et al., 1993). It is seen that urban development and land use change has had a significant impact on aquatic ecosystem (Cooper, 1995). Therefore, the impacts caused by land-use change should be studied. The results can help make strategies in landscape design to address the environment problem in a community(Arnold Jr & Gibbons, 1996). The current study focuses in detail on differences in land use types that can affect the nutrient concentrations. In the meantime, land-use change and CSOs were chosen as indicators of non-point and point pollution, respectively.

2. Methodology and materials

2.1 Study area

The study area chosen for this case study was the Passaic River and Saddle River watersheds (Figure 3-1). The land use type changes along the Passaic River. The upper Passaic River watershed is dominated by rural areas, while the lower Passaic River watershed is primarily urban areas. The New Jersey Harbor Discharge Group (NJHDG) set up twelve water quality monitoring sites along the Passaic River and its tributaries and has been collecting water quality data since December 2003 (Figure 3-1). Station 1 is in the upper Passaic River, which is a freshwater system. Station 6 is in the Saddle River, which is a tributary of the Passaic River. Station 11 is in the lower reach of the Passaic River. This area of the Passaic River is influenced by tides and seawater mixing (Figure 3-1).

2.2 Data collection and processing

The data sets used in this study were mainly retrieved from the New Jersey Department of Environmental Protection public database, which includes land use land cover data sets. This data was surveyed in 2004, 2006, 2008, 2011, 2013, and 2016. The database also includes water quality data, which was monitored during 2003-2013 by the New Jersey Harbor Discharge Group. Water quality data (2004 – 2013) from three (i.e., Stations 1, 6 and 11) out of twelve monitoring stations (i.e., Stations 1 – 12, Figure 3-1), were used for this study. Because land use and land cover data were considered as non-point source pollution sources for the purposes of this study, the 14-digit hydrologic Units (HUC 14) for each station point was used as the nutrient's (N&P) contribution watershed to the monitoring stations. Land use change data was collected from the New Jersey Department of Environmental Protection. Data is available for the years 2004, 2006, 2008, 2011, 2013, and 2016. In order to better match the data set, 14-digit hydrologic Units (HUC 14) sub-watersheds were chosen as a base assessment unit for the whole study including dry cleaner sites, land use land cover data set, and CSO sites.

2.3 Land use land cover

The study utilized LULC data from 2004, 2006, 2008, 2011, 2013, and 2016, which were obtained from the USGS National Land Cover Database (NLCD) product. The LULC data for each year was then clipped for the drainage basin of each NJHDG station. The NLCD provides nationwide data on land cover and land cover change, with a 30m resolution and a 16-class legend that is based on a modified Anderson Level II classification system (J. R. Anderson, 1976). For the purposes of this study, land use land cover data are also calculated for St1, St6 and St11 respectively.

3. Results and discussion

3.1 Overall land use land cover distribution in Passaic River

To better understand how the land use and land cover changes impact the nutrients in Passaic River, the HUC 14 watershed area was chosen as the primary unit for the study area. Figure 3-1 shows the detailed HUC 14 assessment units in Passaic River. Based on the information of land use and land cover in the HUCs, the overall land use type in the study area was calculated separately for 2004, 2006, 2008, 2011, 2013, and 2016 based on the survey data (Figure 3-2). The results show that Evergreen Forest (EForest) has only 0.2% of the total type and Developed, Medium Intensity (MI) has 25.7%-27.1% of the total type (Figure 3-2). From 2004 to 2016, MI increased 5% overall but EForesst decreased 29.4%. Among all the land use type, only MI, Developed, High Intensity (HI), and Woody Wetlands (WWetland)increased 5.3%, 4.8% and 22.5% respectively from 2004 to 2016. Open Water (Water), Devloped, Open Space (Open Space), Developed, Low Intensity (LowI), Barren Land (Barren), Mixed Forest (MForest) and Shrub decreased a little (lower than 10%) from 2004 to 2016. However, Decidous Forest (DForest), EForest, Grassland (Grass), Pasture, Cultivated Crops (Crop) and Emergent Herbaceous Wetlands (Ewetland) decreased 16.2%, 29.4%, 26.2%,51.8% ,57.3 and 46.5% respectively from 2004 to 2016. Most green area decreased a lot according to the data above.

Due to different land use types in each HUC 14 units, land use type for each water quality station (St1, St6 and St11) are also calculated from 2004 to 2016 (Figure 3-3). Figure 3-3 shows the average land use land cover type in each water quality monitoring station (St1, 6 and 11). The results show that for St 1 and St 11, the major land use type is MI. For St6, the major land use type is LowI. St6 has 29.6% of Open Space but St1 and St11 only has 15.9 and 8.5% respectively. Open Space are areas with a mixture of some constructed materials, but mostly vegetation in the form of lawn grasses which means St6 has large area with lawn grasses.

3.2 Nutrients concentration and distribution in Passaic River

Passaic River connects the inner stream to the Newark Bay where is a mixing zone of freshwater and ocean water. Figure 3-4 shows a trend of 10 years (2004-2013) averaged salinity gradient along the Passaic River from the upper river to the downriver. It is obvious that the lower Passaic River is influenced by seawater circulation due to tides.

As shown in Figure 3-5, the upper Passaic River is a freshwater system, but the lower Passaic River, like an estuarine system, is impacted by seawater mixing due to the tides. The shallow water depth and high river discharge at Station 1 causes the nutrient concentrations of N and P to be controlled by the freshwater discharge in the Passaic River (Figure 3-5a & b). A similar situation is observed at Station 6 in Saddle River, which is a tributary of Passaic River and belongs to a freshwater system. The N and P concentrations are also controlled by the river discharge in the Saddle River (Figure 3-5c & d). In a natural pristine environment such as Passaic River and Saddle River, nutrient concentrations exponential decrease with increasing river discharge, which suggests that the natural nutrients from the watershed are removed and diluted with increasing river flow (Figure 3-5). Station 11, located in the lower Passaic River, is under the influence of seawater intrusion as indicated by an increase in salinity (Figure 3-4). Due to urban development and contributions from CSO outlets, there is a less significant correlation between nutrient concentrations and river flow (Figure 3-5e & f). This correlation is more obviously demonstrated in the relationship between nitrogen concentration the river flow than that of the relationship between phosphorus concentration and river flow. The water quality monitoring data showed that phosphorus and total nitrogen concentrations, which in some instances were under low river flow condition, were observed to be high at Station 6 in Saddle River (Figure 3-5c &d). These high nutrient concentrations could be attributed to local point sources along the Saddle River, or from nonpoint sources in the watershed. There is also a possibility that the Saddle River watershed may have high nutrient inputs due to different types of vegetation, or these high concentrations could originate human activities. The precise reason for these heightened concentrations is beyond the scope of this research and will not be discussed here.

3.3 land use change and combined sewer overflows impacts

This study analyzed the land use type in the whole Passaic River watershed. Stations 1, 6 and 11 represent different sections on Passaic River and show different land use changes that may be reflected by different nitrogen and phosphorus concentrations at each station. Figure 3-3 shows the percentage of land use type around Stations 1, 6 and 11.

It is seen in Figure 3-3 that the percentage of DForest around Station 1 has been 8 times higher than that around Station 6 during this time period. The percentage of Shrub around Station 1 was 3 to 4 times higher than that around Station 6. This explains why Station 1 shows a natural pristine environment (Figure 3-5). In addition, station 6 has a high percentage of land that is Open Space, which ranges from 29.6% of the total land use around this station. This percentage of urban land could result in a relatively high N and P concentrations due to the high lawn grasses. The percentage of MI and HI around Station 11 is even higher than that around Station 6 and Station 11 receives N and P input from both upstream and neighboring areas.

It was reported that N/P ratio could be used to indicate potential nutrient sources (J. A. Downing, McCauley, & Oceanography, 1992). For reference, potential nutrient sources include phytoplankton (N:P = 7.2, (Hecky, Campbell, Hendzel, & Oceanography, 1993)), feedlot runoff (N:P = 6.4, (J. A. Downing et al., 1992)), urban stormwater drainage (N:P = 5.8, (Loehr, 1974)) and sewage (N:P = 2.8, (Foy & environment, 2005)).

In this study, it is found that the N/P mass ratio decreases from 6.75 at Station 1 to 5.67 at Station 11 in the Passaic River (Figure 3-6). In the pristine natural environment, the N/P mass ration should be close to 7.25 (Myklestad & Ecology, 1977). The downstream decrease of N/P ratio in the Passaic River suggests that land use change from more natural cover to urban use may introduce anthropogenic contaminants such as excess phosphorus to the lower Passaic River. The excess phosphorus could come from fertilizer use and CSO discharge that contains phosphorus from industrial and domestic uses. The land use in our study area is dominated by LowI, MI and Open Space with a few areas of mixed land cover (Figure 3-3). Agricultural land cover is not a dominant type of land use within the study area (Figure 3-3). Specifically, the results show that the areas around Stations 1 have a high percentage of natural forest lands (8.5%). The relatively high N and P concentrations at Station 6, as compared to that at Station 1, may be attributed to relatively lower discharge from Saddle River than that from the Passaic River. The areas around Station 11 are dominated by urban use, which may cause relatively low N/P ratio due to anthropogenic P input. The lower N and P concentrations measured at Station 11 could be caused by dilution and mixing with the seawater. Therefore, the land use type in a watershed can affect the water quality, as evidenced by the varying N and P concentrations across the Passaic River.

Besides non-point source pollution, which is associated with the land use type, point source pollution should also be considered when considering water quality (Figure 3-1). It was reported that there were active 27 combined sewage outfalls (CSOs) that discharge to the five navigation reaches, which cover approximately 8 miles, in the lower Passaic River (Inc, 2013). Specifically, the number and locations of these CSOs are The Point-No-Point Reach (1 CSO), Harrison Reach (4 CSOs), Newark Reach (10 CSOs), Kearny Reach (8 CSOs) and Arlington Reach (4 CSOs) (Figure 3-1). In addition, there are a number of stormwater outfalls (SWOs) distributed along the Lower Passaic River (Inc, 2013). Besides the CSOs, there are also a number of commercial dry cleaners distributed in our study area (Table 3-1). The effluents from these commercial dry cleaners should flow into the CSOs nearby and may also have a negative impact on the Passaic River water quality.

Combined sewer overflows (CSOs) assemble wastewater from known pollution sources and empty it into rivers, streams, or estuaries as point sources. Table 3 gives the number of CSOs and commercial dry cleaners in each HUC 14 unit in the study area. As shown in Figure 3-1, most CSOs are concentrated in the upper Passaic River near Stations 2 - 4 and the lower Passaic River near Station 8 - 11. It was found in this study that the N/P ratio at Station 1 was 6.82 (Figure 3-6), which was lower than the phytoplankton ratio (N/P = 7.2) (Schreiber et al., 1993). The result suggests that human activities may have affected the nutrients concentrations in this area. The area around Station 1 has 33 dry cleaner sites and 14 CSOs (Table 3-1), which can contribute a significant amount of phosphorus to the nearby CSOs, which would decrease N/P ratio in the area. In the Saddle River which is a tributary of Passaic River, the N/P ratio was found to be 6.38 at Station 6 (Figure 3-6), which was very close to the N/P ratio from feedlot runoff (N/P = 6.4) (J. A. Downing et al., 1992). Because there are no CSOs found in this area, feedlots in this area could be a significant anthropogenic nutrient source. In the lower Passaic River, the N/P ratio at Station 11 was found to be 5.7 (Figure 3-6), which is close to N/P ratio in urban stormwater (N/P = 5.8) (Loehr, 1974). It has been found that there are 190 dry cleaner sites contributing to 47 different CSOs around Station 11 (Inc., 2013). The results indicate that the highly urbanized area and CSOs around Station 11 have significant impacts on nutrient pollution in the area. Knowing the dynamics and toxicity of CSO discharges can enhance the management of CSOs events. In order to reduce nutrient pollution, diverse methods have to be developed to limit or reduce CSO impact. One of the methods is constructing wetlands. A case study in Italy examined effluent quality and quantity from CSOs were monitored at different sites (Cole, Lamarca, Connolly, & Anguelovski, 2017). The results showed that wetland treatment could reduce chemical oxygen demand (COD) and nitrogen concentration by 87% and 93% respectively (Cole et al., 2017). Therefore, wetland treatment showed a significant success(Masi et al., 2017).

4. Conclusion

This study shows that water quality in a riverine-estuarine system can be significantly impacted by the land use type. The water quality in the upper Passaic River watershed, which has a high amount of natural pristine forest landscape, is within the natural level. However, the water quality in the lower Passaic River watershed, which is dominated by urban land use, is significantly affected by anthropogenic sources. The results from this study suggest that both water quality monitoring and land use management are very important for the sustainable development of the Passaic River watershed.

5. Acknowledgement

This study is supported in part by Montclair State University's graduate assistantship program (JN, SM, MV). The authors would like to thank the New Jersey Department of Environmental Protection and Albert Gregory, the senior scientist in Passaic Valley Sewerage Commission.

Reference

Anderson, J. R. (1976). *A land use and land cover classification system for use with remote sensor data* (Vol. 964): US Government Printing Office.

Arnold Jr, C. L., & Gibbons, C. J. (1996). Impervious surface coverage: the emergence of a key environmental indicator. *Journal of the American planning Association, 62*(2), 243-258.

Bateman, I., Agarwala, M., Binner, A., Coombes, E., Day, B., Ferrini, S., . . . Posen, P. (2016). Spatially explicit integrated modeling and economic valuation of climate driven land use change and its indirect effects. *Journal of environmental management, 181*, 172-184.

Cole, H. V., Lamarca, M. G., Connolly, J. J., & Anguelovski, I. J. J. E. C. H. (2017). Are green cities healthy and equitable? Unpacking the relationship between health, green space and gentrification. *71*(11), 1118-1121.

Cooper, S. R. (1995). Chesapeake Bay watershed historical land use: impact on water quality and diatom communities. *Ecological applications*, 703-723.

Downing, J. A., McCauley, E. J. L., & Oceanography. (1992). The nitrogen: phosphorus relationship in lakes. *37*(5), 936-945.

Eklund, B. M., Simon, M. A. J. J. o. t. A., & Association, W. M. (2007). Concentration of tetrachloroethylene in indoor air at a former dry cleaner facility as a function of subsurface contamination: a case study. *57*(6), 753-760.

Farnham, D. J., Gibson, R. A., Hsueh, D. Y., McGillis, W. R., Culligan, P. J., Zain, N., & Buchanan, R. (2017). Citizen science-based water quality monitoring: Constructing a large database to characterize the impacts of combined sewer overflow in New York City. *Science of the total environment, 580*, 168-177.

Fowler, T., & Dockter, C. J. R. J. (2010). Optimizing remediation strategies for a former dry-cleaner site. *20*(4), 83-104.

Foy, R. H. J. P. a., & environment, t. (2005). The return of the phosphorus paradigm: agricultural phosphorus and eutrophication. (phosphorusagric), 911-939.

Geymen, A. (2016). DETERMINATION OF THE IMPACTS OF LAND USE CHANGE ON WATER BASIN USING REMOTE SENSING AND GEOGRAPHICAL INFORMATION SYSTEM. *ANADOLU UNIVERSITY JOURNAL OF SCIENCE AND TECHNOLOGYï A Applied Sciences and Engineering, 17*(3), 484-496.

Hallström, E., Röös, E., & Börjesson, P. (2014). Sustainable meat consumption: A quantitative analysis of nutritional intake, greenhouse gas emissions and land use from a Swedish perspective. *Food Policy, 47*, 81-90.

Hecky, R., Campbell, P., Hendzel, L. J. L., & Oceanography. (1993). The stoichiometry of carbon, nitrogen, and phosphorus in particulate matter of lakes and oceans. *38*(4), 709-724.

Inc, T. S. (2013). *Combined Sewer Overflow/Stormwater Outfall Investigation Quality Assurance Project Plan*. Retrieved from

Inc., T. S. (2013). *CSO/SWO Investigation Quality Assurance Project Plan*. Retrieved from

Kling, C. L., Panagopoulos, Y., Valcu, A. M., Gassman, P. W., Rabotyagov, S. S., Campbell, T., . . . Jha, M. K. (2014). Land Use Model Integrating Agriculture and the Environment (LUMINATE): Linkages between agricultural land use, local water quality and hypoxic concerns in the Gulf of Mexico Basin.

Loehr, R. C. J. J. (1974). Characteristics and comparative magnitude of non-point sources. 1849-1872.

Lucash, M. S., Scheller, R. M., Kretchun, A. M., Clark, K. L., & Hom, J. (2014). Impacts of fire and climate change on long-term nitrogen availability and forest productivity in the New Jersey Pine Barrens. *Canadian journal of forest research, 44*(5), 404-412.

Masi, F., Rizzo, A., Bresciani, R., & Conte, G. (2017). Constructed wetlands for combined sewer overflow treatment: Ecosystem services at Gorla Maggiore, Italy. *Ecological Engineering, 98*, 427-438.

Morgan, D., Xiao, L., & McNabola, A. (2017). Evaluation of combined sewer overflow assessment methods: case study of Cork City, Ireland. *Water and Environment Journal*.

Myklestad, S. J. J. o. E. M. B., & Ecology. (1977). Production of carbohydrates by marine planktonic diatoms. II. Influence of the NP ratio in the growth medium on the assimilation ratio, growth rate, and production of cellular and extracellular carbohydrates by Chaetoceros affinis var. willei (Gran) Hustedt and Skeletonema costatum (Grev.) Cleve. *29*(2), 161-179.

Schreiber, J. S., House, S., Prohonic, E., Smead, G., Hudson, C., Styk, M., & Lauber, J. J. R. A. (1993). An investigation of indoor air contamination in residences above dry cleaners. *13*(3), 335-344.

Figures

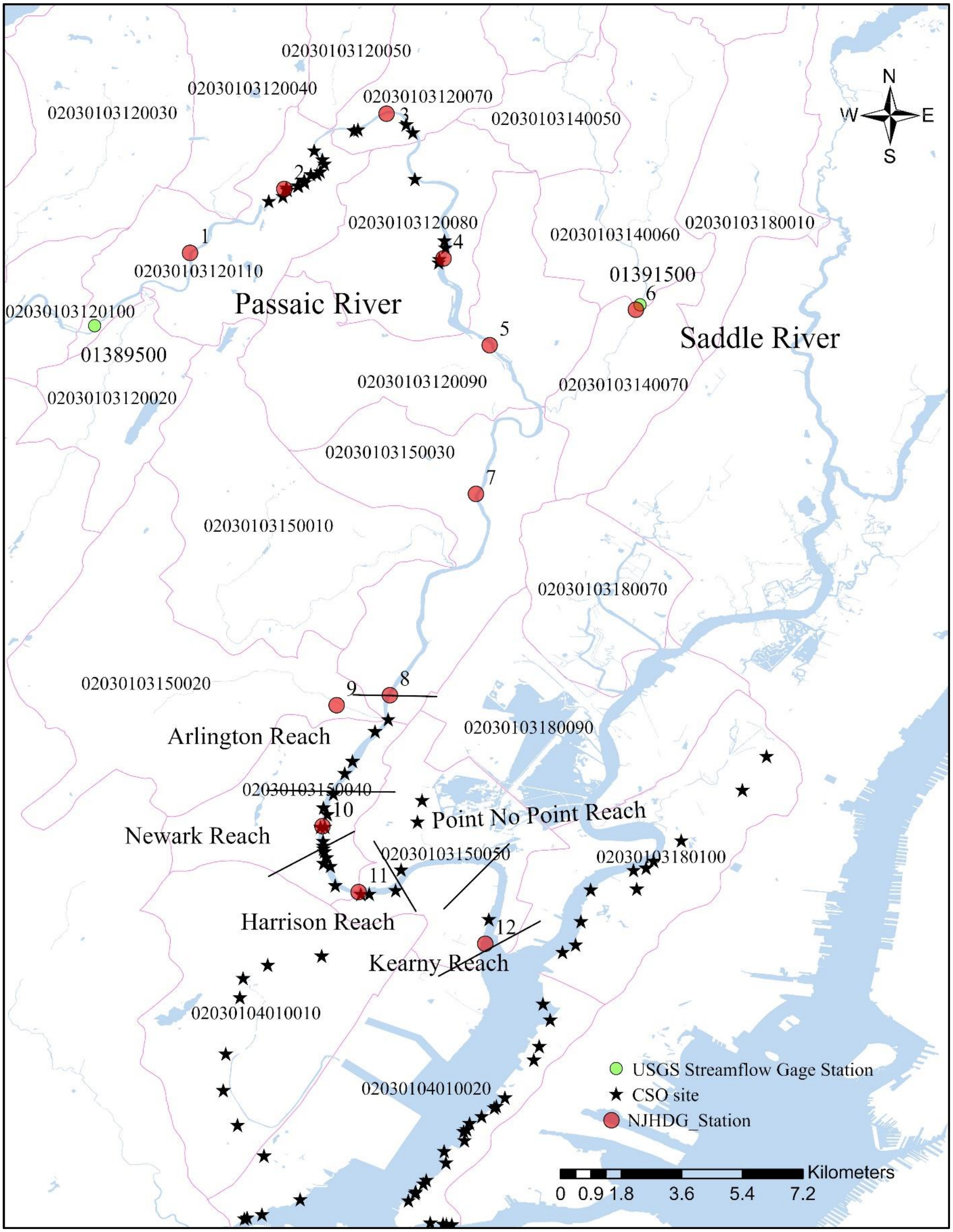


Figure 3-1 Study area map in Passaic River with NJHDG water quality monitoring stations and USGS flow sites and CSOs

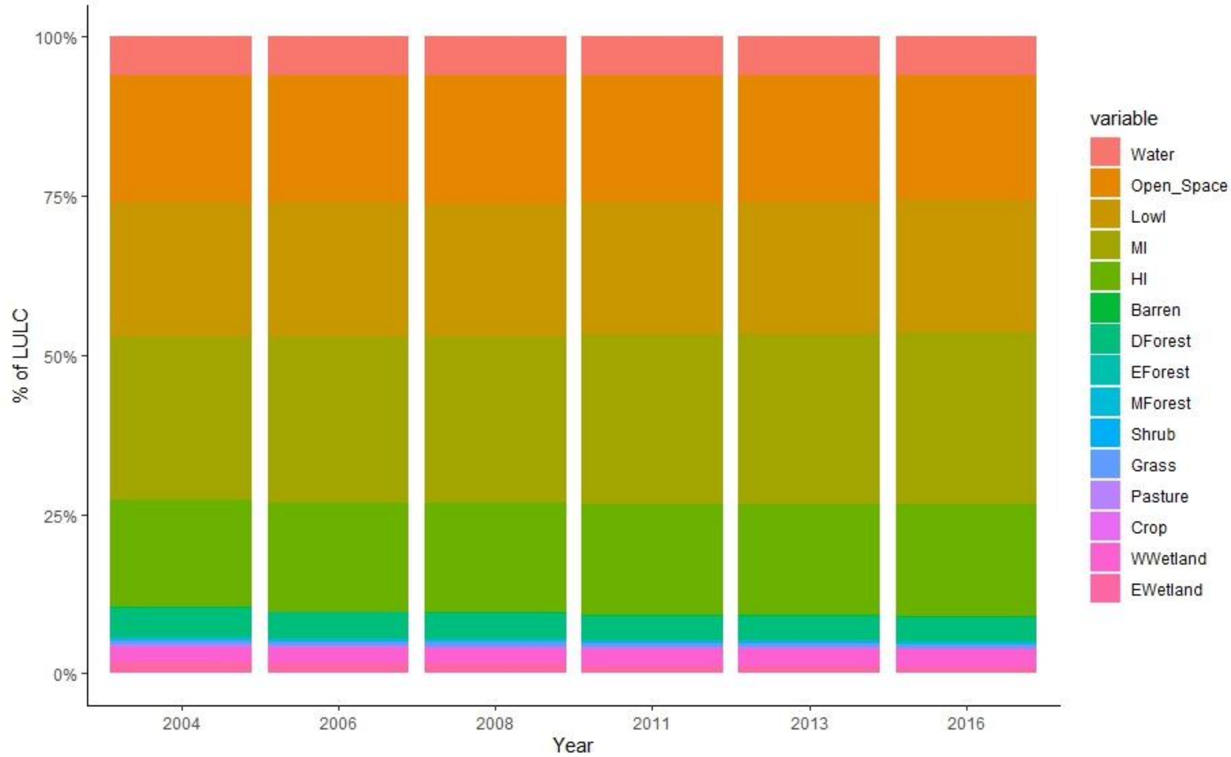


Figure 3-2 Land use type change from 2004 to 2016 for all the HUC 14 units in Passaic River

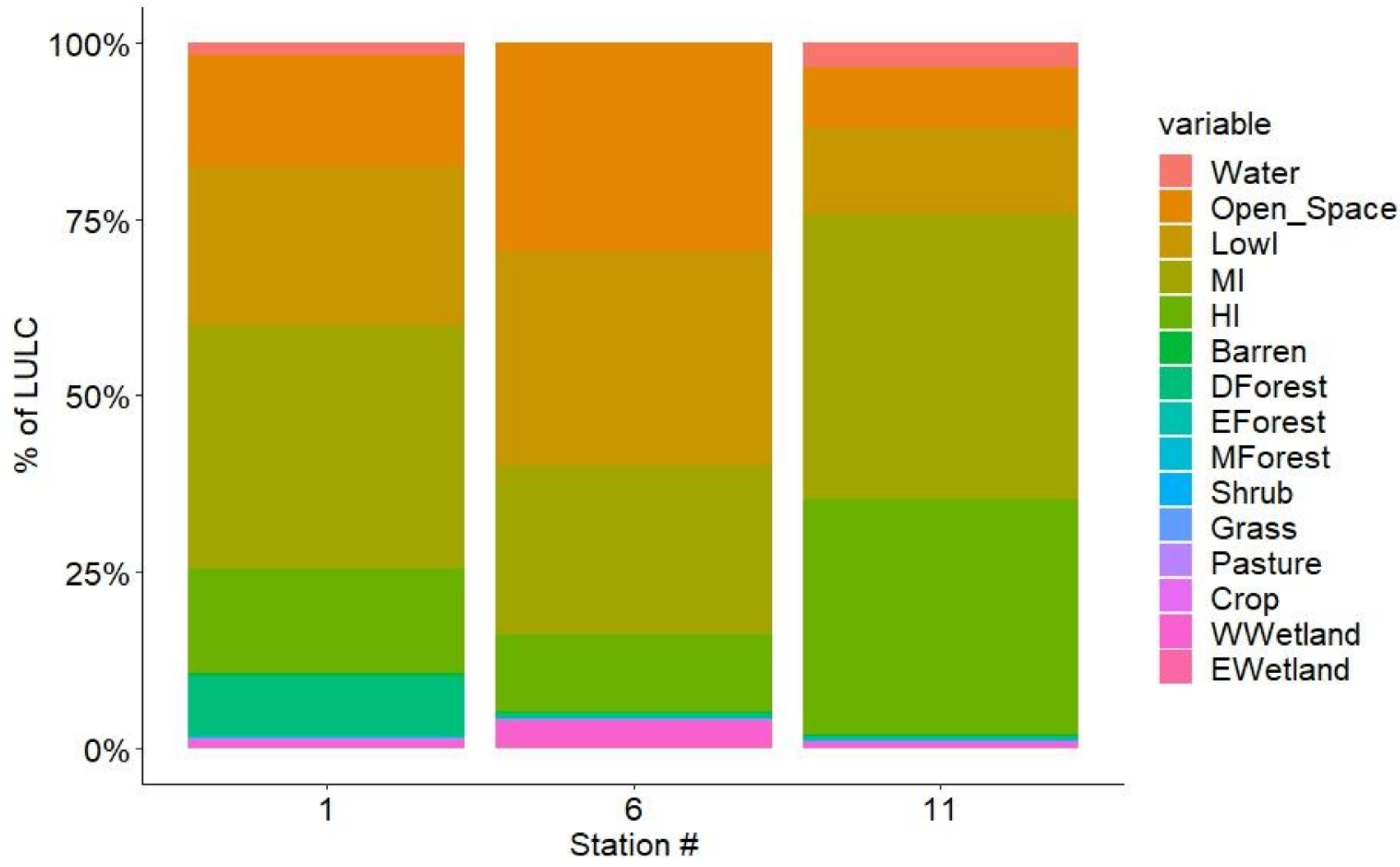


Figure 3-3 The average land use land cover type in each water quality monitoring station (St1, 6 and 11) from 2004 to 2016

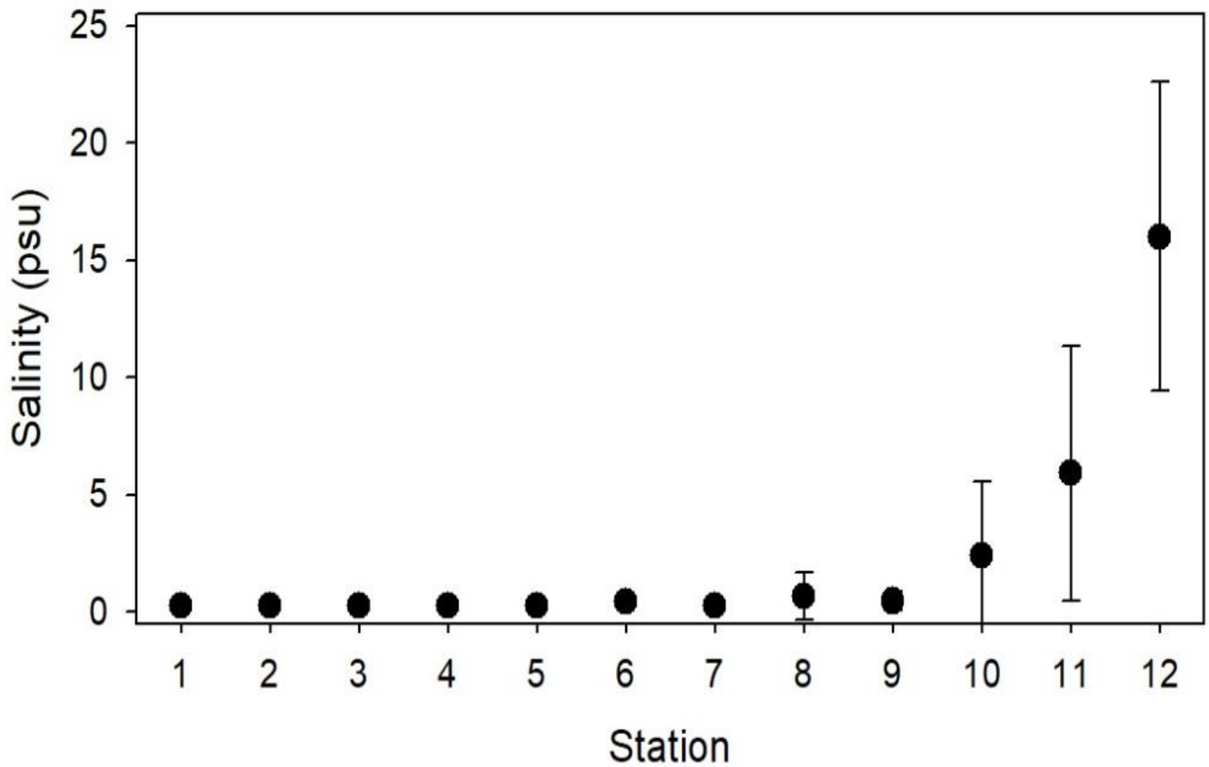


Figure 3-4 Average salinity of each monitoring station (2004-2013)

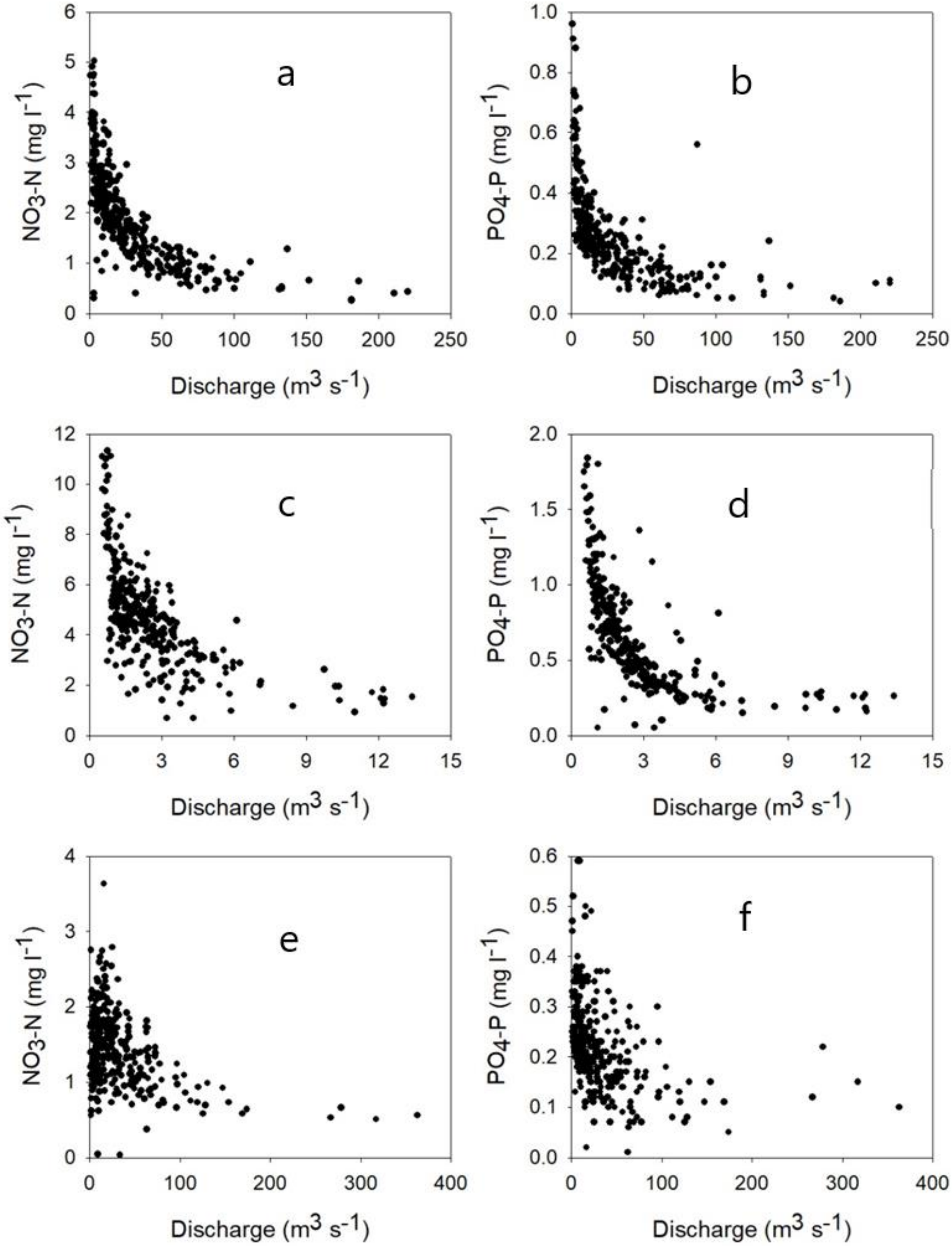


Figure 3-5 Relationship between nutrient (N and P) concentrations and river flow

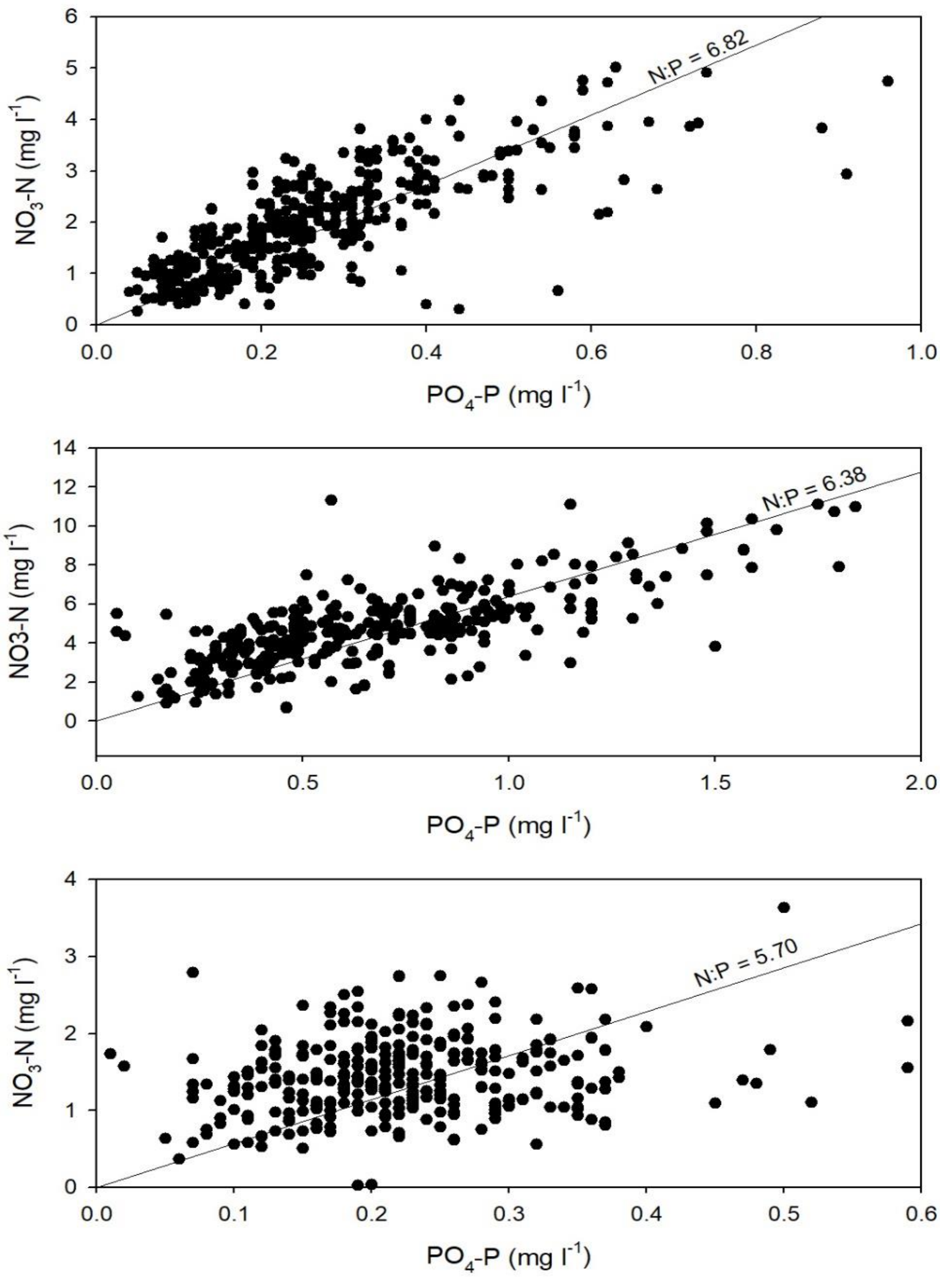


Figure 3-6 Comparison of N and P relationship in different water monitoring sites

Tables

Table 3-1 Numbers of CSOs and dry cleaners in each HUC 14 unit

HUC ID	CSO	Dry cleaner
2030103120020	0	5
2030103120030	0	6
2030103120040	0	3
2030103120050	0	12
2030103120070	4	9
2030103120080	5	12
2030103120090	0	26
2030103120100	0	12
2030103120110	14	10
2030103140050	0	4
2030103140060	0	8
2030103140070	0	10
2030103150010	0	15
2030103150020	0	27
2030103150030	1	18
2030103150040	16	11
2030103150050	7	2
2030103180010	0	4
2030103180070	0	6
2030103180090	0	2
2030103180100	11	8
2030104010010	7	12
2030104010020	28	1

Chapter 4 Estimation of Nutrient (N and P) Fluxes into Newark Bay, USA

Jing Nie[1], Sana Mirza[1], Michael Viteritto[1], Yuanyi Li[2], Benjamin B. Witherell[3], Yang Deng[1], Shinjae Yoo[4], Huan Feng[1, *]

1. Department of Earth and Environmental Studies, Montclair State University, Montclair, New Jersey 07043, USA
2. School of Marine Science and Technology, Tianjin University, Tianjin 300072, PRC
3. New Jersey Board of Public Utilities, Trenton, NJ 08625, USA
4. Computational Science Initiative, Brookhaven National Laboratory, Upton, NY 11973

*Corresponding author: Huan Feng

Email address: fengh@montclair.edu, Tel: 1-973-655-7549, Fax: 1-973-655-4072

Abstract

This study was conducted in northern New Jersey, USA, to estimate the nutrient (N and P) fluxes from the Passaic River, the Hackensack River and other sources into Newark Bay and the nutrient residence time in Newark Bay. Bi-weekly total inorganic nitrogen (TIN) and orthophosphate concentration data in the Passaic River, the Hackensack River, and Newark Bay for over 15 years (2004-2019) were collected along with daily river discharge data from the public database. The annual TIN and orthophosphate (ortho-P) loading from the Passaic River ranged from 915×10^{3} kg y^{-1} to 251×10^{4} kg y^{-1} and 94×10^{3} kg $y^{-1}$to 372×10^{3} kg y^{-1}, respectively. The annual TIN and ortho-P loading from the Hackensack River ranged from 3.13×10^{3} kg y^{-1} to 234×10^{3} kg y^{-1} and 0.28×10^{3} kg y^{-1} to 6.97×10^{3} kg y^{-1}, respectively. Seasonal variation results indicated that hurricane events highly increased TIN and ortho-P loading from riverine input and reduced residence time in Newark Bay.

Keywords: Total inorganic nitrogen, Orthophosphate, Flux, Newark Bay, Residence Time

1. Introduction

Riverine nutrient (N and P) fluxes have significant influences on the estuarine ecosystem. Superfluous nutrient loading can cause eutrophication in estuaries and bays when there is insufficient dilution or water exchange in the system (Yang et al., 2019). Due to the different geomorphological characteristics of riverine and estuarine systems, water quality properties can be different in both spatial and temporal scale, as well as reveal the terrestrial influences (Zhu et al., 2013a). Anthropogenic activities also increase the nutrient loading in coastal area and lead to hypoxia and water quality degradation (Irby & Friedrichs, 2019). Algal blooms caused by eutrophication can result in high risk of ecosystem. Even with the nutrient concentrations controlled within the criteria, previous studies showed that a freshwater system (e.g., lakes and rivers) can still be dominated by cyanobacteria bloom with warmer climate (Huo et al., 2019).

Many studies have been conducted to better understand the impact of high nutrients loading on aquatic systems. The main contaminant sources can be categorized as fertilizer, animal wastes, human sewage, household products, and agriculture field. Fertilizer is primarily used for agricultural purposes and lawn maintenance (Antweiler et al., 1996). A previous study in two freshwater river systems in an area in South Africa indicated that agriculture activities and informal settlement were the major causes of the high nutrient inputs into the river reaches (Malherbe, Le Maitre, Le Roux, Pauleit, & Lorz, 2019). Other studies have indicated that wastewater discharges and urban, or agricultural, stormwater runoff are the main causes of increased nutrient concentration in these systems (John Fillos & William R Swanson, 1975). It was reported that nutrient contamination could cause eutrophication in rivers or estuaries when dissolved inorganic nitrogen and phosphate concentrations in the waterbody reached high level (Huang et al., 2003). A study in Asia found that increased dissolved inorganic nitrogen (DIN) fluxes could cause high phosphorus and silicon limitations in most of estuary system in China, and these changes enhanced the possibilities of altering the phytoplankton communities in an estuary ecosystem (Wurtsbaugh et al., 2019).

Nutrient loading is also a main driven force for interannual hypoxia variability in some lakes and estuaries (Del Giudice, Zhou, Sinha, & Michalak, 2018; M. Li et al., 2016) . Agricultural use of fertilizers can greatly increase nutrient concentrations and, hence, impact the water quality (Teklu, Hailu, Wiegant, Scholten, & Van den Brink, 2018). Livestock can also add nutrients to the local ecosystem and increase ammonium and total phosphorus concentrations in rivers (Subalusky, Dutton, Njoroge, Rosi, & Post, 2018). A study conducted in Europe using SWAT model indicated that baseflow and surface runoff was the main pathways of nutrient pollution although plant uptake captured 58% of total nitrogen and 92% of total phosphates (Malagó, Bouraoui, Vigiak, Grizzetti, & Pastori, 2017). In the United States, a study conducted in Chesapeake Bay suggested that high nutrient loading increased the biological consumption of O_2 in the Bay. These oxygen depletion events caused detrimental impacts on organisms in estuarine ecosystems (M. Li et al., 2016).

Sediment redox condition could also affect the release of nutrients into the aquatic environment (Miao et al., 2006), and sediment resuspension could also affect nutrient concentrations in the water body (Kalnejais et al., 2010; Tengberg et al., 2003). It is clear that excess nutrient fluxes and high nutrient concentrations can adversely impact the riverine, estuarine and coastal water quality and ecosystems worldwide, and riverine nutrient discharge can be the major driver for ecosystem change (Ludwig et al., 2009). Mass flux of chemical constituents from a river or stream is often referred to as loading during a certain period (GESAMP, 1987). In theory, mass flux is the product of chemical constituent concentration and river/stream discharge by integration over time. In principle, flux estimation requires a continuous record of concentration and discharge. Ideally, fluxes of chemical constituents are typically estimated for periods in which there are several consecutive years with at least four sets of water-quality analyses data available for each year, which is to prevent a gap in a long-term continuous flux estimation period. Although discharge can be easily measured at a high frequency as needed, chemical constituent concentrations are measured much less frequently due to the large quantity of samples collected and the expense of collecting and analyzing samples.

With respect to all water quality problems, nutrients contamination is a problem that cannot be ignored. Wastewater discharge from treatment plants is one of the most common point source polluters to surface water. It is relatively easy to identify and control the point source pollution if the information regarding the discharge location is available. However, non-point source pollution, such as stormwater runoff from the agricultural and urban areas, is difficult to quantify and identify. Due to urbanization and population growth, coastal urban areas and agricultural areas have higher nutrient concentrations in water (S. Li et al., 2009). Many factors can affect the nutrients concentration in the aquatic environment. High nutrients concentration not only damage the aquatic ecosystem, but also endanger human health as well (Chaudhary et al., 2017). Therefore, integrated management methods were in high demand for identification and control of possible pollution contributes to the estuarine areas (Gaspar et al., 2017).

The Passaic River and Newark Bay have historically been severely contaminated by organic and inorganic contaminants due to the industrialization in these areas (Parette & Pearson, 2014). Based on the results from previous studies (Friedman & Lohmann, 2014; Parette & Pearson, 2014; Saba & Su, 2013), there is a practical need to conduct pattern and historical trend study to predict the pollution in this aquatic system. A previous study has shown that dissolved nitrate, ammonia, and phosphate concentrations in the Hackensack River were higher than those in the Passaic River and the Newark Bay indicating the nutrient sources to the water and sediment were mainly from sewage treatment plants during dry season and from combined sewer overflows during wet seasons (Hun Bok Jung, 2017). In order to quantify the nutrient input into the Newark Bay from the Passaic River, Hackensack River, and other sources nutrient fluxes from riverine and non-point sources were estimated. This study mainly focused on the long-term nutrient input into the Newark Bay with annual and monthly variations. Residence time and mass loading were also calculated to estimate the pollution time in the Newark Bay as they were affected by ocean water circulation.

2. Methods and materials

2.1 Study Area

The study area is the Passaic River, Hackensack River and Newark Bay, all located in northern New Jersey, USA. The Passaic River is a mainstream in northern New Jersey and one of the most polluted rivers in the United States (Hun Bok Jung, 2020). The Hackensack River spans both New York and New Jersey and is approximately 72 km long (Carswell, 1976). Newark Bay is a tidal bay at the confluence of the Passaic River and the Hackensack River, located wholly within New Jersey, with a rectangular shape of approximately 9 km (5.5 miles) long, varying in width from 1 to 2 km (0.6 to 1.2 miles) (Walker, McNutt, & Maslanka, 1999). It is connected to Upper New York Bay by the Kill Van Kull, and to Raritan Bay by the Arthur Kill (Figure 4-1). There are a number of water quality monitoring stations laid out by New Jersey Harbor Discharge Group (NJHDG) in the study area, of which four stations were chosen for this study (Figure 4-1). Specifically, they are two freshwater endmember stations in the Passaic River (Station 7 (St7)) and the Hackensack River (St 13), respectively, and two water quality monitoring stations in the Newark Bay (St 17 and St 18) (Figure 4-1). Three United States Geological Survey (USGS) discharge gage station (USGS 01389500, USGS 01391500 and US 01378500) were also chosen for obtaining the river discharge data (Figure 4-1).

2.2Nutrient data collection and analysis

All the water quality data used in this study were collected from available federal and state agency databases, including the NJHDG and the USGS, which covered a time period from 2004 to 2019. The nutrient (N and P) concentration data were collected from the NJHDG database for Stations 7, 13, 17 and 18, while the Passaic River and the Hackensack River flow data were obtained from USGS gaging stations at USGS 01389500, USGS 01391500 and USGS 01378500. There are also rainfall data in each station to indicate storm events.

St 7 is the last freshwater station that does exchange with salt water in the Passaic River; therefore, St 7 is the endmember of the freshwater with zero salinity. Because of this, St 7 was chosen to calculate the riverine nutrient fluxes from the Passaic River. As shown in Figure 1, the river flow at St 7 can be calculated by adding discharge data from USGS 01389500 in the Passaic River and discharge data from USGS 01391500 in the Saddle River together. In the Hackensack River, the endmember of freshwater station is St 13. USGS 01378500 is the nearest discharge gaging station to St 13; therefore, the data from St 13 and USGS 01378500 were used to estimate riverine nutrient flux from the Hackensack River. In Newark Bay, NJHDG monitored upper and lower-level nutrient concentrations at St 17 and St 18. In order to analyze the nutrient behavior in Newark Bay, the average daily, monthly and yearly nutrient concentrations at St 17 and St 18 were calculated, respectively. Also, the mean nutrient concentrations in Newark Bay were estimated by calculating the average nutrient concentrations at St 17 and St 18.

The NJHDG set a number of parameters to monitor the water quality, of which orthophosphate (PO_4^{3-}-P) (ortho-P) and total inorganic nitrogen (TIN) were chosen to represent the nutrient content in this study. Ortho-P represents inorganic phosphate in water. Normally, inorganic phosphate is used heavily in fertilizers and introduced into surface waters through runoff. The EPA standard methods (Association, Association, Federation, & Federation, 1915; EPA, 2012) state all the water sampling and sample treatment processes conducted in the field for nutrient (N and P) concentration analysis were not filtered. The results showed in nutrient concentrations were total concentrations including both particulate and dissolved fractions. In this situation, ortho-P data is more suitable than total phosphate data used for the purpose of this study. With oxygenated/anoxic conditions, ammonia-nitrogen and nitrite can convert to/from nitrate (Tan, Anastasi, & Chandra, 2022). Total inorganic nitrogen, which is the sum of ammonia-nitrogen, nitrate and nitrite can better indicate the nitrogen content dissolved in the water. Therefore, TIN and ortho-P concentrations were used as nutrient indicators in this study. Since there were some missing data in the 2004 – 2019 nutrient concentration dataset, Steinman interpolation method was used to interpolate the missing data as needed in this study (Fisher, Lowther, & Shene, 2004; Perillo & Piccolo, 1991; Stineman, 1980).

2.3 Nutrient fluxes calculation methods

In this study, riverine nutrient (TIN and ortho-P) fluxes were estimated using the nutrient concentration data from St 7 and St 13 and river flow data from USGS 01389500, USGS 01391500 and USGS 01378500. Nutrient (TIN and ortho-P) fluxes for a period of time were calculated according to a method recommended by the Joint Group of Experts on the Scientific Aspect of Marine Pollution (GESAMP, 1987), which is

$$\mathrm{F} = \frac{\mathrm{K} \cdot \sum_{i=1}^{n} C_i Q_i}{\sum_{i=1}^{n} Q_i} \ \overline{Q_r} \tag{1}$$

where K is the conversion factor, C_i (mg L^{-1}) is the instantaneous concentration associated with individual samples, Q_i (m^3 d^{-1}) is the instantaneous discharge at time of sampling, $\overline{Q_r}$ (m^3 d^{-1}) is the mean discharge for the period of record. This method is recommended when continuous discharge and non-continuous concentration data are available. The reliability of estimated fluxes in this method depends on the characteristics of the rivers and how representative is the flow-weighted mean concentration value derived from a relatively small size of data population.

2.4 Newark Bay volume estimation

A mesh to describe the New York/New Jersey Harbor bathymetry was created to represent the complex shoreline and water depth feature according to the hydrodynamic study (Figure 4-2a) (Y. Li et al., 2019). The spatial scale of the mesh was set to 100 m in the Newark Bay and the Newark bay mesh is getting from the NY/NJ mesh(Y. Li et al., 2019). The NY/NJ Harbor mesh data was originally input into ArcGIS Pro (Esri, Inc. Redlands, California, USA) using selection tools. Then, the water depth data in Newark Bay with latitude and longitude were obtained (Figure 4-2b). In the ArcGIS Pro, the coordinate system used in this study is WGS_1984_UTM_Zone_18N. The polygon tools in ArcGIS Pro were used to calculate the area of Newark Bay, which is estimated to be 15.7 km^2 (Figure 4-2c). The water depth data at the points in Figure 2b were obtained from a previous hydrodynamic study (Y. Li et al., 2019). Based on the water depth data in Newark Bay, the average depth was estimated to be 4.3m. The volume of Newark Bay was estimated to be 6.75×10^7 m^3 in this study, which is consistent with the result (6.26×10^7 m^3) reported by U.S. Army Corps of Engineers (USACE, 2010).

2.5 Freshwater discharge from the discharge points and combined sewage overflow sites

The freshwater discharge into the Newark Bay includes riverine input and fluxes from discharge points and non-point pollutions caused by anthropogenic activities. In this study, the major discharge points and Combined Sewage Overflow (CSO) sites were shown in Figure 4-3. The CSO discharge data were collected from GIS layer in Department of Environmental Protection (NJDEP) database and National Pollutant Discharge Elimination System (NPDES) database. The discharge sites data were collected from Water Pollutant loading Tool based on NPDES permit limit and Discharge monitoring report (DMR) data. The discharge sites data were collected by hydrologic unit code (HUC). The major discharge sites are from HUC 020301040203 (Figure 4-3). There were only 4 years' complete CSO data available. The data showed (Figure 4-4) the river discharge from Passaic River accounted for 68±13% of the total freshwater flow into Newark Bay, while the discharge from CSO sites around Newark Bay accounted for 3±2% of the total flow; therefore, the contribution from the CSOs were reasonably ignored in the flux calculation. Among all the discharge sites in all drainage area into Newark Bay, the freshwater discharge from discharge sites within HUC0203010402 accounted 27%±14% of the total freshwater discharge (Figure 4-4) and could not be ignored. There are 10 years (2011-2020) of flow data for these discharge sites in HUC0203010402 available for this study. Therefore, the freshwater input into Newark Bay should include the Passaic River, the Saddle River, the Hackensack River and all the discharges within HUC0203010402.

2.6 Freshwater residence time

In an estuarine system or a bay, the freshwater residence time is the flushing time that can be calculated as follows (Guo & Lordi, 2000; Monsen, Cloern, Lucas, & Monismith, 2002):

$$t_f = \frac{V_f}{Q_f} \tag{2}$$

of which

$$V_f = V \times f \tag{3}$$

and

$$f = \frac{S_{sw} - S_{bay}}{S_{sw}} \tag{4}$$

where f is the freshness which is a fraction of the freshwater in the system, S_{sw} is the end-member salinity of seawater in psu, and S_{bay} is the water salinity in the bay in psu, V is the volume of the system in m^3, t_f is the freshwater residence time (d), V_f is the volume of water in the system in m^3, Q_f is the volumetric flow rate through the system in $m^3\ d^{-1}$. In this study, S_{sw}=33.07 psu was used as the end member salinity of seawater according to the measurement at St 30 (latitude: -74.14600 longtitude:40.52000) from NJDEP, which also matches the salinity from New York Bight from previous studies (Blumberg, Ali Khan, & St. John, 1999; W. G. Zhang, Wilkin, & Schofield, 2010). Salinity in Newark Bay was from the field measurement at St 17 and St 18, which varied temporarily. The volumetric freshwater flow rate is based on the USGS gage stations in the Passaic River, the Saddle River, the Hackensack River and the discharge sites within HUC0203010402.

2.7 Mass Balance in the Newark Bay

The residence time of a substance in a reservoir can be described by (Monsen et al., 2002; Nauman, 2008).

$$t_R = \frac{M_R}{F_s} \tag{5}$$

of which

$$M_R = C_R \times V_R \tag{6}$$

where t_R is the residence time of a substance in the reservoir in d, M_R is the total amount of the substance in the reservoir in kg, and F_s is the total flux of the substance into or out of the reservoir in kg d^{-1}, C_R is the concentration of the substance in the reservoir in g l^{-1} or kg m^3, and V_R is the volume of the reservoir in m^3. We assume the nutrient is carried by freshwater into Newark Bay, so the residence time of freshwater (t_f) will equal to the residence time of a substance in the reservoir (t_R). In this study, the total flux of nutrients into Newark Bay is the sum of riverine flux (i.e., fluxes from the Passaic River and the Hackensack River) and the rest of sources including anthropogenic point and non-point sources. Therefore, Equation (6) can be altered to

$$t_R = \frac{M_R}{F_{riv} + F_{other}} \tag{7}$$

$$F_{riv} = C_{riv} \times Q_{riv} \tag{8}$$

where F_{riv} is the riverine flux in kg d^{-1}, F_{other} is other fluxes in kg d^{-1}, C_{riv} is the concentration of the substance in river in g l^{-1} or kg m^3, and Q_{riv} is the river discharge in m^3 d^{-1}. The rest of the parameters are the same as defined before.

The major sources of nutrient input into the Newark Bay include riverine sources (the Passaic and Hackensack Rivers), other point sources (such as combined sewage overflows (CSOs), wastewater treatment plants and discharging point along the bay) and non-point sources in the watershed along the bay (Y. Li et al., 2019). To better quantify the different sources into the Newark Bay, the sources other than riverine sources from St 7 and St 13 were categorized as other sources including point and non-point sources.

3. Results and Discussion

3.1 Annual Riverine Nutrient Loading to Newark Bay

The annual TIN and ortho-P loadings from the Passaic River as recorded at St 7 ranged from 1016×10^3 kg y^{-1} to 2864×10^3 kg y^{-1} and 94.3×10^3 kg y^{-1} to 372×10^3 kg y^{-1}, respectively (Figure 4-5). As shown in Figure 5, annual TIN and ortho-P loadings from the Passaic River (St7) has an overall decreasing trend from 2004 to 2016. From 2004 to 2016, TIN loading had an overall decrease of 54%. From 2004 to 2009, TIN loading kept in a steady state. From 2009 to 2011; however, the TIN loading increased by 44%. Then, the TIN loading decreased by 65% from 2011 to 2016. As to ortho-P loading, there are three high peaks appearing in 2006, 2008 and 2011. From 2009 to 2011, ortho-P loading increased by 78%. From 2011 to 2016, ortho-P loading decreased by 75%. Overall ortho-P loading decreased by 66% from 2004 to 2016. From 2016 to 2019, TIN and ortho-P loading increased by 116% and 86.2%, respectively.

The annual TIN and ortho-P loadings from the Hackensack River (St13) varied from 5.15×10^3 kg y^{-1} to 273×10^3 kg y^{-1} and 0.279×10^3 kg y^{-1} to 6.97×10^3 kg y^{-1}, respectively (Figure 4-5). The overall trend of TIN and ortho-P loadings decreased from 2004 to 2016. TIN and ortho-P loadings decreased by 97% and 95% from 2004 to 2016. Figure 4-5 shows extraordinary peaks in 2011. TIN loading increased by 76% from 2004 to 2011 and decreased by 98% from 2011 to 2016. As shown in Figure 4-5, ortho-P loading increased by 8.8% from 2004 to 2011 and decreased by 96% from 2011 to 2016. Compared TIN and ortho-P loadings, the peak in TIN trend was idiosyncratic high. Both TIN and ortho-P loadings increased by 12 and 8 times respectively from 2016 to 2019.

The high peaks in 2011 occurred in both the Passaic River (St7) and the Hackensack River (St13). According to Figure 4-6, the high peak discharges in 2005, 2011 (both in St 7 and St 13) were not related to rainfall; however, based on historical record, there were 6 hurricanes in New Jersey. Hurricanes Maria and Nate (2005), Irene (2011) all passed North New Jersey (Gump, Klemm, van Westendorp, Wood, & Doroba, 2017). Hurricane Irene is a category 3 hurricane, and highly impacted the water quality in the Passaic River and Hackensack River (Saleh et al., 2017). Different from other flooding events, Hurricane Irene was the first landfall hurricane in New Jersey since 1903, which resulted in high discharge both in Passaic and Hackensack River.

In St 7 (Figure 4-7) the TIN and ortho-P concentrations were diluted by the discharge. Regression for TIN loading and discharge showed a R^2 of 0.65 with $p<0.05$; regression for ortho-P loading and discharge showed a R^2 of 0.73 with $p<0.05$. Ortho-P loading showed a better fit with the discharge data than TIN loading. At St 13 (Figure 4-8), regression for TIN and discharge showed a R^2 of 0.92 with $p<0.05$; regression for ortho-P and discharge showed a R^2 of 0.84 with $p<0.05$. The results indicated that between the two stations, nutrient loading from St 13 was most likely from the natural resource, and nutrient loading from St 7 was more affected by anthropogenic input (Hun Bok Jung, Richards, & Fitzgerald, 2020). High discharge in Hurricane Irene results in high nutrients loading in these two rivers based on the nutrient loading and discharge relations.

3.2 Seasonal Variation from Riverine Resources

Based on annual riverine nutrient loading calculations, hurricane events have a significant impact on the discharge and nutrient loading in both rivers. Figure 4-6 shows high rainfall events (storm events) also had a high impact on the discharge. According to Figure 4-7 & 4-8, high discharge resulted in high nutrient loading in both stations (St 7 and St 13). In order to better estimate the seasonal variation of nutrient loading from the Passaic River and the Hackensack River, hurricane and storm events (rainfall>2.54mm) (Hopkins, Bhaskar, Woznicki, & Fanelli, 2020; Bano Mehdi, Schürz, Grath, & Schulz, 2021) were separated during the monthly-averaged nutrient loading calculations. To exam the difference among four seasons, one-way ANOVA and Tukey's test were conducted under normal, hurricane, and storm conditions. The test results are shown in Table 4-1 for TIN loading and Table 4-2 for ortho-P loading. Both tests are conducted for TIN and ortho-P loading from Passaic River and Hackensack River, freshwater residence time in Newark Bay, TIN mass and ortho-P mass in Newark Bay, and TIN and ortho-P loading from other source.

3.3 Seasonal variation in Passaic River

The seasonal variation of TIN and ortho-P loading are shown in Figure 9 with hurricane, storm, and normal conditions at St 7. In Table 4-1, under normal and hurricane weather conditions, winter and spring have 60%-70% of the total TIN loading. However, under storm conditions, spring and summer have 70% of the total TIN loading. The increase in TIN loading of summer season results from the high discharge with the high rainfall occurs from late spring to early summer (Figure 4-6). In Table 4-2, there is no significant difference among seasons under normal, hurricane and storm conditions. Compared to Table 4-1 results in TIN loading, with the same discharge data, ortho-P loading shows no seasonal difference. The results indicated that ortho-P source from Passaic River is not from natural process and has a high potential of anthropogenic input.

The ANOVA test was also conducted on the three different weather conditions for TIN and ortho-P loadings to identify the impact of hurricane and storm events. The results are shown in Tables 4-1 and 4-2. The Total inorganic nitrogen (TIN) loading from hurricane event has a significant difference from storm event. Hurricanes Maria and Nate, in 2005, and Irene in 2011 all hit in the fall. The TIN loading under hurricane conditions shows a high increase in Fall other than storm and normal conditions (Figure 4-8). There is no significant difference shown in ortho-P loading from three weather conditions. The results further proved that ortho-P loading input was mainly from anthropogenic activities in Passaic River.

3.4 Seasonal variation in Hackensack River

The seasonal variation of TIN and ortho-P loading under hurricane, storm and normal weather conditions at St 13 are shown in Figure 4-10. There is no significant seasonal variation of TIN loading under normal and storm conditions (Table 4-1). Under hurricane conditions, 60% of total TIN loading appears in spring season. Ortho-P loading has a similar trend and spring season contributed 64% of total ortho-P loading under hurricane conditions (Table 4-2). Spring is the major season that accounts for most of the nutrients because the discharge peaks in spring in 2011(Figure 4-6). The results of no seasonal variation in Hackensack River of TIN and ortho-P are the same which indicate that most of the nutrient source are from natural process. This is because the concentration of orthophosphate/TIN in water depends on a balance between inputs (such as runoff and wastewater) and removal processes (such as uptake by plants and algae, sedimentation, and adsorption onto particles). While the relative importance of these inputs and removal processes may vary over time, the overall balance is thought to remain relatively constant. There are some studies that suggest that TIN loading may not show a clear seasonal pattern in certain ecosystems. For example, a study of an intermittent stream draining an unpolluted Mediterranean forested catchment (10.5 km^2) in Catalonia (Spain) found that TIN loading did not show a significant seasonal variation over the course of a year (Bernal, Butturini, & Sabater, 2005).

An ANOVA test was also conducted on three different weather conditions for nitrate and ortho-P loadings to identify the impact of hurricane and storm events. Both TIN and ortho-P loadings show significant difference between hurricane and storm conditions. Even though there is not much seasonal variations of TIN and ortho-P loading in Hackensack River, hurricane events show a high impact to nutrient loading.

By comparison of seasonal TIN and ortho-P loading from St 7 and St 13, Spring loading makes up the largest portion of the total loading (nitrate and ortho-P) under normal, hurricane and storm conditions. A similar trend was observed in a study conducted in the Red River Basin, Manitoba, Canada (Rattan et al., 2017). The study indicated that high loading occurred during snowmelt period (March to May). Hurricane conditions play a more important role of TIN and ortho-P loading in Hackensack River than in Passaic River.

3.5 Residence Time in Newark Bay

Residence time represents the time cost by nutrients to flux out of the Newark Bay. A longer residence time means nutrients remain in the bay for a longer period of time. Figure 4-11 (A) shows the yearly averaged residence time in Newark Bay from 2011 to 2019 ranged from 5.1 days to 12.0 days. There were extreamly short residence times in 2011, which indicates that hurricanes impact nutrients residence times in Newark Bay. So the residence time were categorized into hurricane, storm and normal conditions to study the seasonal variation.

Residence time shows no seasonal difference in hurricane conditions but seasonal variation in storm and normal conditions (Table 4-1). Summer and fall contribute 60%-70% of total residence time. ANOVA test of three weather conditions is also conducted to test the impact of hurricane and storm events. Results show that residence time in hurricane conditions is significantly different from storm and normal conditions (Table 4-1). Normal conditions residence time is 1.8 times more than hurricane conditions. This indicates under hurricane conditions; the nutrients are easily flushed out into the ocean. The nutrient residence time in Newark Bay is shorter relative to that in the Chesapeake Bay, which is 180 days based on a study conducted during 1980-2012 (Du & Shen, 2016). A similar study conducted in the Bay of Gdańsk, Baltic Sea showed 53-60 days of residence time. The result from that study indicates that the nutrient and other pollutants in Newark Bay have a relatively less residence time than that in nearby Chesapeake Bay and some other places.

3.6 Nutrient Mass in Newark Bay

TIN and ortho-P mass were calculated by multiplying nutrient concentration and total Newark Bay volume. The TIN and ortho-P concentration were the average of St 17& 18. Figure 4-12(A) shows yearly averaged nutrient (TIN and ortho-P) mass in Newark Bay from 2004 to 2019. TIN mass in Newark Bay ranges from 54.4×10^3 kg to 189×10^3 kg; ortho-P mass in Newark Bay ranges from 6.8×10^3 kg to 11.8×10^3 kg. TIN and ortho-P mass has an overall decreasing trend in Newark Bay. TIN mass decreased 71% from 2004 to 2015 and increased 8% from 2015 to 2019. Ortho-P decreased 42% from 2004 to 2019.

Figure 4-12 (B&C) provides a comparison of seasonal variations of TIN (B) and ortho-P (C) under hurricane, storm, and normal weather conditions in Newark Bay from 2004 to 2019. Table 4-1 shows that there is no seasonal varitation of TIN mass under normal and storm conditions. Hurricane conditions bring the spring season to 60% of total TIN mass in Newark Bay. Ortho-P mass has seasonal variation under normal, hurricane and storm conditions (Table 4-2). Summer and fall together have 60-80% of the total ortho-P mass under three weather conditions. Hurricane and storm conditions have no imapct on the nutrient mass in Newark Bay. In this case, seasonal variation plays a more important role than hurricane and storm events in ortho-P mass. There is no impact from weather conditions to TIN mass in the bay.

3.7 Other Nutrient Source Input to Newark Bay

Other nutrient source inputs were not limited to point sources contributing to TIN and ortho-P loadings to Newark Bay. Other sources include all the point and non-point nutrient sources located below St 7 in the Passaic River and St13 in the Hackensack River (Figure 4-3). Figure 4-13 (A) shows yearly averaged nutrient load from other sources to Newark Bay from 2011 to 2019. The TIN and ortho-P loading from other sources ranges from 5.4×10^3 kg to 20.9×10^3 kg and from 0.3×10^3 kg to 0.8×10^3 kg respectively. The TIN and ortho-P loading from other sources decreased 74% and 62% from 2011 to 2017 respectively. The calculations for the other source of nitrate and load are based on the assumption that the residence time of freshwater equals to the residence time of nutrients.

There is significant seasonal variation of TIN loading under normal and storm conidtions and no variation under hurricane conditions (Table 4-1) . Under normal and storm conditions, summer and spring together contributes 60%-70% of the total TIN loading. Ortho-P loading shows the similar trend (Table 4-2). However, for ortho-P loading, summer and fall together contributes 60%-90% of the total loading under normal and storm conditions. Both TIN and ortho-P loading shows significant difference under diffferent weather conidtions. In this case, hurricane conditions eliminate the seasonal varitation of TIN and ortho-P loading from other sources.

Seasonal varitation happends both ortho-P loading from other sources and ortho-P mass in Newark Bay under normal and storm conditions. However, both Passaic River and Hackensack River don’t show any seasaonl variation under these conditions. Thus, mostly the seasonal variation in ortho-P mass in Newark Bay is from other source. For TIN mass in Newark Bay, there is no seasonal variation under normal and storm conditions. TIN loading from Passaic River and Other souces shows significant seasonal variation. TIN loading from Hackensack River shows no variation. Thus, TIN loading from Hackensack River is important on changes of TIN mass in Newark Bay.

Under hurricane conditions, both Passaic River and Hackensack River show seasonal variation of TIN loading and spring contributes highest loading. However, with no seasonal variation of TIN loading from other sources. TIN mass in Newark Bay results in just a little bit higher spring loading. Under hurricane conditions, There is no seasonal variation of ortho-P loading from Passaic River and other sources. Hackensack River has seasonal varitaion with a highest spring loading. However, ortho-P mass in Newark Bay shows seasonal variation with higher summer and fall loading. Thus, under hurricane conditions, ortho-P mass may come from the inner sediments of the Newark Bay as the mass calculation is based on the shallow water ortho-P concentratoin.

Under hurricane conditions, both ortho-P loading from other sources and ortho-P mass in Newark Bay shows high peak in Summer and lowest in Winter, which is due to the hurricane season is late August and early September. However, nitrate shows high peak in Spring and lowerst in Summer. The major reason is that in Summer, with high temperature, there will be less dissolved oxgen in water resulting in more nitrite coverting from nitrate (Bristow et al., 2017). Under normal conditions, ortho-P loading from St7 and St13 shows high peak in Spring; ortho-P mass in Newark Bay shows high peak in Summer; ortho-P loading from other sources shows high peak in Fall. The differences shows in different sources indicated that most ortho-P fluxes were major from Point Source which related to anthropogenic activities along the Newark Bay and Hackensack River (Shin, Artigas, Hobble, Lee, & assessment, 2013).

4. Conclusion

This study investigated yearly and seasonal variations of TIN and ortho-P fluxes in Newark Bay and residence time of TIN and ortho-P in Newark Bay. Other sources fluxes were also calculated based on the riverine TIN and ortho-P flux input and nutrient mass in Newark Bay. The yearly results showed that hurricane events peak the nutrient loading from riverine and reduce the residence time in Newark Bay. The annual TIN and ortho-P loadings from the Passaic River ranged from 1016×10^3 kg y^{-1} to 2864×10^3 kg y^{-1} and 94.3×10^3 kg y^{-1} to 372×10^3 kg y^{-1} and is the major riverine nutrient source to Newark Bay.

The seasonal variation study showed that in Passaic River, winter and spring have a higher TIN loading under normal and hurricane conditions. Storm events move higher loading to summer instead of winter. In Hackensack River, there is no seasonal variation of TIN loading under normal and storm conditions. Under hurricane conditions, TIN loading shows higher amount in spring and winter which is the similar trend with Passaic River. Both Passaic River and Hackensack River show no seasonal variations in ortho-P loading under three weather conditions. Residence time of nutrients are highly reduced by the hurricane events. Under normal and storm conditions, seasonal changes in ortho-P mainly from ortho-P loading from other sources. TIN loading from Hackensack River is important to TIN mass variation in Newark Bay. Under hurricane conditions, TIN mass in Newark Bay increased a little in spring as a result of higher spring loading from Passaic River and Hackensack River. Otho-P mass might come mainly from the inner sediments in Newark Bay under hurricane conditions.

Reference

Antweiler RC, Goolsby DA, Taylor HE. Nutrients in the Mississippi river. US GEOLOGICAL SURVEY CIRCULAR USGS CIRC 1996: 73-86.

Association APH, Association AWW, Federation WPC, Federation WE. Standard methods for the examination of water and wastewater. Vol 2: American Public Health Association., 1915.

Bernal S, Butturini A, Sabater FJB. Seasonal variations of dissolved nitrogen and DOC: DON ratios in an intermittent Mediterranean stream. 2005; 75: 351-372.

Blumberg AF, Ali Khan L, St. John JP. Three-dimensional hydrodynamic model of New York Harbor region. Journal of Hydraulic Engineering 1999; 125: 799 - 816.

Bristow LA, Callbeck CM, Larsen M, Altabet MA, Dekaezemacker J, Forth M, et al. N2 production rates limited by nitrite availability in the Bay of Bengal oxygen minimum zone. 2017; 10: 24-29.

Carswell L. Appraisal of water resources in the Hackensack River basin, New Jersey. J Water-Resources Investigations Report 1976; 76: 74.

Chaudhary M, Mishra S, Kumar A. Estimation of water pollution and probability of health risk due to imbalanced nutrients in River Ganga, India. International Journal of River Basin Management 2017; 15: 53-60.

Del Giudice D, Zhou Y, Sinha E, Michalak AM. Long-term phosphorus loading and springtime temperatures explain interannual variability of hypoxia in a large temperate lake. 2018; 52: 2046-2054.

Du J, Shen J. Water residence time in Chesapeake Bay for 1980–2012. Journal of Marine Systems 2016; 164: 101-111.

EPA. Phosphorus, https://archive.epa.gov/water/archive/web/html/vms56.html, 2012.

Fillos J, Swanson WR. The release rate of nutrients from river and lake sediments. Journal (Water Pollution Control Federation) 1975; 47: 1032-1042.

Fisher J, Lowther J, Shene C. Curve and surface interpolation and approximation: knowledge unit and software tool. ACM SIGCSE Bulletin 2004; 36: 146-150.

Friedman CL, Lohmann R. Comparing sediment equilibrium partitioning and passive sampling techniques to estimate benthic biota PCDD/F concentrations in Newark Bay, New Jersey (USA). Environmental Pollution 2014; 186: 172-179.

Gaspar R, Marques L, Pinto L, Baeta A, Pereira L, Martins I, et al. Origin here, impact there—The need of integrated management for river basins and coastal areas. Ecological Indicators 2017; 72: 794-802.

GESAMP. Land/Sea Boundary Flux of Contaminants: Contributions from Rivers. Report and Studies No. 32. GESAMP - IMP/FAO/UNESCO/WMO/WHO/IAEA/UN/UNEP Joint Group of Experts on the Scientific Aspects of Marine Pollution, New York, 1987, pp. 172.

Gump D, Klemm A, van Westendorp C, Wood D, Doroba J. Hurricane Maria, 2017 Hydrographic Response for Puerto Rico and US Virgin Islands by the NOAA Ship Thomas Jefferson. AGU Fall Meeting Abstracts. 2017, 2017, pp. NH23E-2857.

Guo Q, Lordi GP. Method for quantifying freshwater input and flushing time in estuaries. Journal of Environmental Engineering 2000; 126: 675-683.

Hopkins KG, Bhaskar AS, Woznicki SA, Fanelli RMJHp. Changes in event-based streamflow magnitude and timing after suburban development with infiltration-based stormwater management. 2020; 34: 387-403.

Huang X, Huang L, Yue W. The characteristics of nutrients and eutrophication in the Pearl River estuary, South China. Marine Pollution Bulletin 2003; 47: 30-36.

Huo S, He Z, Ma C, Zhang H, Xi B, Xia X, et al. Stricter nutrient criteria are required to mitigate the impact of climate change on harmful cyanobacterial blooms. Journal of Hydrology 2019; 569: 698-704.

Irby ID, Friedrichs M. Evaluating confidence in the impact of regulatory nutrient reduction on Chesapeake Bay water quality. Journal of Estuaries Coasts 2019; 42: 16-32.

Jung HB. Nutrients and heavy metals contamination in an urban estuary of Northern New Jersey. J Geosciences 2017; 7: 108.

Jung HB. Geochemical and hydrological study of coastal groundwater discharging to an urban estuary in northern New Jersey. J Environmental Earth Sciences 2020; 79: 1-14.

Jung HB, Richards J, Fitzgerald A. Temporal and spatial variations of water quality in the Newark Bay Estuary. Regional Studies in Marine Science 2020: 101589.

Kalnejais LH, Martin WR, Bothner MH. The release of dissolved nutrients and metals from coastal sediments due to resuspension. Marine Chemistry 2010; 121: 224-235.

Li M, Lee YJ, Testa JM, Li Y, Ni W, Kemp WM, et al. What drives interannual variability of hypoxia in Chesapeake Bay: Climate forcing versus nutrient loading? Geophysical Research Letters 2016; 43: 2127-2134.

Li S, Liu W, Gu S, Cheng X, Xu Z, Zhang Q. Spatio-temporal dynamics of nutrients in the upper Han River basin, China. Journal of Hazardous Materials 2009; 162: 1340-1346.

Li Y, Feng H, Zhang H, Sun J, Yuan D, Guo L, et al. Hydrodynamics and water circulation in the New York/New Jersey Harbor: A study from the perspective of water age. Journal of Marine Systems 2019; 199: 103219.

Ludwig W, Dumont E, Meybeck M, Heussner S. River discharges of water and nutrients to the Mediterranean and Black Sea: major drivers for ecosystem changes during past and future decades? Progress in Oceanography 2009; 80: 199-217.

Malagó A, Bouraoui F, Vigiak O, Grizzetti B, Pastori M. Modelling water and nutrient fluxes in the Danube River Basin with SWAT. J Science of the Total Environment 2017; 603: 196-218.

Malherbe H, Le Maitre D, Le Roux J, Pauleit S, Lorz C. A simplified method to assess the impact of sediment and nutrient inputs on river water quality in two regions of the southern coast of South Africa. J Environmental management 2019; 63: 658-672.

Mehdi B, Schürz C, Grath B, Schulz KJSoTTE. Storm event impacts on in-stream nitrate concentration and discharge dynamics: A comparison of high resolution in-situ measured data with model simulations. 2021; 755: 143406.

Miao S, DeLaune R, Jugsujinda A. Influence of sediment redox conditions on release/solubility of metals and nutrients in a Louisiana Mississippi River deltaic plain freshwater lake. Science of the Total Environment 2006; 371: 334-343.

Monsen NE, Cloern JE, Lucas LV, Monismith S. A comment on the use of flushing time, residence time, and age as transport time scales. Limnology & Oceanography 2002; 47: 1545-1553.

Nauman EB. Residence time theory. Industrial Engineering Chemistry Research 2008; 47: 3752-3766.

Parette R, Pearson WN. 2, 4, 6, 8-Tetrachlorodibenzothiophene in the Newark Bay Estuary: The likely source and reaction pathways. Chemosphere 2014; 111: 157-163.

Perillo GME, Piccolo MC. An interpolation method for estuarine and oceanographic data. Computers & Geosciences 1991; 17: 813-820.

Rattan K, Corriveau J, Brua R, Culp J, Yates A, Chambers P. Quantifying seasonal variation in total phosphorus and nitrogen from prairie streams in the Red River Basin, Manitoba Canada. Science of the Total Environment 2017; 575: 649-659.

Saba T, Su S. Tracking polychlorinated biphenyls (PCBs) congener patterns in Newark Bay surface sediment using principal component analysis (PCA) and positive matrix factorization (PMF). Journal of hazardous materials 2013; 260: 634-643.

Saleh F, Ramaswamy V, Wang Y, Georgas N, Blumberg A, Pullen J. A multi-scale ensemble-based framework for forecasting compound coastal-riverine flooding: The Hackensack-Passaic watershed and Newark Bay. Advances in Water Resources 2017; 110: 371-386.

Shin JY, Artigas F, Hobble C, Lee Y-S, assessment. Assessment of anthropogenic influences on surface water quality in urban estuary, northern New Jersey: multivariate approach. J Environmental monitoring 2013; 185: 2777-2794.

Stineman RW. A consistently well-behaved method of interpolation. Creative Computing 1980; 6: 54-57.

Subalusky AL, Dutton CL, Njoroge L, Rosi EJ, Post DM. Organic matter and nutrient inputs from large wildlife influence ecosystem function in the Mara River, Africa. Ecology 2018; 99: 2558-2574.

Tan JF, Anastasi A, Chandra SJCOiE. Electrochemical detection of nitrate, nitrite and ammonium for on-site water quality monitoring. 2022; 32: 100926.

Teklu BM, Hailu A, Wiegant DA, Scholten BS, Van den Brink PJ. Impacts of nutrients and pesticides from small-and large-scale agriculture on the water quality of Lake Ziway, Ethiopia. Environmental Science & Pollution Research 2018; 25: 13207-13216.

Tengberg A, Almroth E, Hall P. Resuspension and its effects on organic carbon recycling and nutrient exchange in coastal sediments: in situ measurements using new experimental technology. Journal of Experimental Marine Biology and Ecology 2003; 285: 119-142.

USACE. US Army Corps of Engineers: Newark Bay, 2010.

Walker WJ, McNutt RP, Maslanka CK. The potential contribution of urban runoff to surface sediments of the Passaic River: sources and chemical characteristics. Chemosphere 1999; 38: 363-377.

Wu G, Cao W, Wang F, Su X, Yan Y, Guan Q. Riverine nutrient fluxes and environmental effects on China's estuaries. Science of The Total Environment 2019; 661: 130-137.

Yang S, Büttner O, Kumar R, Jäger C, Jawitz JW, Rao P, et al. Spatial patterns of water quality impairments from point source nutrient loads in Germany's largest national River Basin (Weser River). Science of The Total Environment 2019; 697: 134145.

Zhang WG, Wilkin JL, Schofield OME. Simulation of Water Age and Residence Time in New York Bight. Journal of Physical Oceanography 2010; 40: 965-982.

Zhu W, Tian YQ, Yu Q, Becker BL. Using Hyperion imagery to monitor the spatial and temporal distribution of colored dissolved organic matter in estuarine and coastal regions. Remote Sensing of Environment 2013; 134: 342-354.

Statements &Declarations

Funding

This work was supported in part by Montclair State University Graduate Assistantship, and Montclair State University's Faculty Scholarship Program.

Compliance with ethical standards

Conflict of Interest On behalf of all authors, the corresponding author declares that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Human and animal rights This article does not contain any studies with human or animal subjects performed by any of the authors.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Credit authorship contribution statement

Jing Nie: conceptualization, designing and writing—original draft.

Huan Feng: supervision, writing, reviewing, and editing—original draft.

Sana Mirza: reviewing and editing—original draft

Michael Viteritto: reviewing and editing—original draft

Yuanyi Li: map design and data supply—original draft

Benjamin B Witherell: concept guidance and data supply—original draft

Yang Deng: reviewing and editing—original draft

Shiniae Yoo: reviewing and editing—original draft

Data availability

All the data is from the online open data source mentioned in the study.

Acknowledgement

This work was supported in part by the Montclair State University's Graduate Assistantship (JN, SM and MV). The authors would like to give thanks to Dr. Micheal Boufadel, Chief Editors of Marine Pollution Bulletin, and the two anonymous reviewers for their comments and advice which have improved the quality of this manuscript.

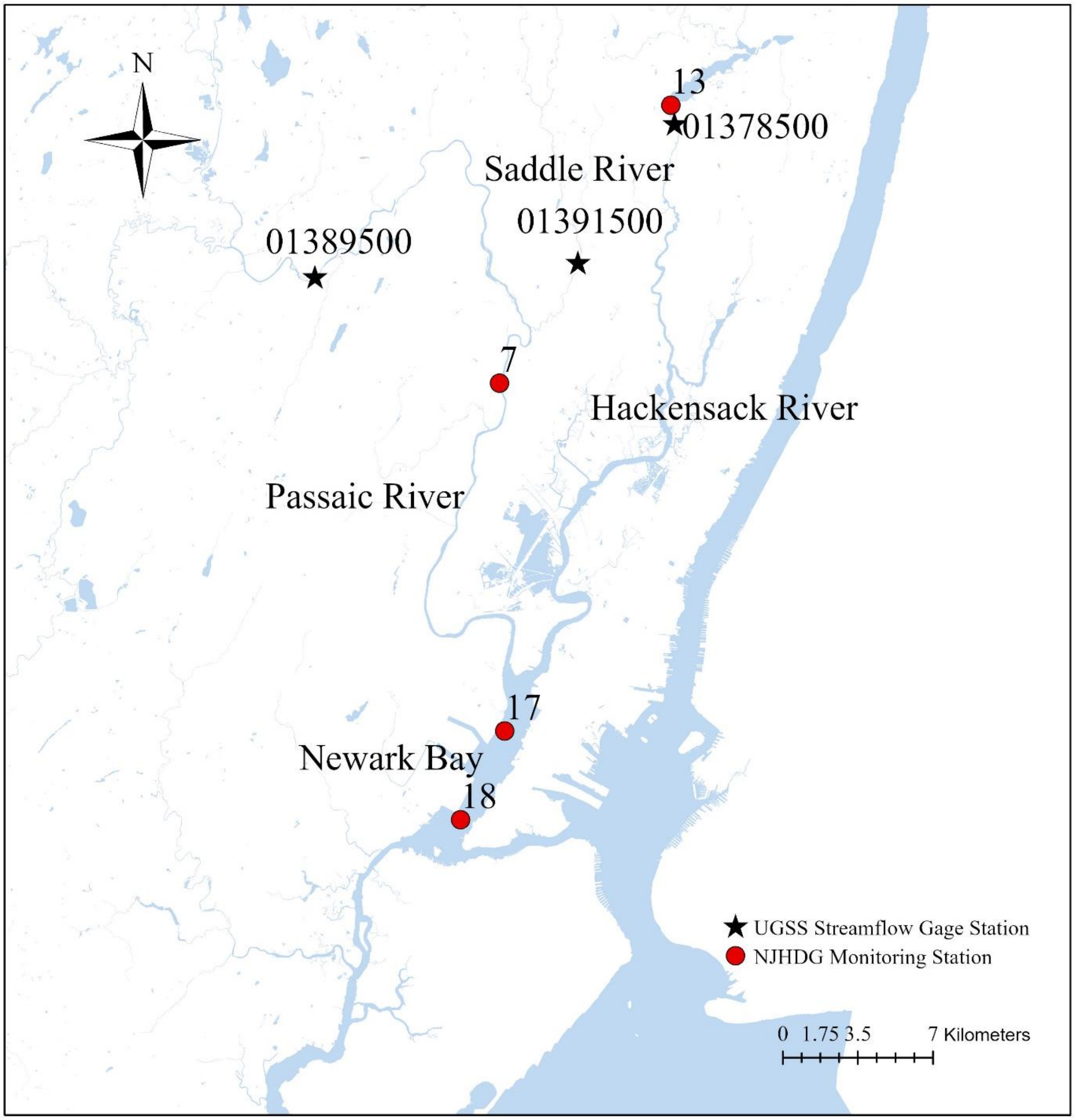
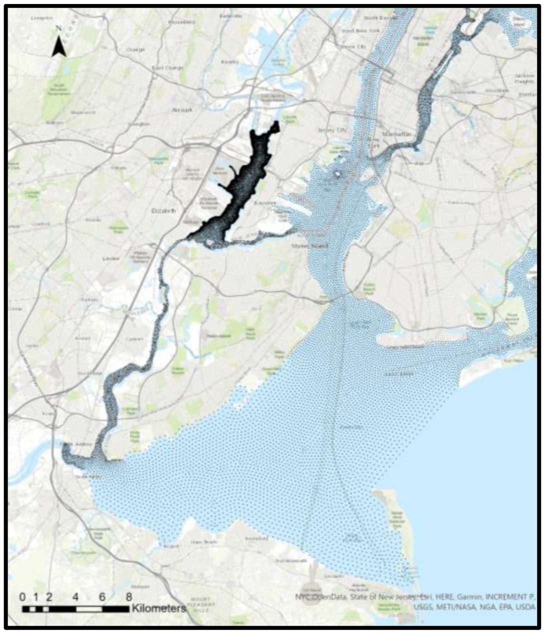
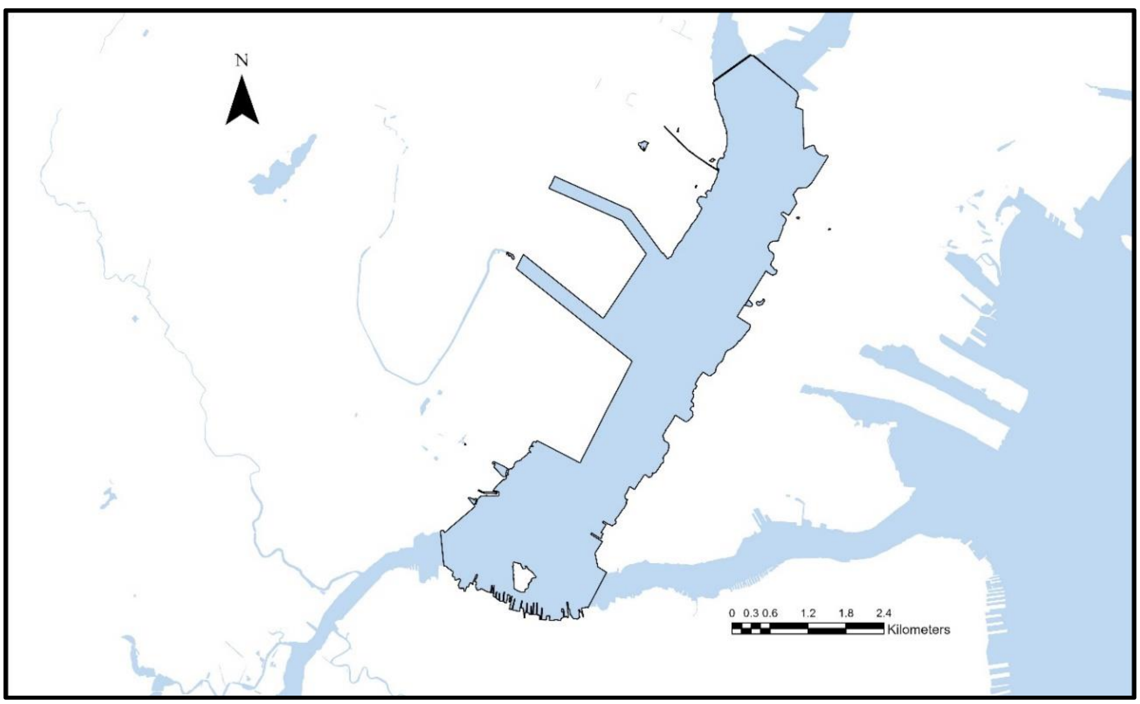


Figure 4-1 Map showing NJHDG monitoring stations and USGS streamflow gaging stations in the Passaic River, the Hackensack River, Saddle River, and Newark Bay. Round dots represent the NJHDG water monitoring stations. Star dots indicated the United States Geological Survey (USGS) gage stations.



a

b



c

Figure 4-2 ArcGIS Pro process with Newark Bay volume calculation. (a) NYNJ harbor mesh (b) Newark Bay mesh (c) Polygon of Newark Bay.

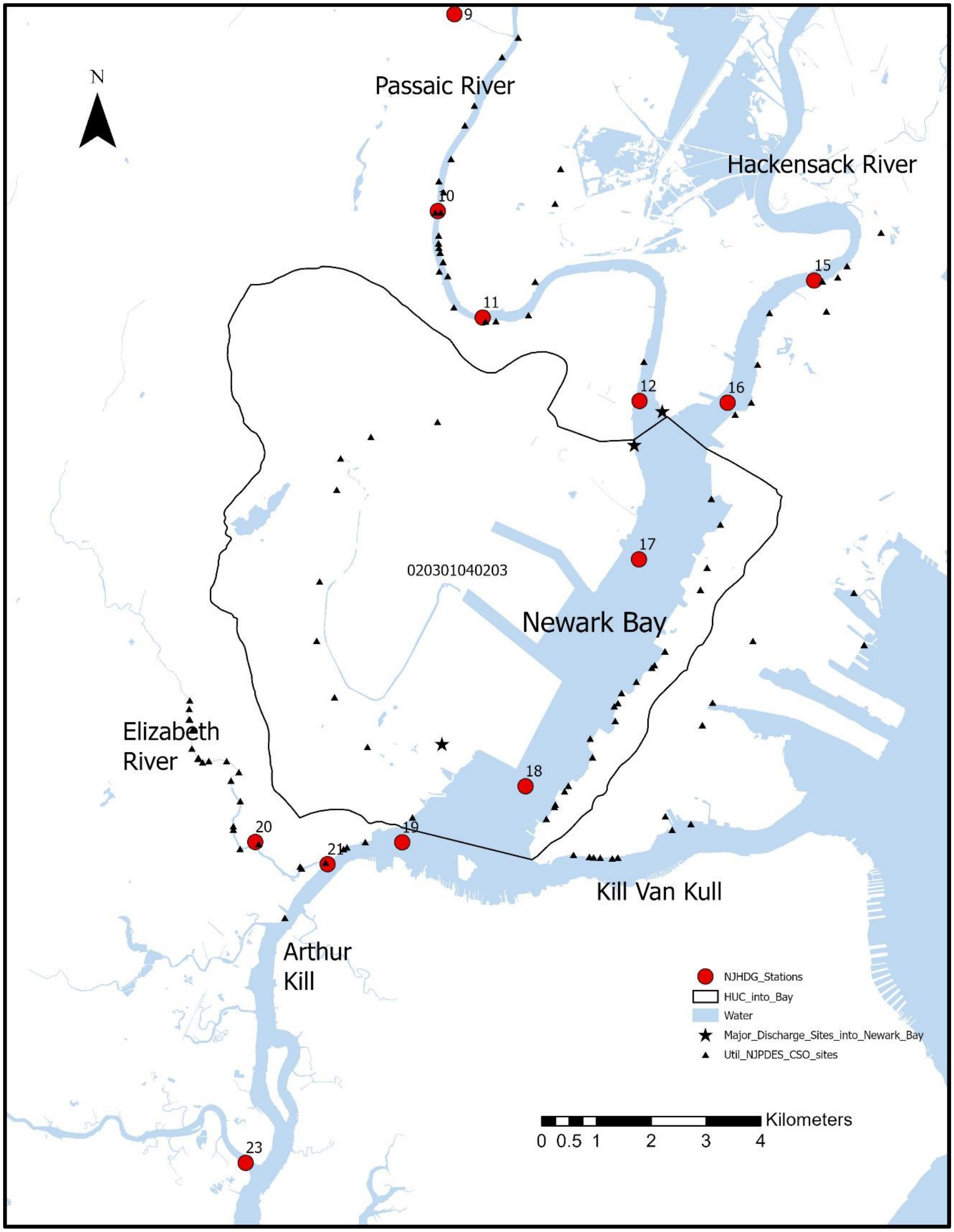


Figure 4-3 Map showing NJPDES CSO sites and other major dicharge sites in the study area.

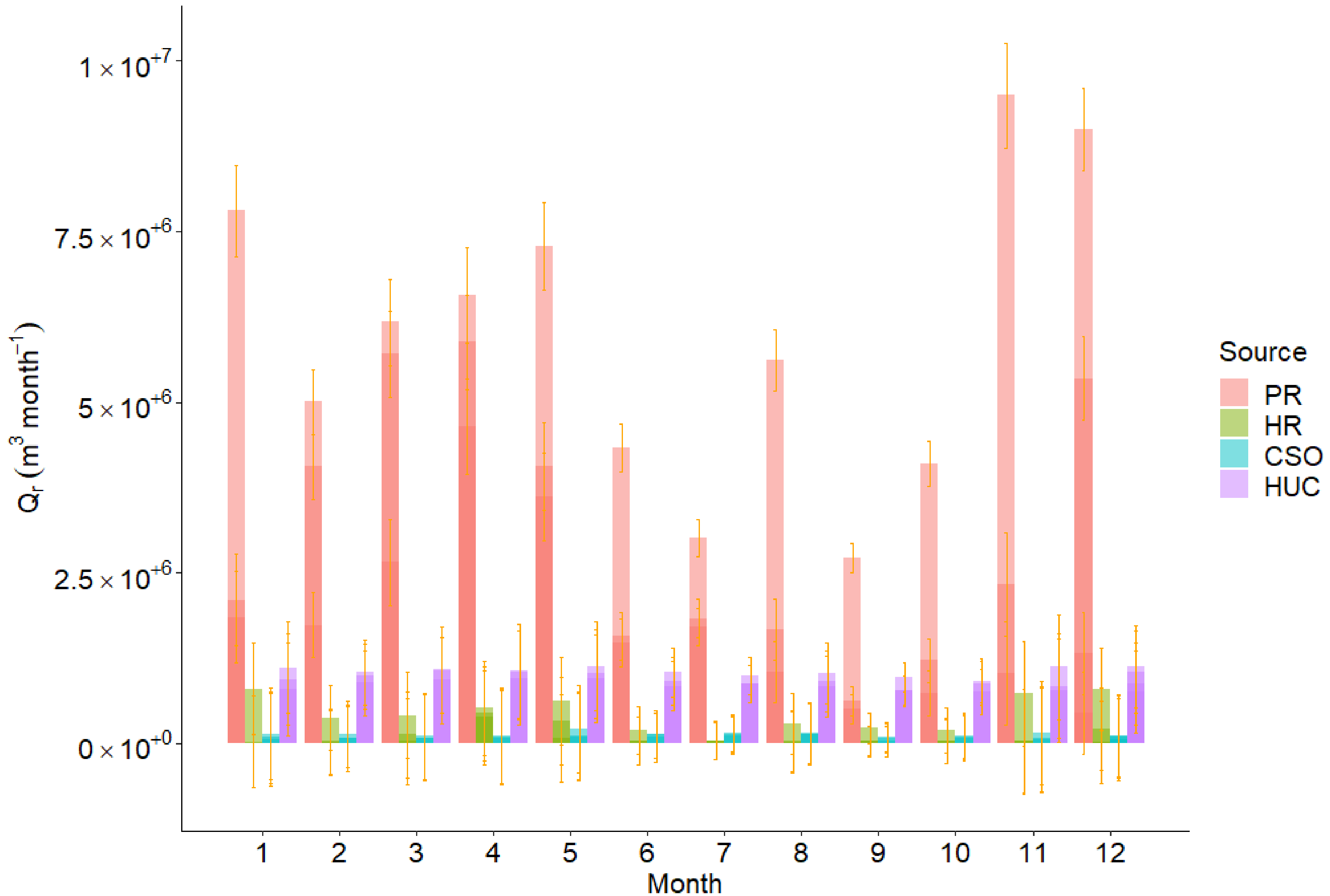


Figure 4-4 Freshwater discharges from PR (Passaic River), HR (Hackensack River), CSO sites, and HUC discharge sites from December, 2016 to December, 2019.

Figure 4-5 Annual-averaged nutrient (TIN and ortho-P) loadings from the Passaic River measured at St 7 and the Hackensack River measured at St 13 (nutrient load in log10 scale) from 2004 to 2019.

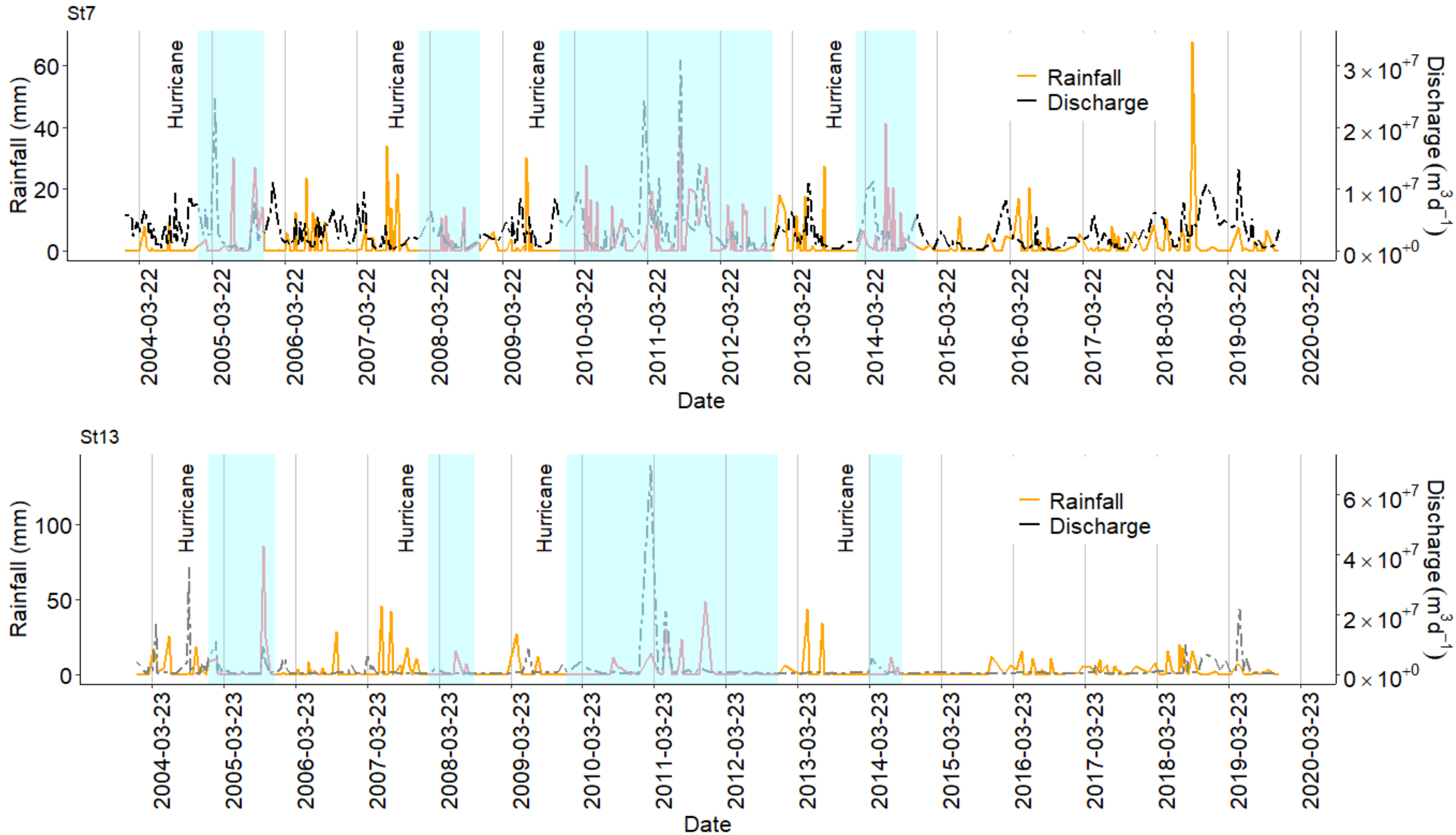


Figure 4-6 Daily discharge and rainfall data at St 7 (Passaic River) and St 13 (Hackensack River) from 2004 to 2019.

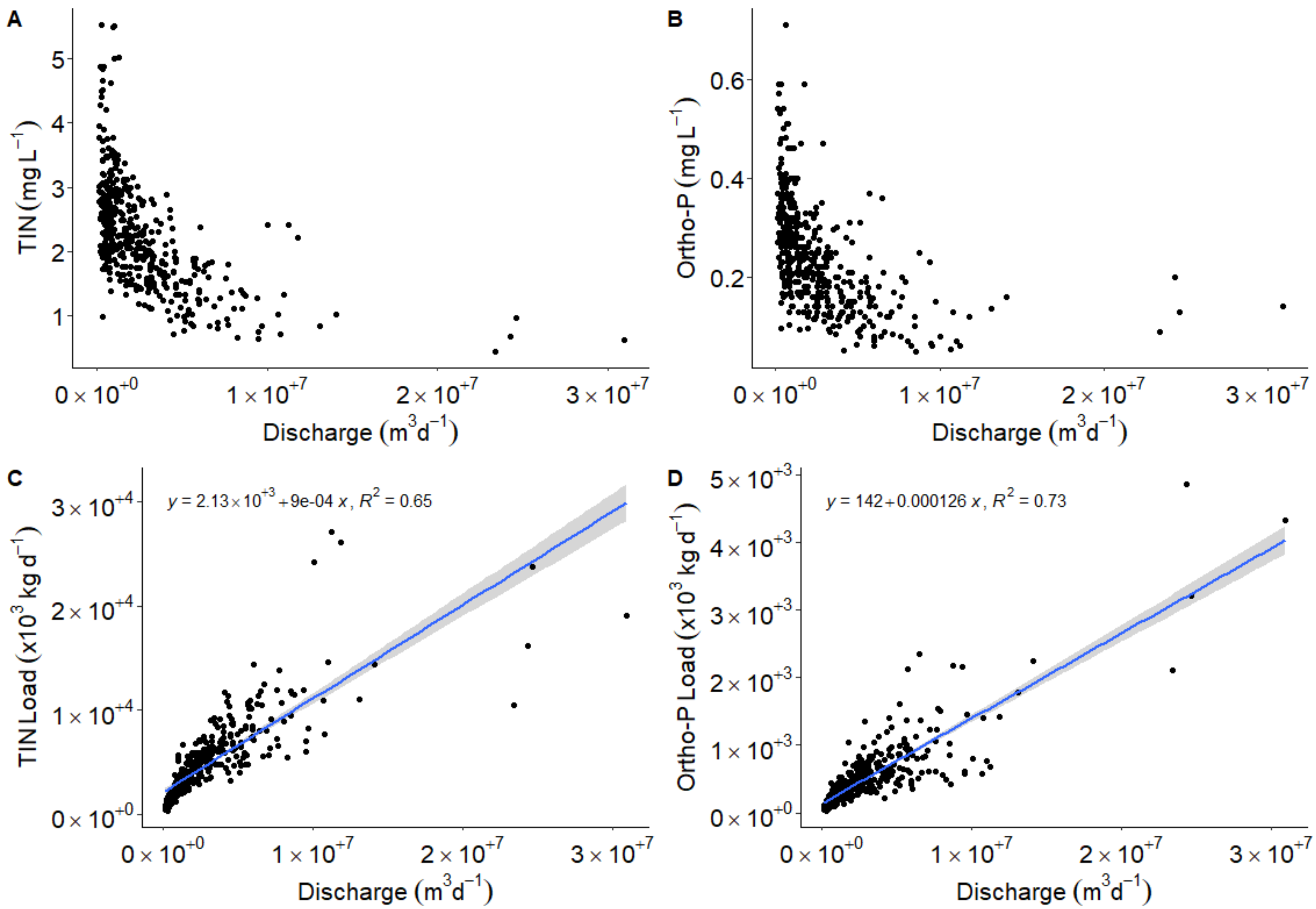


Figure 4-7 Relationship between TIN concentration and discharge (A); Ortho-P concentration and discharge (B); TIN load and discharge (C) and Ortho-P load and discharge (D) in St7 (Passaic River) from 2004 to 2019

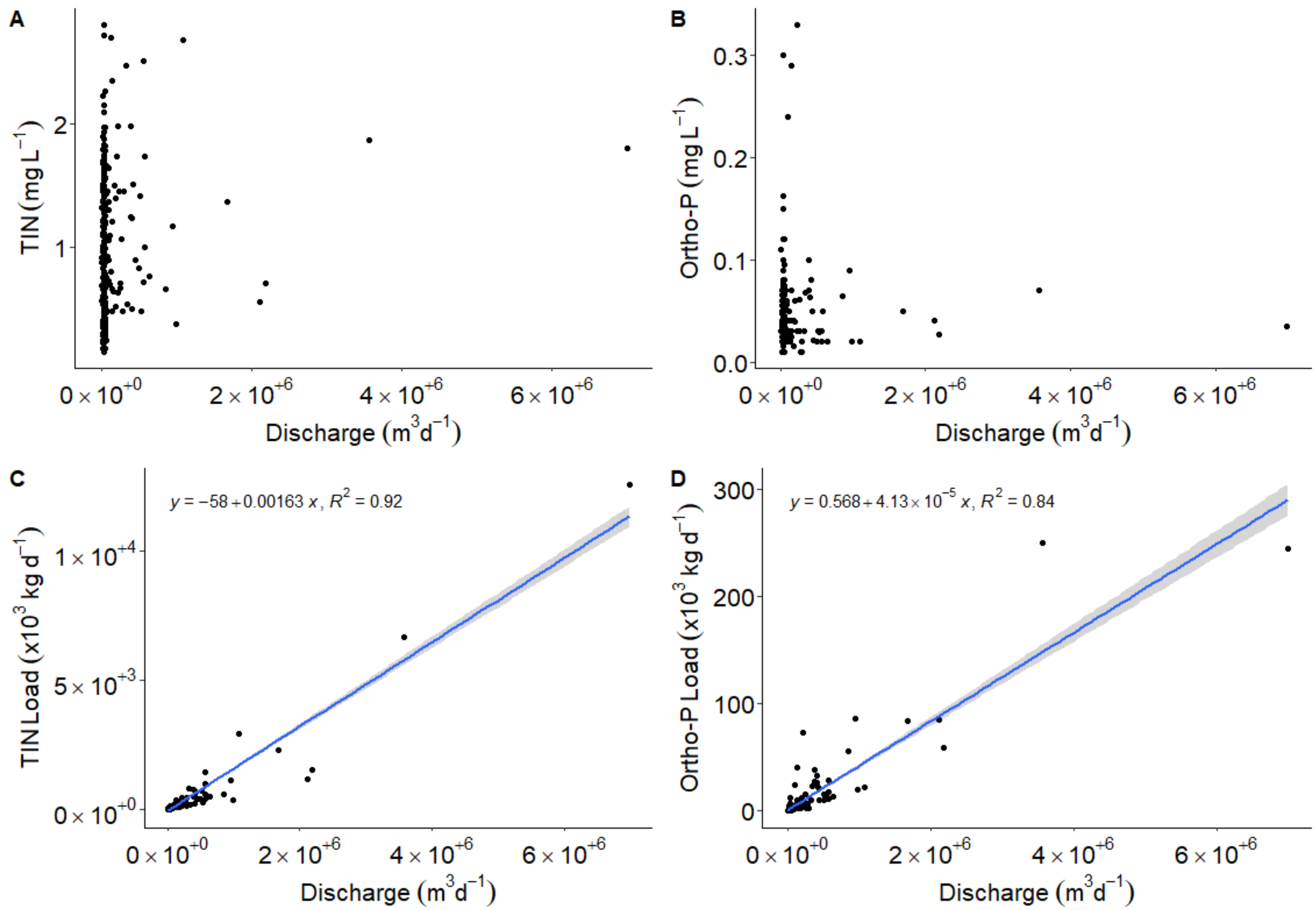


Figure 4-8 Relationship between TIN concentration and discharge (A); Ortho-P concentration and discharge (B); TIN load and discharge (C) and Ortho-P load and discharge (D) in St13 (Hackensack River) from 2004 to 2019

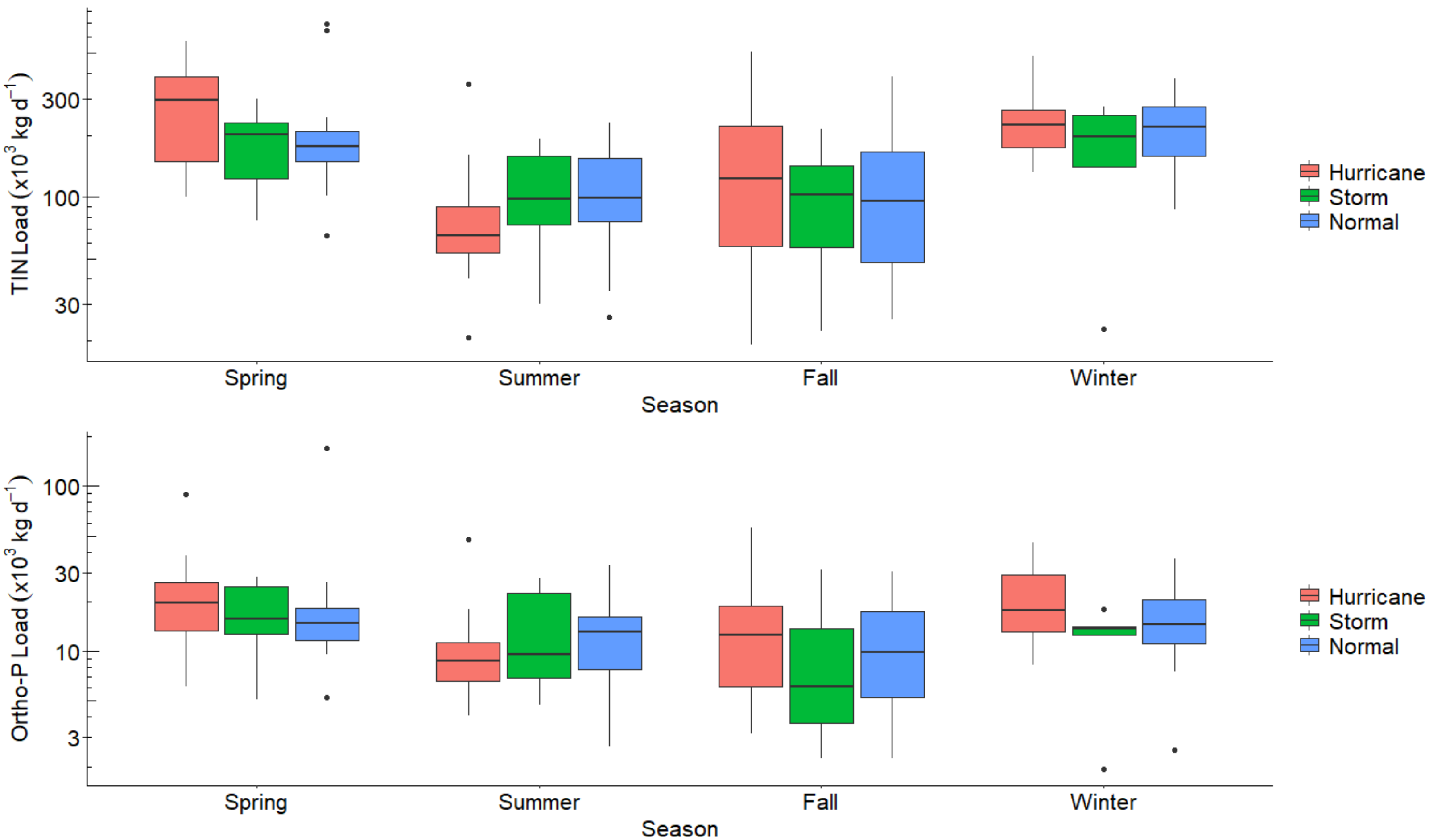


Figure 4-9 Comparison of seasonal variations of TIN and ortho-P loadings (nutrient load in log10 scale) under hurricane, storm and normal weather conditions in St7 from 2004 to 2019. Note: Normal weather are years without any hurricane or storm event and hurricane conditions are hurricane occurrence years.

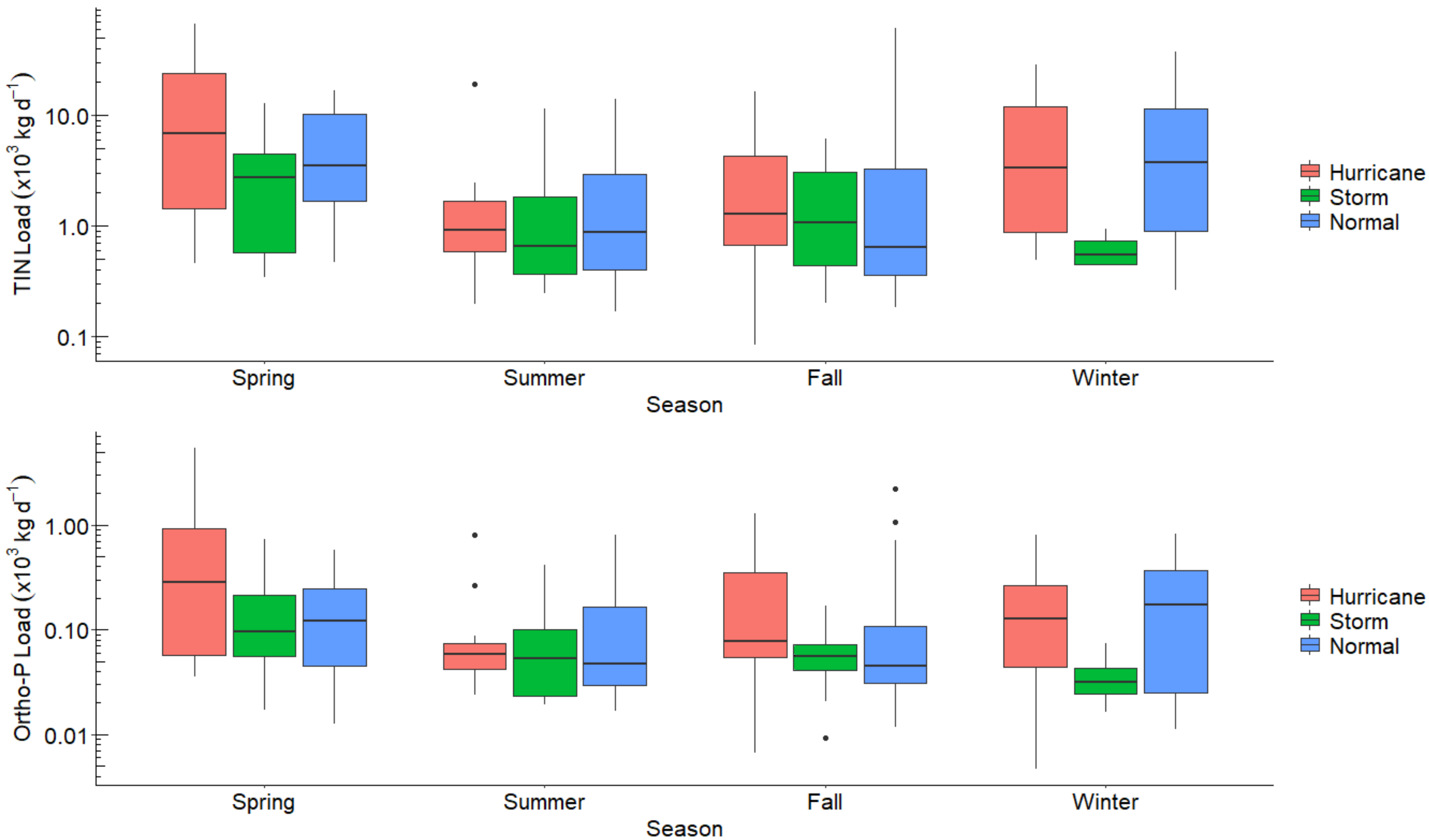


Figure 4-10 Comparison of seasonal variations of TIN and ortho-P loadings (nutrient load in log10 scale) under hurricane, storm and normal weather conditions in St13 from 2004 to 2019. Note: Normal weather are years without any hurricane or storm event and hurricane conditions are hurricane occurrence years.

Figure 4-11 (A) Yearly residence time in Newark Bay from 2011 to 2019. (B) Comparison of seasonal variations of residence time under hurricane, storm and normal weather conditions in Newark Bay from 2011 to 2019. Note: Normal weather are years without any hurricane or storm event and hurricane conditions are hurricane occurrence years.(No correction)

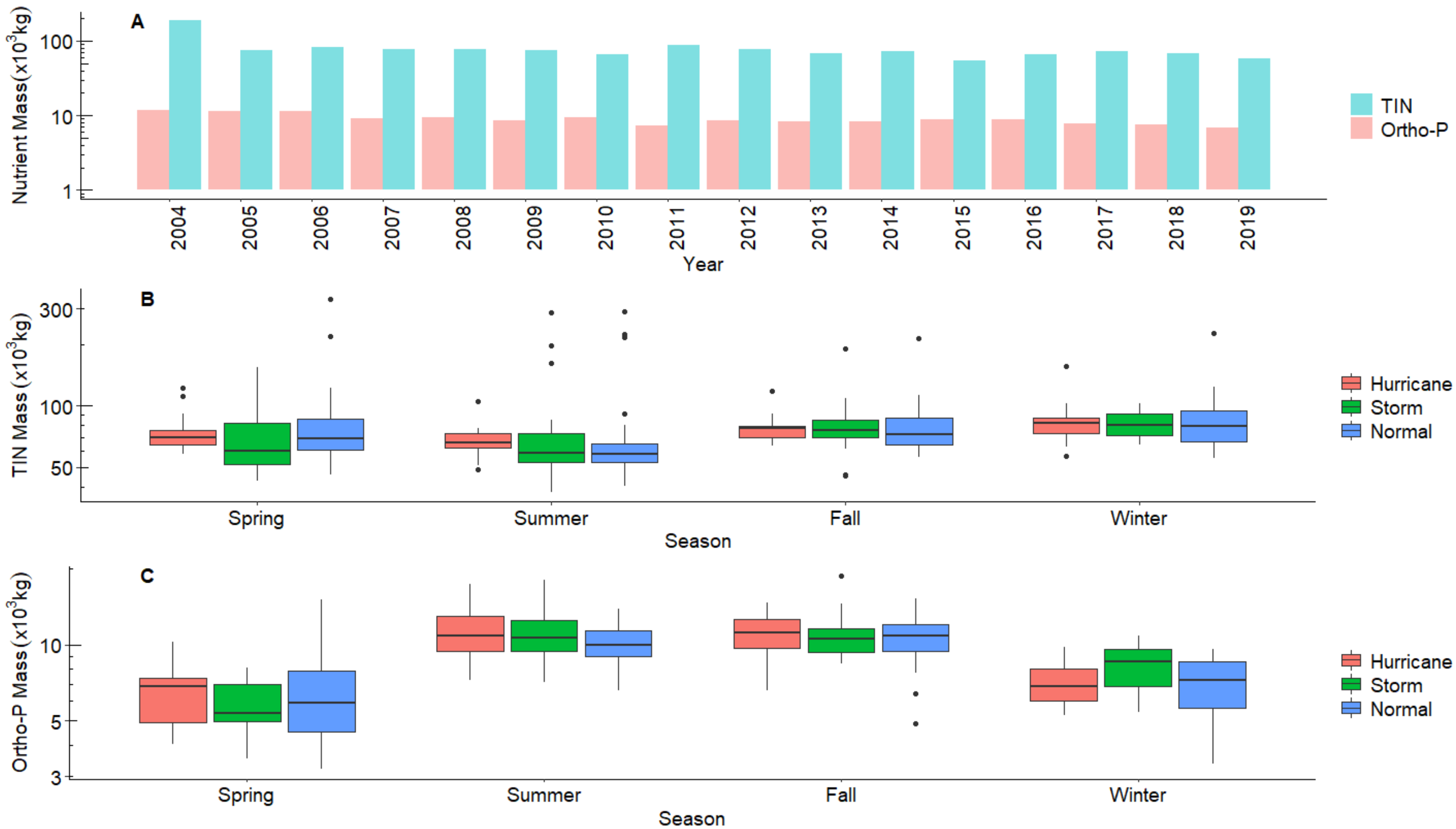


Figure 4-12 (A) Yearly averaged nutrient mass in Newark Bay from 2004 to 2019 (nutrient mass in log10 scale). (B&C) Comparison of seasonal variations of TIN (B) and ortho-P (C) (nutrient mass in log10 scale) under hurricane, storm and normal weather conditions in Newark Bay from 2004 to 2019. Note: Normal weather are years without any hurricane or storm event and hurricane conditions are hurricane occurrence years.

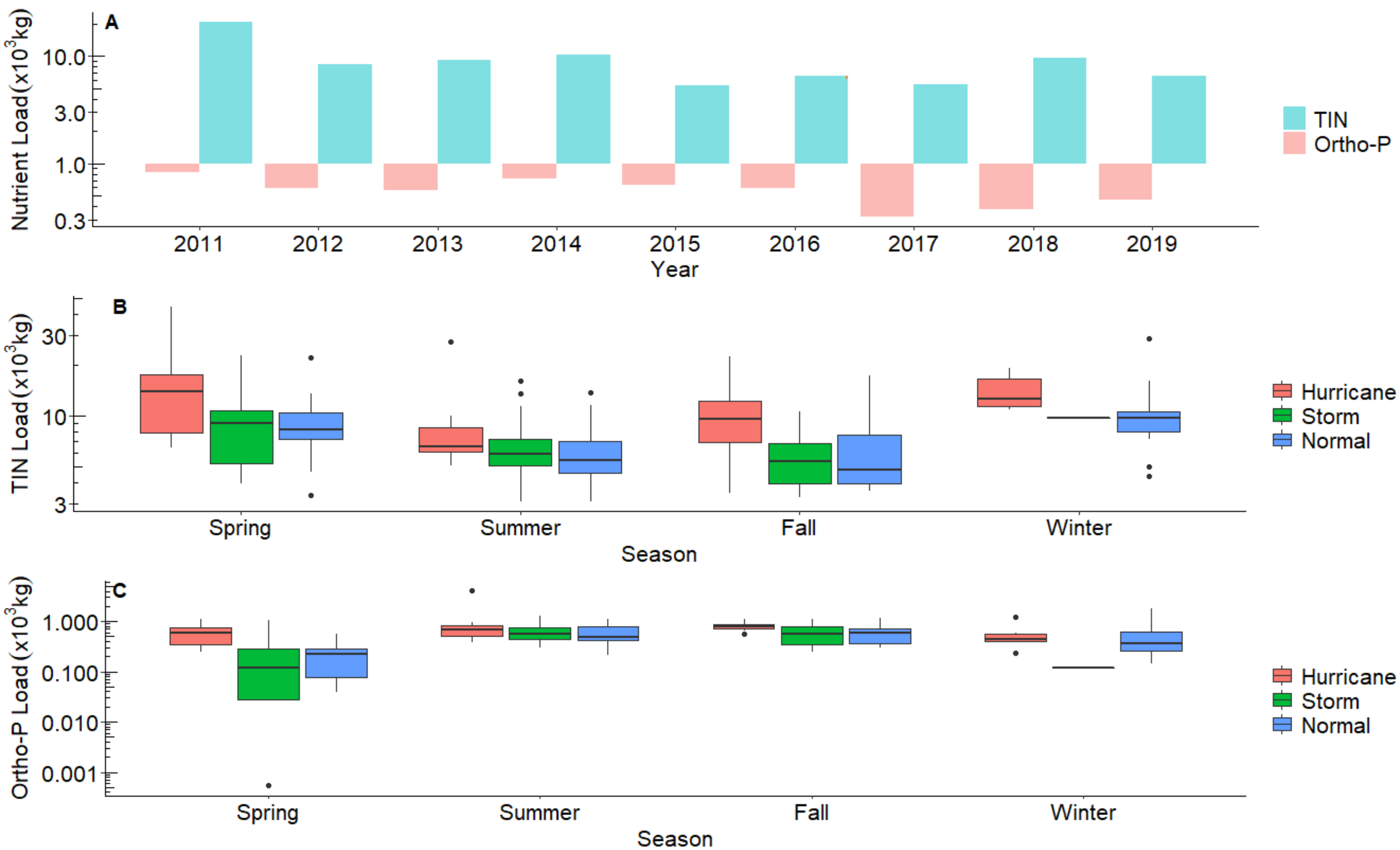


Figure 4-13 (A) Yearly averaged nutrient loadings from other sources to Newark Bay from 2011 to 2019 (nutrient load in log10 scale). (B&C) Comparison of seasonal variations of TIN (B) and ortho-P (C) (nutrient load in log10 scale) under hurricane, storm and normal weather conditions from other sources to Newark Bay from 2011 to 2019. Note: Normal weather are years without any hurricane or storm event and hurricane conditions are hurricane occurrence years.

Tables

Table 4-1 ANOVA and Turkey's test of significance of seasonal variation in TIN loading in different stations (2004-2019).

Nutrient Source	Weather Condition			
	Normal			
	ANOVA		Turkey	Seasonal loading Percentage
Passaic River	<0.01	Spring-Fall	<0.05	Winter (31.2%)>Spring (29.7)>Summer (19.8%)>Fall (19.3%)
		Winter-Fall	<0.05	
		Summer-Spring	<0.05	
		Winter-Summer	<0.05	
Hackensack River	>0.01		>0.05	Winter (32.8%)>Fall (27.7%)>Spring (24.2%)>Summer (15.3%)
Residence Time	<0.01	Spring-Fall	<0.05	Summer (31.3%)>Fall (28.6%)>Winter (21.2%)>Spring (18.9%)
		Winter-Fall	<0.05	
Nutrient Mass	>0.01		>0.05	Spring (25.5%)>Winter (25.3%)>Summer (25.2%)>Fall (24.0%)
Other Source	<0.01	Spring-Fall	<0.05	Summer (31.3%)>Spring (25 3%)>Winter (22.7%)>Fall (20.7%)
		Winter-Fall	<0.05	
		Summer-Spring	<0.05	
		Winter-Summer	<0.05	
	Hurricane			
	ANOVA		Turkey	Seasonal loading Percentage
Passaic River	<0.01	Spring-Fall	<0.05	Spring (39.3%)>Winter (28.6%)>Fall (19.8%)>Summer (12.3%)
		Summer-Spring	<0.05	
		Winter-Summer	<0.05	
Hackensack River	<0.01	Spring-Fall	<0.05	Spring (60.0%)>Winter (19.9%)>Fall (11.7%)>Summer (8.4%)
		Summer-Spring	<0.05	
Residence Time	>0.01		>0.05	Summer (34.1%)>Winter (26 9%)>Fall (22.4%)>Spring (16.6%)
Nutrient Mass	>0.01	Winter-Summer	<0.05	Spring (26.6%)>Fall (26.3%)>Summer (23.8%)>Winter (23.3%)
Other Source	>0.01		>0.05	Spring (37.8%)>Summer (23 3%)>Winter (20.7%)>Fall (18.2%)
	Storm			
	ANOVA		Turkey	Seasonal loading Percentage
Passaic River	>0.01	Summer-Spring	<0.05	Spring (43.6%)>Summer (26.7%)>Fall (15.4%)>Winter (14.3%)
Hackensack River	>0.01		>0.05	Spring (55.1%)>Summer (25.8%)>Fall (16.7%)>Winter (2.4%)
Residence Time	>0.01	Spring-Fall	<0.05	Summer (36.4%)>Fall (36.1%)>Spring (20.2%)>Winter (7.3%)
Nutrient Mass	>0.01		>0.05	Summer (42.5%)>Fall (31.1%)>Spring (19.7%)>Winter (6.7%)
Other Source	<0.01	Spring-Fall	<0.05	Summer (44.5%)>Spring (28 2%)>Fall (25.2%)>Winter (2.1%)
		Summer-Spring	<0.05	
	Difference among Hurricane, normal and Storm			
	ANOVA		Turkey	Loading sequence
Passaic River	>0.01	Storm-Hurricane	<0.05	Hurricane>Normal>Storm
Hackensack River	<0.01	Storm-Hurricane	<0.05	Hurricane>Normal>Storm
Residence Time	<0.01	Normal-Hurricane	<0.05	Normal>Storm>Hurricane
		Storm-Hurricane	<0.05	
Nutrient Mass	>0.01		>0.05	Normal>Storm>Hurricane
Other Source	<0.01	Normal-Hurricane	<0.05	Hurricane>Normal>Storm
		Storm-Hurricane	<0.05	

Table 4-2 ANOVA and Turkey's test of significance of seasonal variation in Ortho-P loading in different stations (2004-2019).

Nutrient Source	Weather Condition			
	Normal			
	ANOVA		Turkey	Seasonal loading Percentage
Passaic River	>0.01		>0.05	Spring (31.5%)>Winter (24.5%)>Summer (24.4%)>Fall (19.6%)
Hackensack River	>0.01		>0.05	Fall (31.8%)>Winter (25.5%)>Summer (23.7%)>Spring (19.0%)
Nutrient Mass	<0.01	Spring-Fall	<0.05	Summer (31.6%)>Fall (30.3%)>Winter (19.9%)>Spring (18.2%)
		Winter-Fall	<0.05	
		Summer-Spring	<0.05	
		Winter-Summer	<0.05	
Other Source	<0.01	Spring-Fall	<0.05	Summer (46.9%)>Fall (30.3%)>Winter (16.5%)>Spring (6.3%)
		Summer-Spring	<0.05	
		Winter-Spring	<0.05	
	Hurricane			
	ANOVA		Turkey	Seasonal loading Percentage
Passaic River	>0.01		>0.05	Spring (33.3%)>Winter (28.3%)>Fall (22.3%)>Summer (16.1%)
Hackensack River	>0.01	Summer-Spring	<0.05	Spring (63.7%)>Fall (16.6%)>Winter (11.3%)>Summer (8.4%)
Nutrient Mass	<0.01	Spring-Fall	<0.05	Summer (33.5%)>Fall (30.5%)>Spring (19.3%)>Winter (16.7%)
		Winter-Fall	<0.05	
		Summer-Spring	<0.05	
		Winter-Summer	<0.05	
Other Source	>0.01		>0.05	Summer (44.7%)>Fall (24.1%)>Spring (17.0%)>Winter (14.2%)
	Storm			
	ANOVA		Turkey	Seasonal loading Percentage
Passaic River	>0.01		>0.05	Spring (41.8%)>Summer (33.9%)>Fall (14.7%)>Winter (9.6%)
Hackensack River	>0.01		>0.05	Spring (55.7%)>Summer (30.2%)>Fall (11.1%)>Winter (3.0%)
Nutrient Mass	<0.01	Spring-Fall	<0.05	Summer (47.4%)>Fall (34.4%)>Spring (12.7%)>Winter (5.5%)
		Summer-Spring	<0.05	
Other Source	<0.01	Spring-Fall	<0.05	Summer (60.2%)>Fall (35.4%)>Spring (4.1%)>Winter (0.3%)
		Summer-Spring	<0.05	
	Difference among Hurricane, normal and Storm			
	ANOVA		Turkey	Loading sequence
Passaic River	>0.01		>0.05	Hurricane>Normal>Storm
Hackensack River	<0.01	Storm-Hurricane	<0.05	Hurricane>Normal>Storm
Nutrient Mass	>0.01		>0.05	Hurricane>Storm>Normal
Other Source	>0.01	Normal-Hurricane	<0.05	Hurricane>Storm>Normal

Chapter 5 Statistical Analysis of Total Suspended Solids Loadings and Potential Relations with Land Use Land Cover Type in New Jersey

Abstract

The study investigates the relationship between total suspended solids (TSS) loadings and Land Use Land Cover (LULC) type across six drainage basin areas in New Jersey. Sixteen years of published monitoring data from the government public sources were used in this study. The results show that the water discharge has a strong correlation with the area of a drainage basin. In this investigation, Pearson's correlation matrix shows positive correlations between TSS concentration and medium and high developed LULC types, while forests and wetlands have a significant negative impact on TSS concentration. Annual and seasonal variations of TSS loading are analyzed to examine the impact of weather conditions on TSS loading. This study indicates that hurricane and storm events have a significant impact on TSS loading, with Hurricane Irene being the most significant event. This study also employed the ARIMA model to forecast future TSS loading trends and fluctuations over time, which showed well-suited for capturing cyclic patterns especially with seasonal variations in time series data.

keyword: Total Suspended Solids, Land Use Land Cover, ARIMA model, Seasonal Variations

1. Introduction

Total suspended solids (TSS) are an important parameter in assessing the water quality of rivers and streams (Varol, 2020). High levels of TSS can negatively impact aquatic life, water clarity, and recreational activities. Additionally, TSS can act as a carrier of pollutants, such as heavy metals and organic compounds, which can further harm the ecosystem and public health (Jeong, Choi, Lee, Lim, & Ra, 2020). Land use and land cover (LULC) changes are a fundamental aspect of human development, driven by various factors such as urbanization, agriculture, and industrialization (Hussain et al., 2020). These changes have significant impacts on the environment, particularly on the quality of water resources (Tsegaye & Technology, 2019). Therefore, total suspended solids (TSS) are one of the key water quality parameters that are influenced by LUCC (Harianja et al., 2019). LUCC can alter the landscape and lead to changes in runoff and erosion, which can increase TSS levels in water bodies (Z. Li, Wang, Song, Wang, & Musakwa, 2021). For example, urbanization can lead to the construction of impervious surfaces such as roads and buildings, which increases the amount of runoff and erosion (Sunardi et al., 2022). In addition, agricultural practices such as tillage and overgrazing can increase soil erosion and nutrient runoff, leading to elevated TSS levels in nearby water bodies (Ly, Metternicht, & Marshall, 2020). Deforestation can also increase TSS levels by increasing soil erosion and sedimentation (Kadir, Ahmed, Uddin, Xie, & Kumar, 2022). Conversely, areas with more natural vegetation and riparian buffers can help to reduce TSS levels by filtering and absorbing pollutants before they reach the water (W. Chen et al., 2021). Wetlands and other natural areas can also help to retain and slow down water, allowing sediment and other particles to settle out (Gedefaw, Geli, & Abera, 2021). In the meantime, LULC change can alter the hydrology of a watershed, which can impact sediment transport and deposition

(Alamdari, Claggett, Sample, Easton, & Yazdi, 2022). Changes in vegetation cover can also impact the infiltration and storage of water, which can influence the timing and volume of water flow (Nkwanda, Feyisa, Zewge, Makwinja, & Resources, 2021). Nutrient cycling can also be affected by LULC, which can impact the growth of algae and other aquatic plants that contribute to TSS levels (Ruan, Kuang, He, Zhen, & Ding, 2020). In this study, drainage basin is used for each river to quantify LULC that may cause TSS level changes. Seasonal variations of TSS fluxes for river streams are also conducted to further study the patterns of TSS fluxes in each river. Better acknowledgement of the relationships between TSS and LULC can help make effective management strategies for LULC and TSS levels include practices such as conservation tillage, riparian buffer restoration, and stormwater management techniques that reduce runoff and erosion.

2. Methodology

2.1 Study Area characterization

There are six rivers involved in this study (Table 5-1): Passaic River, Saddle River, Hackensack River, Elizabeth River, Rahway River and Raritan River. Figure 5-1 shows the study area of North New Jersey. Station (St) 1 represent Passaic River, St6 for Saddle River, St13 for Hackensack River, St20 for Elizabeth River, St22 for Rahway River and St25 for Raritan River. Passaic River is 120 km long (Kenneth R. Olson & Tharp, 2020) and within an oval-shaped river basin with an area of 2,135 km^2 (Oteng Mensah & Alo, 2023). In Figure 5-1(a), the Passaic River is a major waterway in northeastern New Jersey, United States. It is approximately 130 km long and flows through portions of Morris, Somerset, Union, Essex, Passaic, and Bergen counties (Oteng Mensah et al., 2023). The Passaic River has a long history of industrial use, particularly during the 19th and early 20th centuries, when many factories and mills were built along its banks (Ophori et al., 2019). The Saddle River is a tributary of the Passaic River in northeastern New Jersey, United States. It is approximately 40 km long and flows through Bergen County (Winfield, 1923). The Hackensack River is another major river that flows approximately 72 km through Bergen County and Hudson County, and it is one of the most heavily used waterways in the New York City metropolitan area (Reinfelder & Janssen, 2019).The Elizabeth River is a tributary of Newark Bay and is approximately 6.4 km long. The river is navigable by vessels up to 12 m in draft, and it provides access to the Arthur Kill and the New York Harbor(Bozinovic et al., 2021). The Elizabeth River has been impacted by pollution from a variety of sources, including industrial discharges and stormwater runoff due to its special location (Wieczerak et al., 2020). The Rahway River flows for approximately 39 km from its headwaters in Essex County through Union and Middlesex counties before emptying into Arthur Kill, a tidal strait separating Staten Island, New York from mainland New Jersey (Mousa et al., 2022). The Rahway River has historically been prone to flooding, particularly in the lower reaches of the river (Alagrabawi, 2022). The Raritan River flows approximately 137 km from its headwaters in Morris County to its mouth at Raritan Bay in Middlesex County (Y.-n. Li et al., 2020). The river's watershed covers an area of approximately 2845 km^2 and includes parts of several counties, including Morris, Somerset, Hunterdon, Middlesex, and Union (Slattery, 2022). The Raritan River has been impacted by pollution from industrial and agricultural sources. Efforts are underway to clean up the river and restore its ecological health (Y. Wang et al., 2022).

Figure 5-1(a) shows the water quality monitoring stations used by the New Jersey Harbor Discharge Group (NJHDG). Unlike typical river drainage basins, the drainage basins shown in Figure 1(a) were calculated using ArcGIS hydro tools and specific to each water quality monitoring station. Each drainage basin represents a particular area of land where all surface water flows to the water quality monitoring station other than a shared watershed. The black stars in Figure 5-1(a) are discharge sites from the United States Geological Survey (USGS). Figure 5-1(b) shows the Land use land cover (LULC) data for each drainage basin in 2004. The LULC data is based on the specific drainage basin calculated in Figure 5-1(a).

2.2 Data and Method

2.2.1 LULC and Watershed Data

To obtain a drainage basin for a specific station in a river using ArcGIS Pro, a digital elevation model (DEM) for the study area is downloaded from USGS websites and then create a flow direction raster and a flow accumulation raster using the Spatial Analyst toolbox based on DEM. After identifying the location of the station in the river, use the "Watershed" tool to create a polygon representing the drainage basin for the station. The "Stream Order" tool can be used to refine the boundaries of the drainage basin and identify the main channel of the river. The LULC data for the study area in 2004, 2006, 2008, 2011, 2013, and 2016 are downloaded from the National Land Cover Database (NLCD) product provided by USGS. Each year's LULC data is then clipped for each drainage basin of NJHDG stations. Nationwide data on land cover and land cover change is available with a 30m resolution and a 16-class legend, which is based on a modified Anderson Level II classification system (Jin et al., 2019). The land area for each category in LULC is calculated and organized into land use types, station numbers, and years, based on the downloaded data.

Water quality data (TSS concentration) was obtained from NJHDG (2004-2019), and discharge data was from USGS (2004-2019) website. The TSS fluxes were estimated for a certain duration using a method suggested by the Joint Group of Experts on the Scientific Aspect of Marine Pollution (GESAMP) (GESAMP, 1987). The equation used for the calculation is:

$$F = \frac{K \cdot \sum_{i=1}^{n} C_i Q_i}{\sum_{i=1}^{n} Q_i} \overline{Q_r} \quad (1)$$

where K represents the conversion factor, C_i (mg L^{-1}) denotes the instantaneous concentration linked with individual samples, Q_i (m^3 d^{-1}) signifies the instantaneous discharge at the time of sampling, and $\overline{Q_r}$ (m^3 d^{-1}) denotes the mean discharge for the duration under consideration. This method is advised when there is continuous discharge data and non-continuous concentration data available. The precision of the estimated fluxes using this approach depends on the features of the rivers and the representativeness of the flow-weighted mean concentration value that is derived from a relatively small amount of data.

2.2.2 Statistic Analysis of TSS and LULC data

From previous studies (Khoi et al., 2022), TSS concentration has been influenced by LULC. In order to quantify the relationship, Pearson correlation analysis is used in this study to measure the strength and direction of the linear relationship between TSS concentration and LULC data. The correlation coefficient ranges from -1 to 1. The significance of the correlation coefficient is determined by using a significance level (alpha) of 0.05. Considering the impact of storm and hurricane conditions, TSS fluxes are calculated with different weather categories (normal, storm and hurricane). Based on the flux calculation equation above, TSS fluxes are calculated based on the daily-average, monthly average and yearly average data. With the monthly-averaged fluxes data, ANOVA tests are used to determine if there are significant differences in TSS fluxes among the different seasons (e.g., winter, spring, summer, and fall). The Tukey test is a post-hoc test that is commonly used to identify which groups (in this case, seasons) have significantly different means. In addition, Autoregressive Integrated Moving Average (ARIMA) model is a statistical method that captures the underlying patterns and trends in a time series data and makes predictions based on those patterns. The ARIMA method was applied to forecast TSS loadings from six rivers in this study, utilizing available TSS data. The autoregressive terms (p) and moving-average terms (q) were determined by analyzing the plots of partial autocorrelation function (PACF) and autocorrelation function (ACF). The selection of the final ARIMA model was based on the Akaike Information Criterion (AIC). The dataset for this study comprised 187 observations from January 2004 to December 2019, with 127 observations utilized for prediction and the remaining 60 observations used for validation.

3. Results and Discussion

3.1 Characteristics of the LULC

Figure 5-2 illustrates the average (2004-2016) Land Use Land Cover (LULC) compositions and proportions within the drainage area of each water quality monitoring station. Deciduous forest accounts for 44.5% of the drainage basin around St1 (Passaic River), where more than 75% of the tree species respond simultaneously to seasonal changes. St6 (Saddle River) and St13 (Hackensack River) have 44.2% and 46.0% of developed, open space, respectively, where lawn grasses are the predominant vegetation and impervious surface area comprises less than 20% of total cover. St20 (Elizabeth River) has 44.6% of developed, medium intensity, where impervious surface area accounts for 50% to 79% of the total cover. St22 (Rahway River) and St25 (Raritan River) have 31.2% and 27.1% of developed, low intensity, respectively, with single-family housing units being the most common type of area. As shown in Figure 5-3, the LULC data did not show any significant change from 2004 to 2016. The main LULC type of St1 and St13 decreased by only 2.3% and 1.4%, respectively, while the main

LULC type of St6, St20, St22, and St25 increased by 0.72%, 1.52%, 0.4%, and 0.5%, respectively. These results suggest that the main type of LULC in the drainage basin remained relatively stable over the 12-year period, with the percentage of the main type not changing by more than 3%. Therefore, this study neglected the effects of LULC change.

3.2 Relations between LULC and TSS concentration

A Pearson's correlation matrix was used to examine the relationship between land use and land cover (LULC) type and total suspended solids (TSS) concentration, as LULC type greatly impacts TSS concentration. Figure 4 displays the results of the correlation matrix, with a significance level of $(p)<0.05$. The findings reveal that there is a positive correlation between TSS concentration and the spatial difference (i.e., station number), as well as the level of development (low intensity (LowI), medium intensity (MI), and high intensity (HI)). This suggests that as the station number increases from St1 to St25, both TSS concentration and the percentage of developed LULC also increase.

As shown in Table 5-1, St1, St6, and St13 are located in the upper level of New Jersey (Figure 5-1(A)), whereas St20, St22, and St25 are located in the lower level. The lower level of New Jersey has a more developed LULC area than the upper level. MI and HI LULC types have a strong positive correlation with TSS concentration, as these areas contain between 50% to 100% impervious surface area of the total cover (Jin et al., 2019). With more impervious surfaces, there will be more runoff carrying more TSS into the water body (Rio, Salles, Cernesson, Marchand, & Tournoud, 2020). Evergreen forest (EForest) and woody wetlands (WWetland) have a significant negative impact on TSS concentration. These areas have more than 20% vegetation cover, and the soil is saturated with water all year long. With high vegetation cover, TSS is challenging to flush into the water body (W. Chen et al., 2021). There are also negative correlations between TSS concentration and developed open space (Open Space), mixed forest (MForest), barren land (Barren), cultivated crop (Crop), water, shrub/scrub (Shrub), deciduous forest (DForest), emergent herbaceous wetlands (Ewetland), grassland/herbaceous (Grass), and pasture/hay (Pasture) (Nkwanda et al., 2021). MForest, Crop, Water, Shrub, DForest, Ewetland, Grass, and Pasture have over 20% vegetation cover, resulting in lower TSS concentration. Open Space has less than 20% impervious surface cover and consists mostly of lawn grasses, which also helps to reduce TSS concentration. However, Barren land, which is composed of bedrock, sand, and clay and has less than 15% vegetation cover, still exhibits a negative correlation with TSS concentration due to the limited capacity for soil organic matter to contribute to TSS concentration in the presence of high amounts of rocks, sand, and clay (Kadir et al., 2022).

3.3 Annual and seasonal variations in TSS loading in six rivers.

3.3.1 Annual variation of TSS loading in six rivers.

Figure 3-6 shows that the Passaic River, as measured at St1, had annual TSS loadings ranging from 3395×10^3 kg y^{-1} to 59581×10^3 kg y^{-1}. However, there was an overall decreasing trend (52%) in TSS loadings from 2004 to 2019, with a peak in 2011. The Saddle River had TSS loadings ranging from 358×10^3 kg y^{-1} to 4686×10^3 kg y^{-1}, with a slight overall increase of 9% in total loading. This river also had two peaks in TSS loading in 2011 and 2013. The Hackensack River had TSS loadings ranging from 91×10^3 kg y^{-1} to 1447×10^3 kg y^{-1}, with an overall decrease of 61%, though there was still a peak in 2011. The Elizabeth River had TSS loadings ranging from 277×10^3 kg y^{-1} to 3204×10^3 kg y^{-1}, with an overall increase of 41%, and a peak in 2009. Finally, the Rahway River had TSS loadings ranging from 813×10^3 kg y^{-1} to 4659×10^3 kg y^{-1}, with an overall increase of 260%, while the Raritan River had TSS loadings ranging from 515×10^3 kg y^{-1} to 3595×10^3 kg y^{-1}, with an overall increase of 146%.

The Raritan River had the highest TSS loading in 2007, 2018, and 2019, while the Passaic River contributed the highest TSS loading among the six rivers analyzed. The Hackensack River had the lowest TSS loading. The Rahway and Raritan Rivers experienced a significant increase in TSS loading in 2019. In 2011, the Passaic, Saddle, and Hackensack Rivers had their highest peak in TSS loading. Historical records indicate that New Jersey experienced six hurricanes, including Hurricanes Maria and Nate in 2005 and Hurricane Irene in 2011 (Gump et al., 2017), which impacted the water quality in the Passaic and Hackensack Rivers. Hurricane Irene, a Category 3 hurricane, was the first landfall hurricane in New Jersey since 1903, resulting in high discharge in both rivers (Saleh et al., 2017). The Elizabeth River, located along Newark Bay, was also impacted by the flooding event. The peaks in TSS loading in 2018 and 2019 for the Raritan and Rahway Rivers, respectively, can be attributed to high rainfall and discharge (Figure 5-7) during those years. To better understand the exceptional peaks in TSS loading, storm and hurricane events were considered during the seasonal variation analysis.

3.3.2 Seasonal variation of TSS loading in six rivers

Seasonal variation of TSS loading under different weather conditions in six rivers from 2004 to 2019 is shown in Figure 5-8. ANOVA tests were conducted for each river to determine whether there is a seasonal variation among TSS loading. Subsequently, the Turkey test was carried out to identify which season has the distinct TSS loading mean. The results of ANOVA and the Turkey test are presented in Table 5-2. Among the six rivers under hurricane conditions, there is no significant difference in TSS loading in the four seasons. However, for Passaic River, Saddle River, and Hackensack River, the total TSS loading is higher in spring. Elizabeth River and Rahway River have higher TSS loading in summer, and Raritan River has higher loading in fall. Under normal conditions, there is seasonal variation in TSS loading for Saddle River, Rahway River, and Raritan River. Both Saddle River and Rahway River have higher total (and mean) TSS loading in summer, while Raritan River has higher loading in spring. Under storm conditions, Rahway River displays seasonal variation and results in higher TSS loading in summer.

Based on the ANOVA and Turkey tests conducted in three different weather conditions, it was observed that Saddle River, Hackensack River, and Rahway River were significantly affected by weather conditions. Storm events had a significant impact on both Saddle River and Rahway River, while hurricane events highly impacted Hackensack River. In summer, Rahway River had a higher TSS loading than the other two rivers. The percentage of summer TSS loading was increased from 45.1% to 62.5% due to the impact of storm events. Under normal conditions, Saddle River had the highest TSS loading in the summer, but with storm and hurricane events, the loading in spring increased from 25.4% to 45.4%. Raritan River had a total TSS loading of 9.3% in fall under normal conditions, which increased to 31.3% under hurricane conditions, mainly due to Hurricane Irene, which hit in late August (Gong, Wang, & Joseph, 2022). Hackensack River showed a significant response to storm and hurricane events, changing its highest TSS loading season from fall to spring. Elizabeth River and Passaic River did not exhibit any significant differences in TSS loading seasons and showed no response to storm or hurricane events. Therefore, among the six rivers, the Rahway River and Saddle River showed significant seasonal differences and were highly impacted by storm events.

3.4 Time series analysis and prediction of TSS loading.

An autoregressive integrated moving average (ARIMA) model was developed to predict the nutrient loadings to Newark Bay based on 16 years (2004-2019) of TSS fluxes (loading) data in six rivers. The augmented Dickey-Fuller Test on the monthly nutrient loading data did not detect any non-stationarity data in each data set ($p<0.05$), which suggests that the data was suitable for ARIMA model prediction. ARIMA models were established for TSS loadings from the Passaic River based on the AIC values of the models (Table 5-3). Specifically, ARIMA (1,0,0) model was selected as the final model for TSS loading, which is as follows:

$$(1-\phi_1 B)(X_t-\mu)=w_t \qquad (2)$$

where B is the backshift operator, and ϕ_1 the coefficient of the AR (1) term (Table 5-3). Figure 9(a) and (b) shows that there is no seasonal term in the model as the lag spikes in the ACF and PACF. Figure 9(c) shows the prediction of the Passaic River monthly TSS loading in the next 6.5 years (or 78 months from January 2020 to June 2027).

The utilization of the ARIMA model for predicting TSS loading from the Saddle River has yielded unsatisfactory results, rendering it unsuitable for accurate prediction at this site (as depicted in Figure 5-10c). The model's output of ARIMA (0,0,0) signifies its lack of utility for reliable predictions. Therefore, alternative modeling approaches or data sources should be explored to facilitate accurate TSS loading prediction at this location. In the meantime, ARIMA $(1,0,0) \times (2,0,0)_{12}$ model was selected as the final model for TSS loading from Hackensack River, which is shown below:

$$(1-\Phi_1 B^S-\Phi_2 B^{2S})(1-\phi_1 B)(X_t-\mu)=w_t \qquad (3)$$

where B is the backshift operator, B^S is the seasonal backshift operator, ϕ_1 and ϕ_2 are the coefficients of AR (1) and AR (2) terms, Φ_1 is the coefficient of the seasonal AR (1) term

(Table 5-3). Figure 5-11(b) shows seasonal lags indicating the seasonal patterns in Hackensack River. The prediction of monthly OP loading for the next 6.5 years (or 78 months from January 2020 to June 2027) is shown in Figure 5-11(c).

Both TSS loading from Elizabeth and Raritan River show no seasonal term in ARIMA model. For Elizabeth River, ARIMA (0,0,1) is the best model, which is below:

$$X_t - \mu = (1 + \Theta B) w_t \tag{4}$$

For Raritan River, ARIMA (1,0,0) is the best model, equation shows below:

$$(1 - \phi_1 B)(X_t - \mu) = w_t \tag{5}$$

where B is the backshift operator, Θ is the coefficient of MA term, ϕ_1 is the coefficient of AR terms (Table 5-3). Figure 5-12 (a) and (b), Figure 5-14 (a) and (b) show no seasonal pattern in ACF and PACF plots. Figure 5-12(c) and Figure 5-14(c) shows the prediction of the TSS loading from Elizabeth and Raritan River.

As for Rahway River, ARIMA (0,0,1) × $(1,0,2)_{12}$ are selected as best model. With seasonal terms, Figure 5-13(a) and (b) show seasonal lag patterns. Equation for model shows below:

$$(1 - \Phi_1 B^S)(X_t - \mu) = (1 + \Theta_1 B^S + \Theta_2 B^{2S})(1 + \Theta B) w_t \tag{6}$$

where B is the backshift operator, B^S is the seasonal backshift operator, Θ is the coefficient of MA term, ϕ_1 is the coefficient of AR term, Θ_1 and Θ_2 are the coefficients of seasonal MA terms (Table 5-3). Figure 13(c) shows the prediction of TSS loading for Rahway River. The results obtained from the ARIMA model indicate that there is similarity in the seasonal variation test of TSS loading across all the rivers. Furthermore, the ARIMA model demonstrates a superior fitting ability when it comes to capturing seasonal changes. In other words, the ARIMA model yields comparable outcomes in terms of seasonal variation testing of TSS loading for all the rivers, while also exhibiting better performance in modeling seasonal changes.

4. Conclusion

This study explores the relationship between TSS loadings and LULC type among six different drainage basin areas. In addition, annual and seasonal variation of TSS loadings is conducted to study the pattern of seasonal change and impact of storm and hurricane events. Pearson's correlation matrix is used to investigate the relationship between LULC and TSS concentration. The study finds that there is a positive correlation between TSS concentration and the spatial difference (station number) and the level of development (low, medium, and high intensity). MI and HI LULC types have a strong positive correlation with TSS concentration, while EForest and WWetland have a significant negative impact on TSS concentration. Additionally, the study reports negative correlations between TSS concentration and several other LULC types, including Open Space, MForest, Barren, Crop, Water, Shrub, DForest, Ewetland, Grass, and Pasture.

Annual and seasonal variation of TSS loading in six rivers in New Jersey under different weather conditions are conducted. The Raritan River had the highest TSS loading in 2007, 2018, and 2019, while the Passaic River contributed the highest TSS loading among the six rivers analyzed. The study also examines the impact of hurricanes and storms on TSS loading, with Hurricane Irene being the most significant event affecting the rivers' water quality. The study finds that Saddle River, Hackensack River, and Rahway River were significantly affected by weather conditions, and storm events had a significant impact on Saddle River and Rahway River, while hurricanes highly impacted Hackensack River. The Rahway River and Saddle River showed significant seasonal differences and were highly impacted by storm events, while Elizabeth River and Passaic River did not exhibit any significant differences in TSS loading seasons and showed no response to storm or hurricane events.

The model results indicate that the ARIMA model is well-suited for capturing the cyclic patterns that occur within the time series data, allowing for more accurate and reliable forecasting of future trends and fluctuations over time. Furthermore, the incorporation of seasonal variations into the ARIMA model helps to account for the effects of seasonal factors such as holidays, weather patterns, or other periodic events that may impact the behavior of the data over time. As a result, the ARIMA model is a powerful tool for analyzing and forecasting time series data with seasonal components and can provide valuable insights into patterns and trends that might otherwise be difficult to detect using other modeling techniques.

Reference

Alagrabawi Y. The Effects and Costs of Changes in River Flooding in New Jersey. 2022.

Alamdari N, Claggett P, Sample DJ, Easton ZM, Yazdi MNJJoCP. Evaluating the joint effects of climate and land use change on runoff and pollutant loading in a rapidly developing watershed. 2022; 330: 129953.

Bozinovic G, Shea D, Feng Z, Hinton D, Sit T, Oleksiak MFJPo. PAH-pollution effects on sensitive and resistant embryos: Integrating structure and function with gene expression. 2021; 16: e0249432.

Chen W, Wang J, Cao X, Ran H, Teng D, Chen J, et al. Possibility of using multiscale normalized difference vegetation index data for the assessment of total suspended solids (TSS) concentrations in surface water: A specific case of scale issues in remote sensing. 2021; 194: 110636.

Gedefaw MG, Geli HM, Abera TAJRS. Assessment of rangeland degradation in New Mexico using time series segmentation and residual trend analysis (tss-restrend). 2021; 13: 1618.

GESAMP. Land/Sea Boundary Flux of Contaminants: Contributions from Rivers. Report and Studies No. 32. GESAMP - IMP/FAO/UNESCO/WMO/WHO/IAEA/UN/UNEP Joint Group of Experts on the Scientific Aspects of Marine Pollution, New York, 1987, pp. 172.

Gong J, Wang Y, Joseph H. Assessing and mitigating transportation infrastructure vulnerability to coastal storm events with the convergence of advanced spatial analysis, infrastructure modeling, and storm surge simulations. Rutgers University. Center for Advanced Infrastructure and Transportation, 2022.

Gump D, Klemm A, van Westendorp C, Wood D, Doroba J. Hurricane Maria, 2017 Hydrographic Response for Puerto Rico and US Virgin Islands by the NOAA Ship Thomas Jefferson. AGU Fall Meeting Abstracts. 2017, 2017, pp. NH23E-2857.

Harianja AH, Suoth AE, Nazir E, Saragih GS, Fauzi R, Hidayat MY. Impact of land use and socio-economic changes in water catchment area on total suspended solid (TSS) in Lake Toba. IOP Conference Series: Earth and Environmental Science. 407. IOP Publishing, 2019, pp. 012005.

Hussain S, Mubeen M, Ahmad A, Akram W, Hammad HM, Ali M, et al. Using GIS tools to detect the land use/land cover changes during forty years in Lodhran District of Pakistan. 2020; 27: 39676-39692.

Jeong H, Choi JY, Lee J, Lim J, Ra KJEP. Heavy metal pollution by road-deposited sediments and its contribution to total suspended solids in rainfall runoff from intensive industrial areas. 2020; 265: 115028.

Jin S, Homer C, Yang L, Danielson P, Dewitz J, Li C, et al. Overall methodology design for the United States national land cover database 2016 products. 2019; 11: 2971.

Kadir A, Ahmed Z, Uddin M, Xie Z, Kumar PJW. Integrated Approach to Quantify the Impact of Land Use and Land Cover Changes on Water Quality of Surma River, Sylhet, Bangladesh. 2022; 14: 17.

Khoi DN, Loi PT, Trang NTT, Vuong ND, Fang S, Nhi PTTJES, et al. The effects of climate variability and land-use change on streamflow and nutrient loadings in the Sesan, Sekong, and Srepok (3S) River Basin of the Lower Mekong Basin. 2022; 29: 7117-7126.

Li Y-n, Li J, Shao Z, Duan Z, Xie Y, Cui Z, et al. Quantitative analysis of 209 polychlorinated biphenyl (PCB) congeners in sediments of the Raritan River estuary, USA. 2020; 20: 2400-2414.

Li Z, Wang S, Song S, Wang Y, Musakwa WJJoAE. Detecting land degradation in Southern Africa using time series segment and residual trend (TSS-RESTREND). 2021; 184: 104314.

Ly K, Metternicht G, Marshall L. Linking Changes in Land Cover and Land Use of the Lower Mekong Basin to Instream Nitrate and Total Suspended Solids Variations. Sustainability 2020; 12.

Mousa AA, Hussein M, Kineber AFJTRR. Value-Engineering Methodology for the Selection of an Optimal Bridge System. 2022; 2676: 483-498.

Nkwanda I, Feyisa G, Zewge F, Makwinja RJIJoE, Resources W. Impact of land-use/land-cover dynamics on water quality in the Upper Lilongwe River basin, Malawi. 2021; 5: 193-204.

Olson KR, Tharp M. How did the Passaic River, a Superfund site near Newark, New Jersey, become an Agent Orange dioxin TCDD hotspot? Journal of Soil and Water Conservation 2020; 75: 33A-37A.

Ophori D, Firor C, Soriano PJEES. Impact of road deicing salts on the Upper Passaic River Basin, New Jersey: a geochemical analysis of the major ions in groundwater. 2019; 78: 1-13.

Oteng Mensah F, Alo CA. Modeling monthly actual evapotranspiration: an application of geographically weighted regression technique in the Passaic River Basin. Journal of Water and Climate Change 2023; 14: 17-37.

Oteng Mensah F, Alo CAJJoW, Change C. Modeling monthly actual evapotranspiration: an application of geographically weighted regression technique in the Passaic River Basin. 2023; 14: 17-37.

Reinfelder JR, Janssen SEJJohm. Tracking legacy mercury in the Hackensack River estuary using mercury stable isotopes. 2019; 375: 121-129.

Rio M, Salles C, Cernesson F, Marchand P, Tournoud M-GJJoH. An original urban land cover representation and its effects on rain event-based runoff and TSS modelling. 2020; 586: 124865.

Ruan Z, Kuang Y, He Y, Zhen W, Ding SJRS. Detecting Vegetation Change in the Pearl River Delta Region Based on Time Series Segmentation and Residual Trend Analysis (TSS-RESTREND) and MODIS NDVI. 2020; 12: 4049.

Saleh F, Ramaswamy V, Wang Y, Georgas N, Blumberg A, Pullen J. A multi-scale ensemble-based framework for forecasting compound coastal-riverine flooding: The Hackensack-Passaic watershed and Newark Bay. Advances in Water Resources 2017; 110: 371-386.

Slattery K. The Raritan River. 2022.

Sunardi S, Nursamsi I, Dede M, Paramitha A, Arief MCW, Ariyani M, et al. Assessing the influence of land-use changes on water quality using remote sensing and GIS: A Study in Cirata Reservoir, Indonesia. 2022; 7: 106-114.

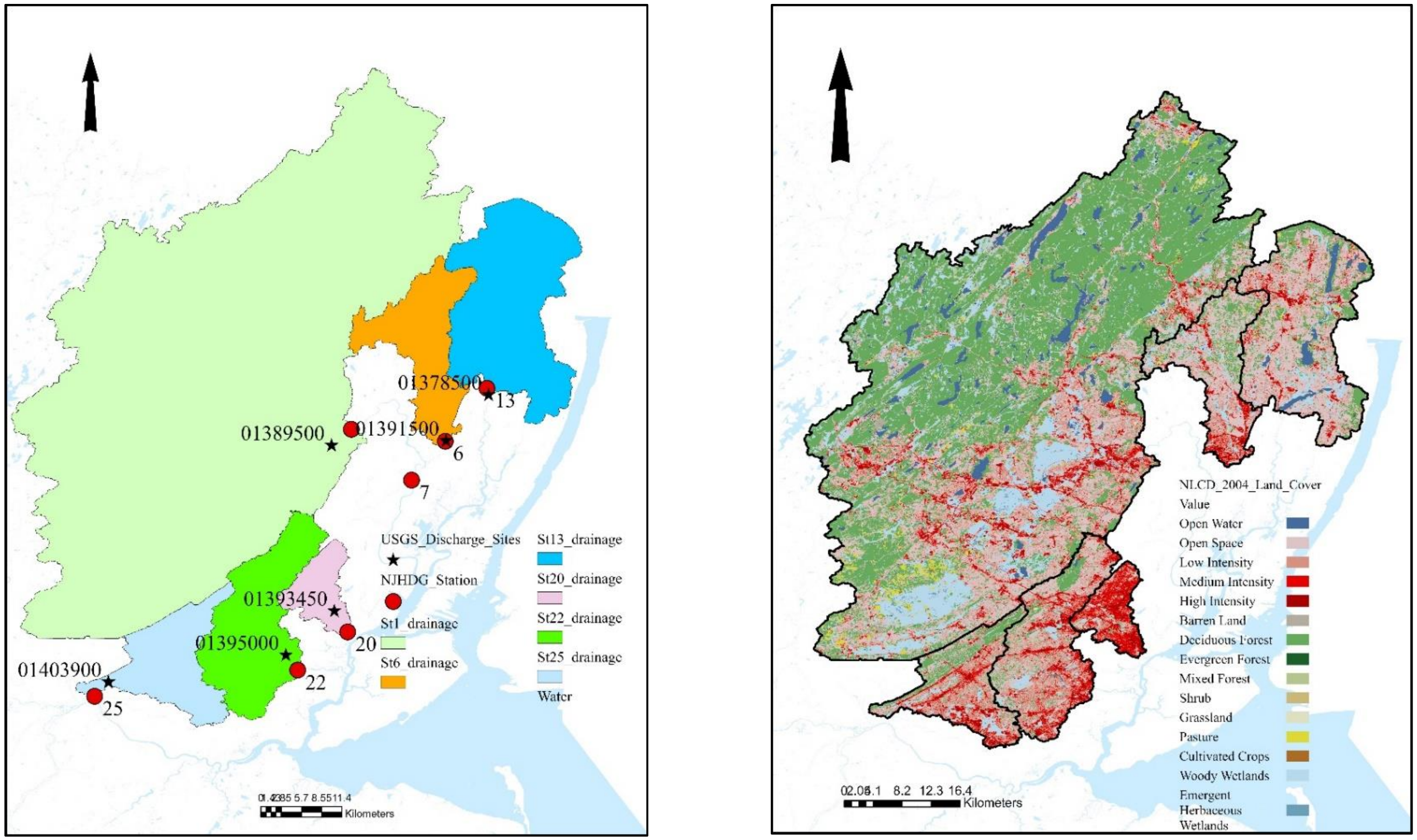
Tsegaye BJIJoAS, Technology F. Effect of land use and land cover changes on soil erosion in Ethiopia. 2019; 5: 26-34.

Varol M. Use of water quality index and multivariate statistical methods for the evaluation of water quality of a stream affected by multiple stressors: A case study. Environ Pollut 2020; 266: 115417.

Wang Y, Gong J, Di CJAP. A building-scale hydrodynamic model for extreme urban flash flooding simulation: A Confluence Area in Raritan River Basin during Hurricane Ida. 2022.

Wieczerak T, Wolde B, Lal P, Witherell B, Deng YJMSG. Public Perception and willingness to pay for green infrastructure improvements in northern New Jersey. 2020; 53: 32-42.

Winfield CH. "Saddle River, New Jersey." History of Bergen County, New Jersey, 1630-1923, 1923.



a b

Figure 5-1 The study area of North New Jersey displaying (a) NJHDG water quality and USGS discharge stations as well as drainage basin of each water quality stations, (b) a 2004 land use/land cover (LULC) map of drainage basin.

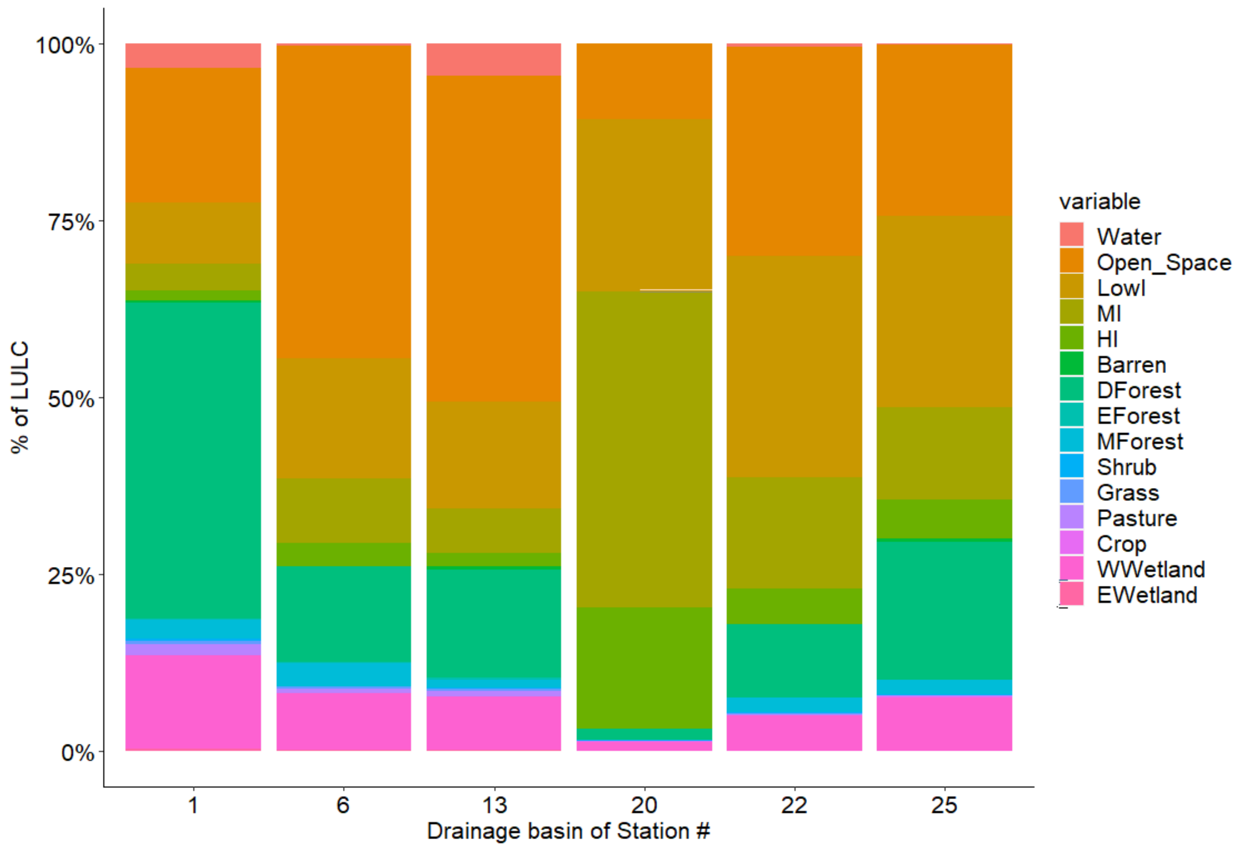


Figure 5-2 Overall (2004-2016) LULC compositions and their proportions within the drainage area of each water quality monitoring station.

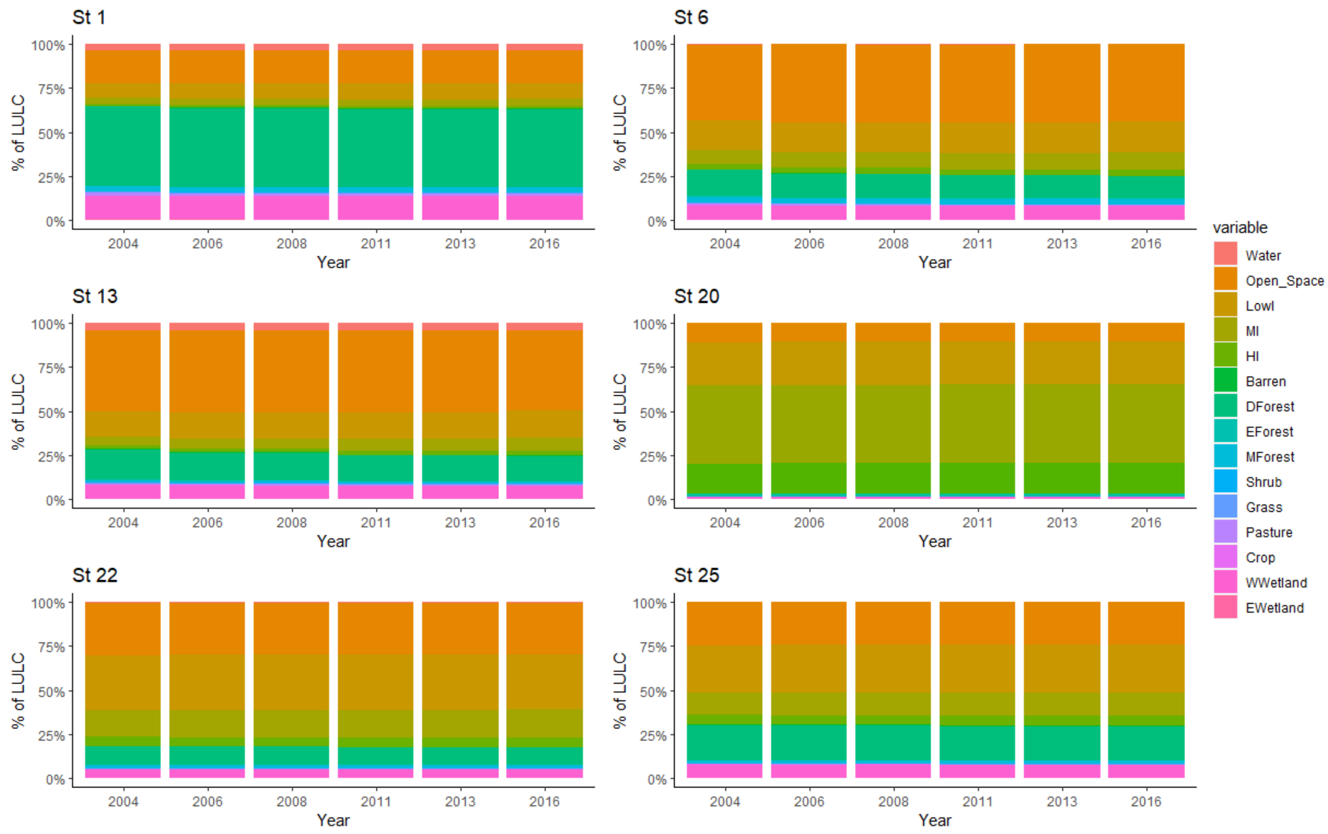


Figure 5-3 The land use land cover (LULC) compositions and proportions of drainage area of each water quality monitoring station during 2004,2006,2008,2011,2013 and 2016.

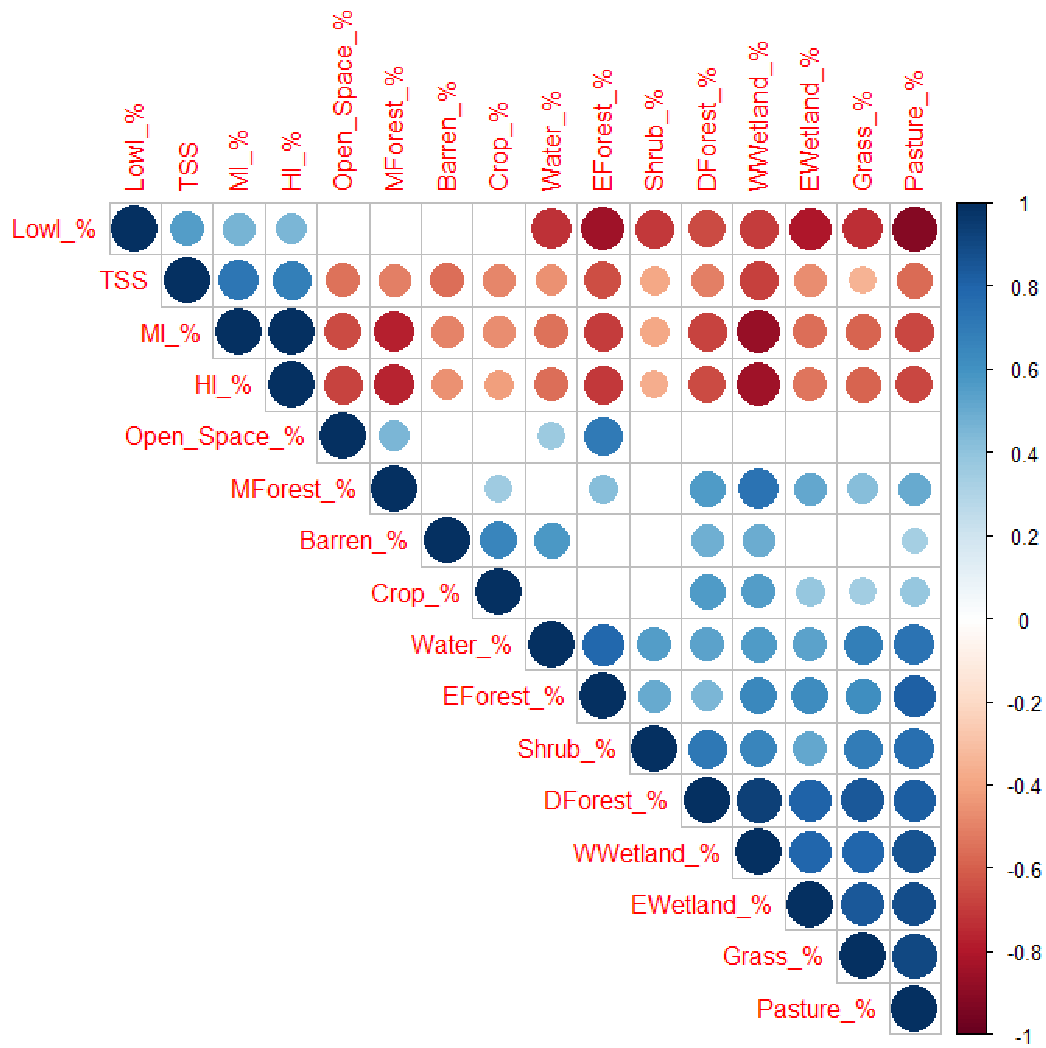


Figure 5-4 Pearson's correlation matrix among different land use type and water quality indicators with significance (p)<0.05.

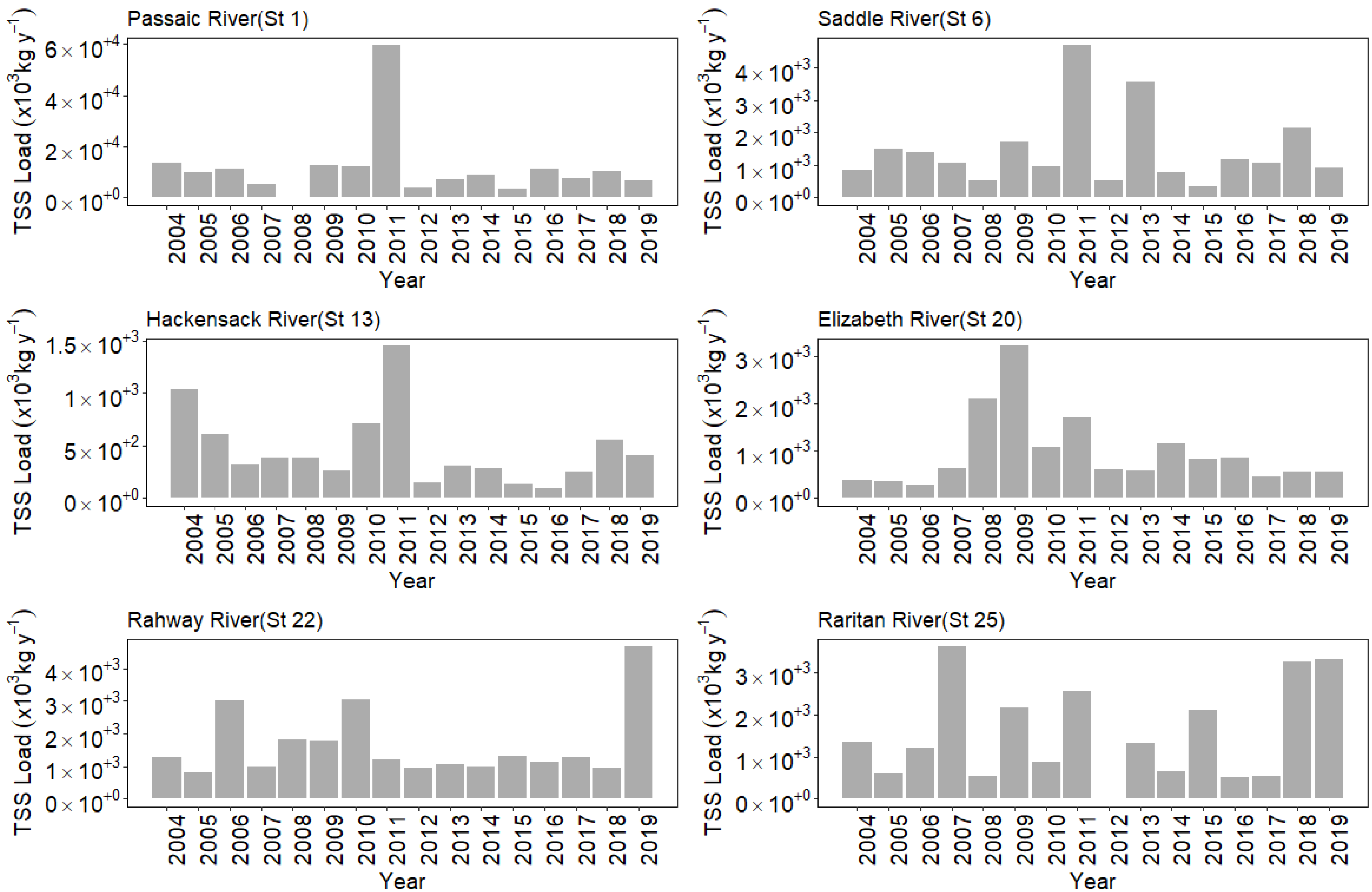


Figure 5-5 Annual-averaged TSS loadings from the six rivers from 2004 to 2019.

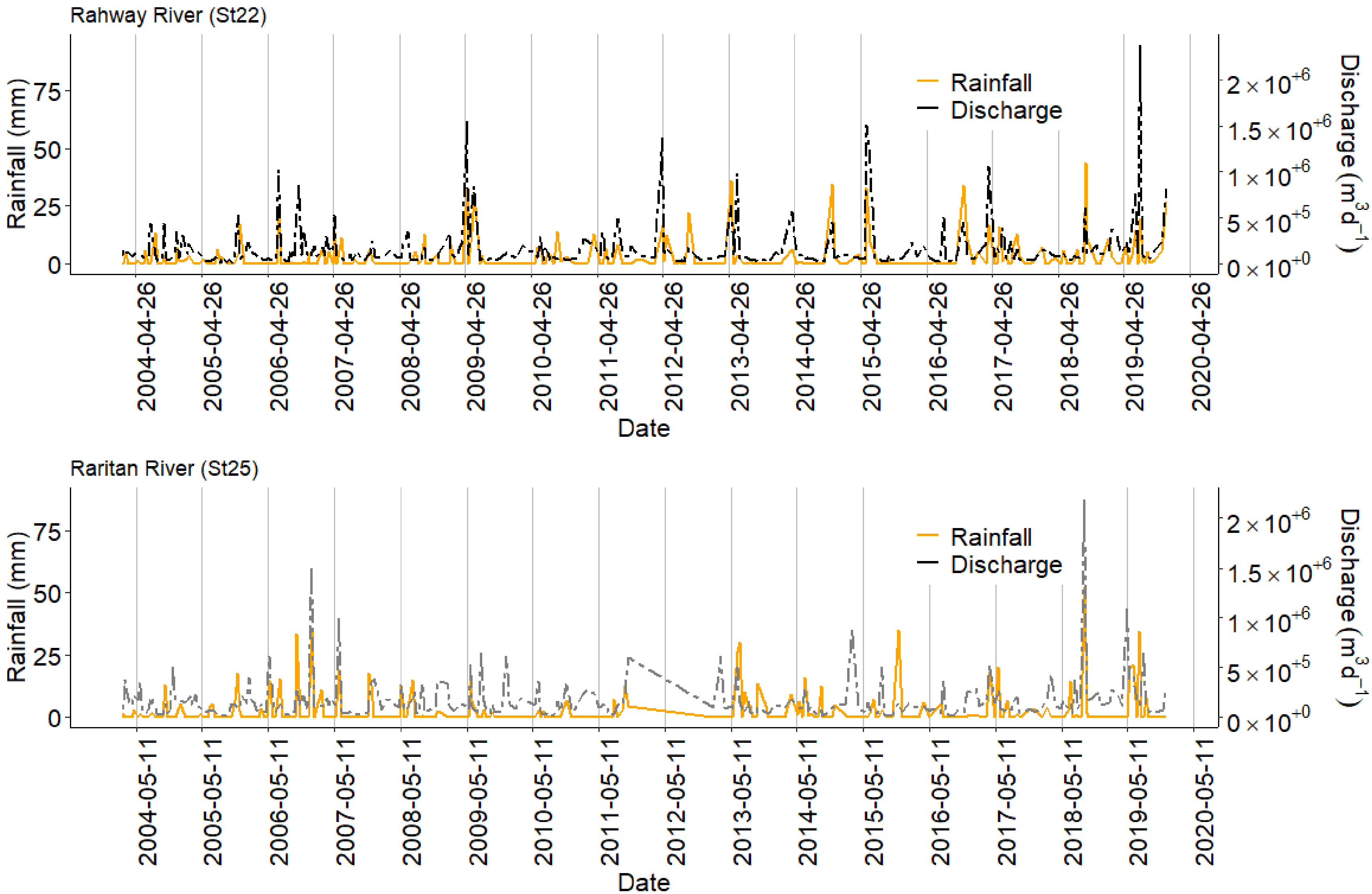


Figure 5-6 Rainfall and discharge data from Rahway River (St22) and Raritan River (St25) (2004-2019).

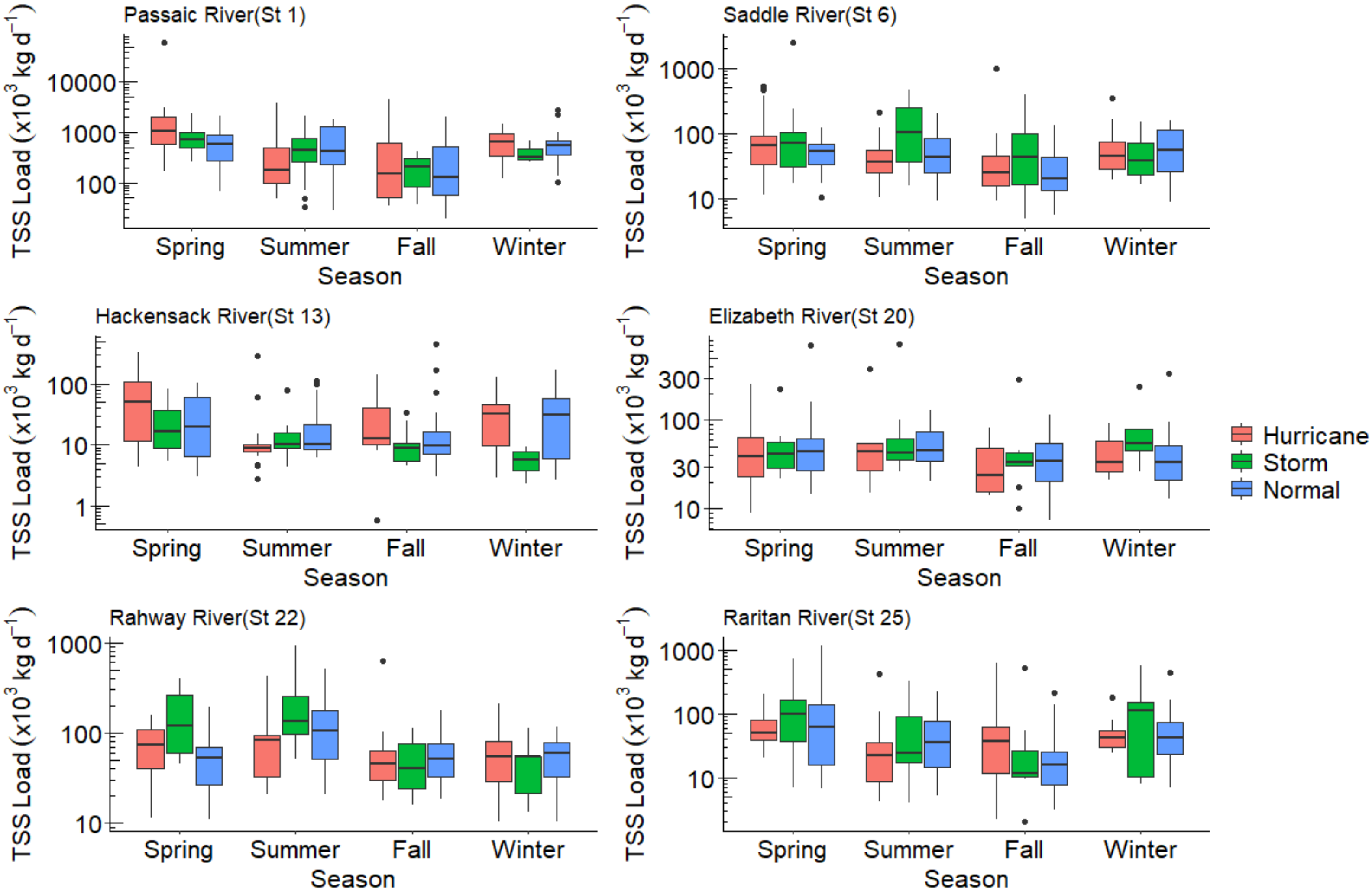
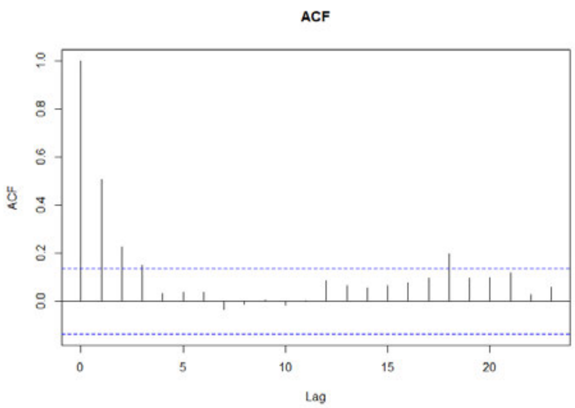
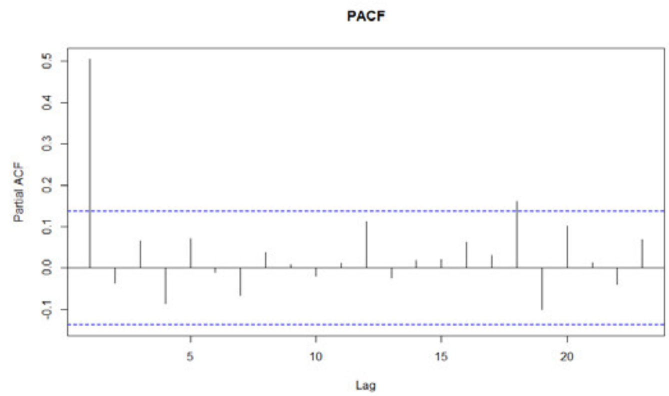


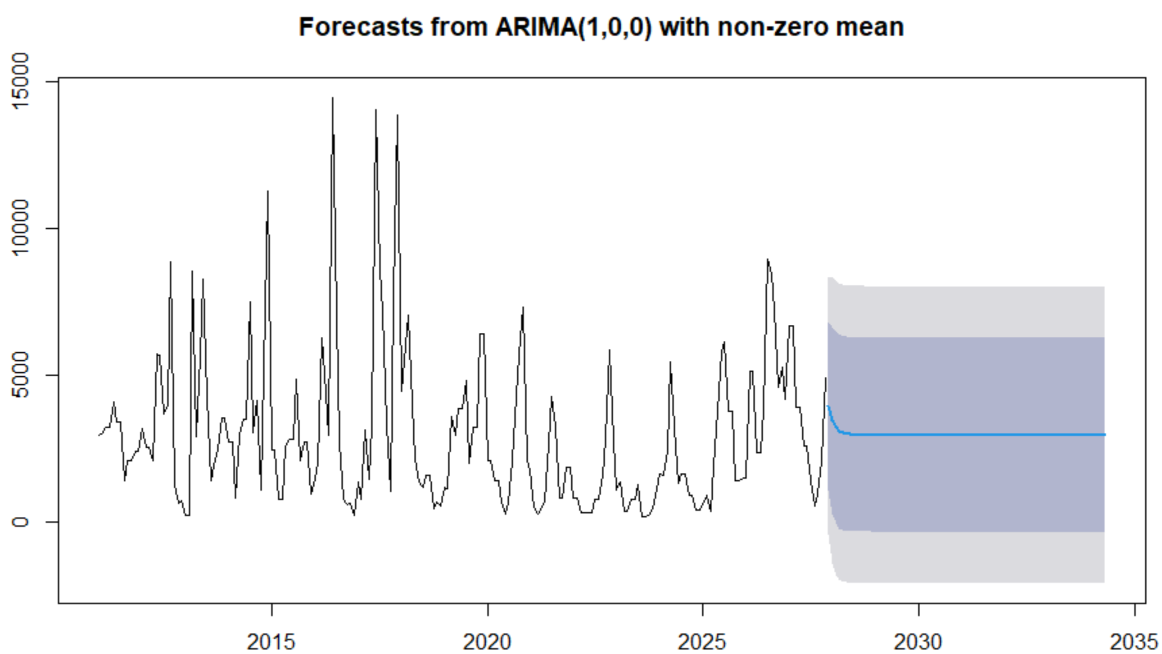
Figure 5-7 Seasonal variations of TSS loadings under hurricane, storm and normal weather conditions in six rivers from 2004 to 2019. Note: Normal weather are years without any hurricane or storm event and hurricane conditions are hurricane occurrence years.



a

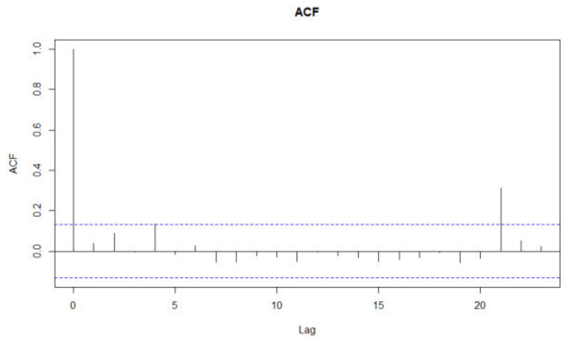


b

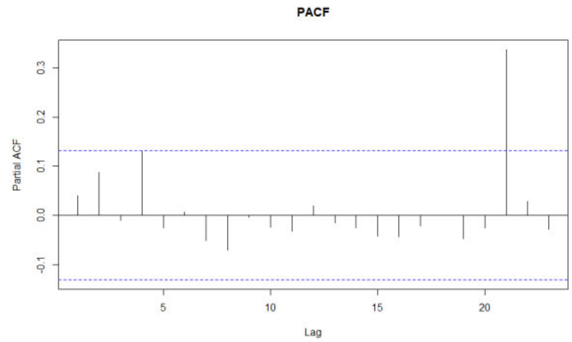


c

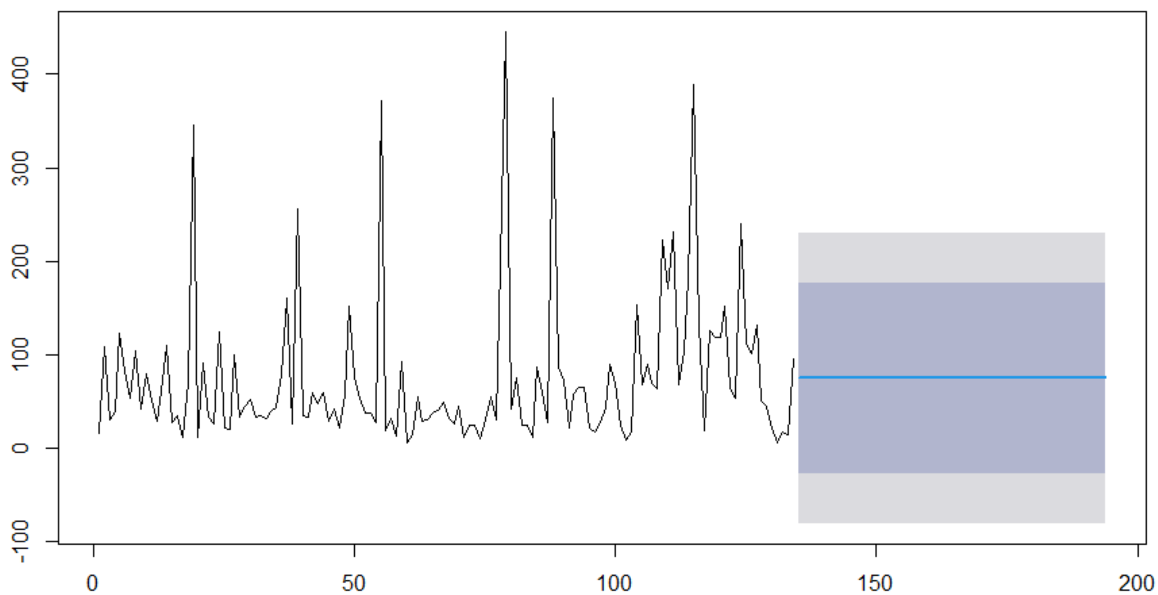
Figure 5-8 Results of the autoregression integrated moving average (ARIMA) analysis of TSS data from Passaic River (**a**) ACF plots of the training data **(b**) PACF plots of the training data; (**c**) model monthly forecasts of TSS load from January 2020 to June 2027.



a

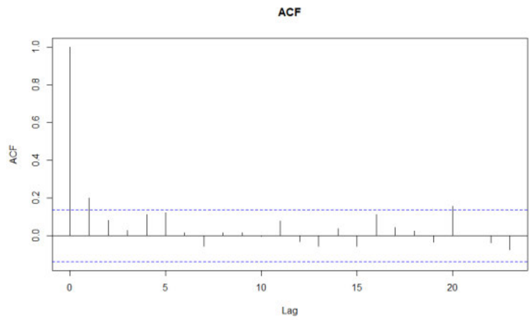


b

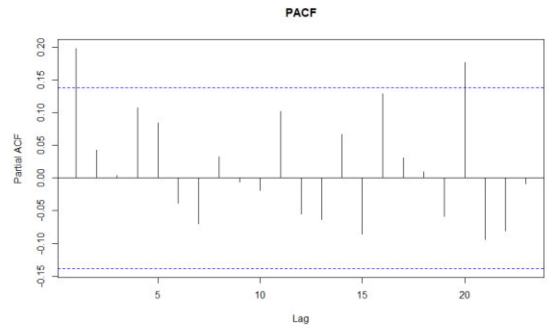


c

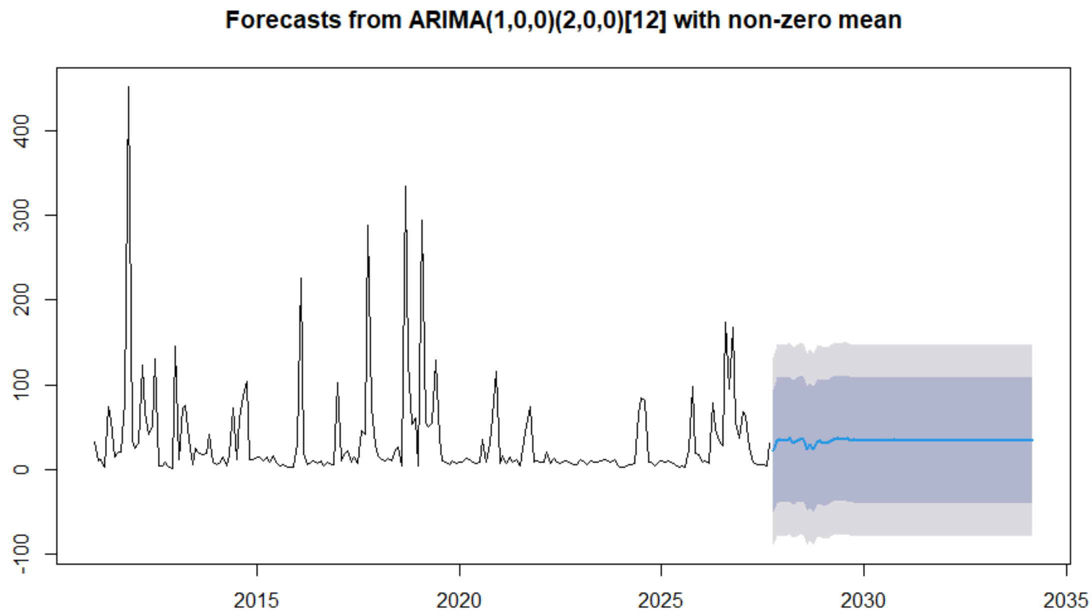
Figure 5-9 Results of the autoregression integrated moving average (ARIMA) analysis of TSS data from Saddle River (a) ACF plots of the training data (b) PACF plots of the training data; (c) model monthly forecasts of TSS load from January 2020 to June 2027.



a



b



c

Figure 5-10 Results of the autoregression integrated moving average (ARIMA) analysis of TSS data from Hackensack River (a) ACF plots of the training data (b) PACF plots of the training data; (c) model monthly forecasts of TSS load from January 2020 to June 2027.

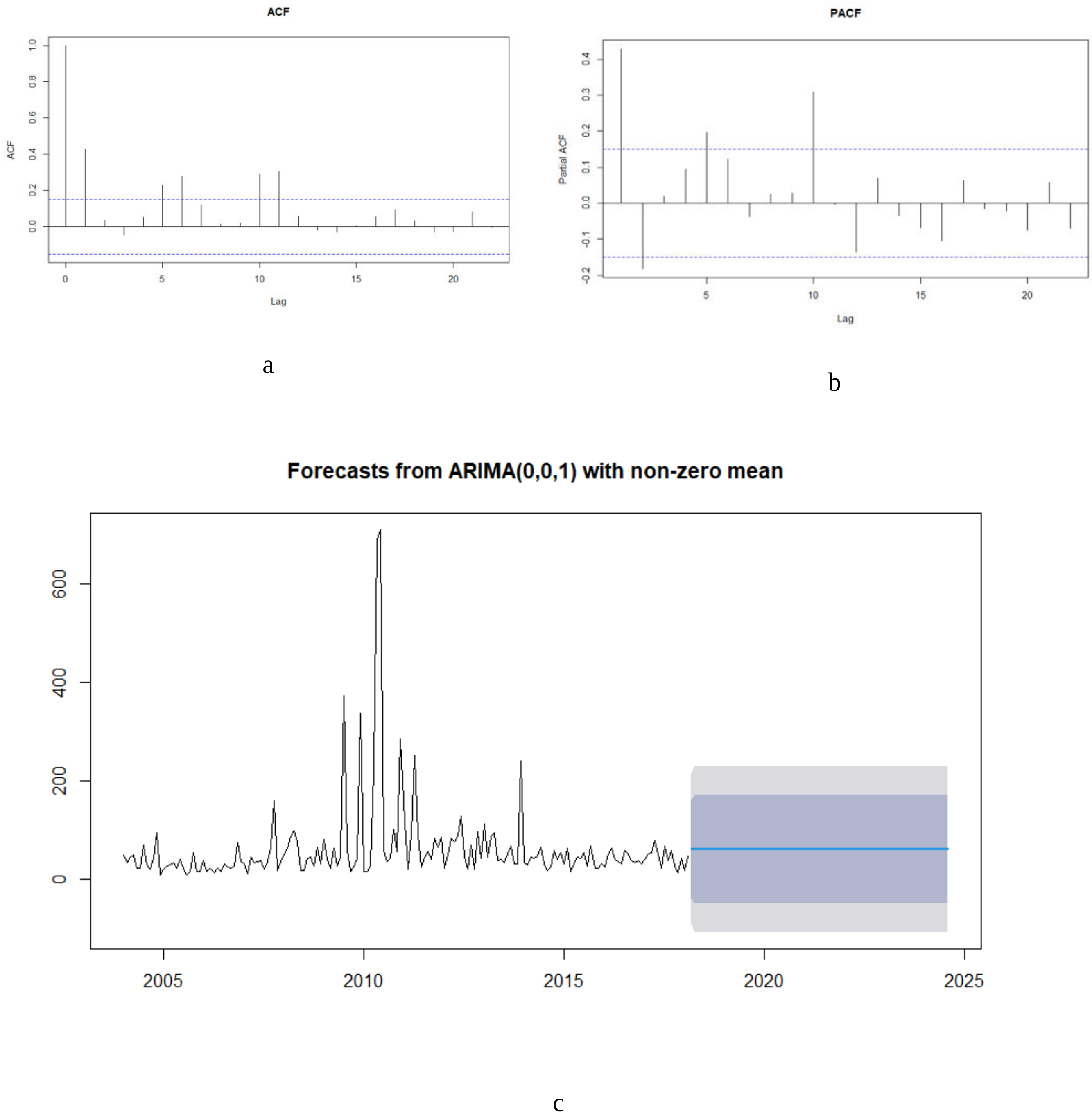
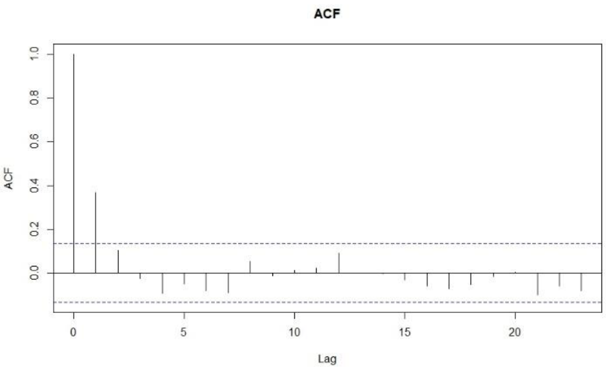
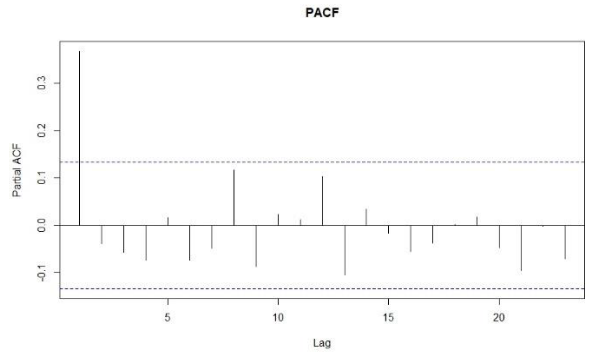


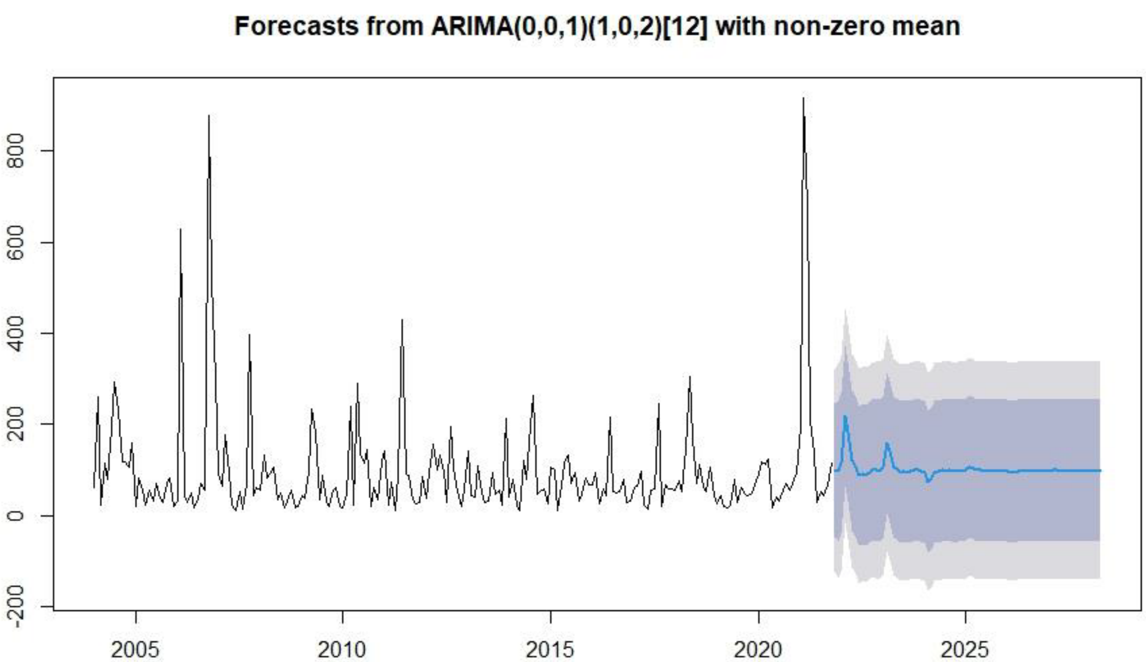
Figure 5-11 Results of the autoregression integrated moving average (ARIMA) analysis of TSS data from Elizabeth River (a) ACF plots of the training data (b) PACF plots of the training data; (c) model monthly forecasts of TSS load from January 2020 to June 2027.



a

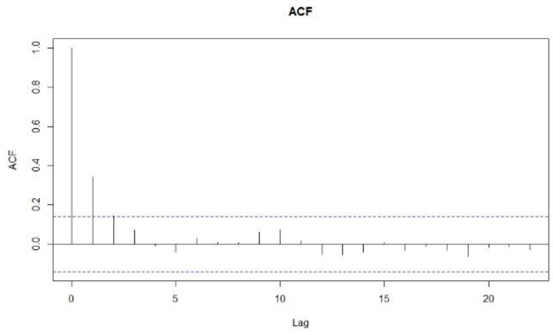


b

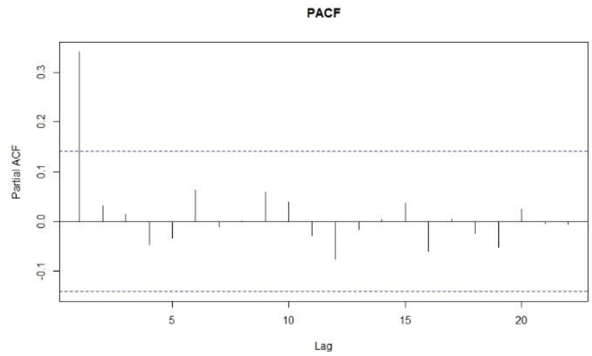


c

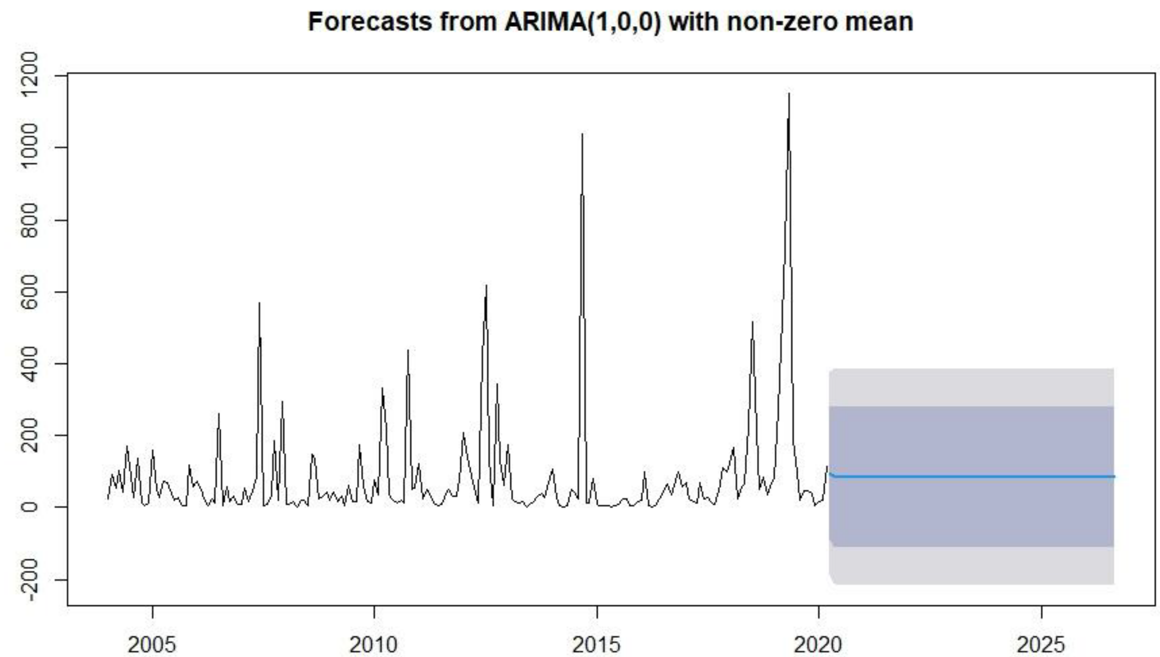
Figure 5-12 Results of the autoregression integrated moving average (ARIMA) analysis of TSS data from Rahway River (a) ACF plots of the training data (b) PACF plots of the training data; (c) model monthly forecasts of TSS load from January 2020 to June 2027.



a



b



c

Figure 5-13 Results of the autoregression integrated moving average (ARIMA) analysis of TSS data from Raritan River (a) ACF plots of the training data (b) PACF plots of the training data; (c) model monthly forecasts of TSS load from January 2020 to June 2027.

Tables

Table 5-1 Water quality stations, discharge sites and river area for this study

River Name	Water Quality Monitoring Station No.	Flow Stations (USGS sites)
Passaic River	1	01389500
Saddle River	6	01391500
Hackensack River	13	01378500
Elizabeth River	20	01393450
Rahway River	22	01395000
Raritan River	25	01403900

Table 5-2 ANOVA and Turkey's test of significance of seasonal variation in TSS loading in different stations (2004-2019).

River Name	Station ID	Weather Condition		
		Hurricane		
		ANOVA	Turkey	Seasonal loading Percentage
Passaic River	1	>0.01	>0.05	Spring (72.9%)>Fall (10.3%)>Winter (9.4%)>Summer (7.5%)
Saddle River	6	>0.01	>0.05	Spring (38.1%)>Fall (26.2%)>Winter (20.0%)>Summer (15.7%)
Hackensack River	13	>0.01	>0.05	Spring (52.5%)>Summer (16.9%)>Winter (16.1%)>Fall (14.5%)
Elizabeth River	20	>0.01	>0.05	Summer (37.0%)>Spring (31.3%)>Winter (17.9%)>Fall (13.8%)
Rahway River	22	>0.01	>0.05	Summer (32.5%)>Fall (24.5%)>Spring (22.7%)>Winter (20.3%)
Raritan River	25	>0.01	>0.05	Fall (31.3%)>Spring (28.4%)>Summer (21.3%)>Winter (19.0%)
		Normal		
		ANOVA	Turkey	Seasonal loading Percentage
Passaic River	1	>0.01	>0.05	Summer (31.4%)>Spring (27.6%)>Winter (24.8%)>Fall (16.2%)
Saddle River	6	>0.01	<0.05	Summer (30.5%)>Winter (27.0%)>Spring (25.4%)>Fall (17.1%)
Hackensack River	13	>0.01	>0.05	Fall (31.1%)>Spring (23.9%)>Winter (24.5%)>Summer (20.5%)
Elizabeth River	20	>0.01	>0.05	Spring (33.3%)>Summer (27.3%)>Winter (21.2%)>Fall (18.2%)
Rahway River	22	<0.01	<0.05	Summer (45.1%)>Spring (20.3%)>Fall (19.7%)>Winter (14.9%)
Raritan River	25	>0.01	<0.05	Spring (54.3%)>Winter (18.4%)>Summer (18.0%)>Fall (9.3%)
		Storm		
		ANOVA	Turkey	Seasonal loading Percentage
Passaic River	1	>0.01	>0.05	Summer (49.4%)>Spring (40.8%)>Winter (4.9%)>Fall (4.9%)
Saddle River	6	>0.01	>0.05	Spring (45.4%)>Summer (41.7%)>Fall (9.6%)>Winter (3.3%)
Hackensack River	13	>0.01	>0.05	Spring (55.3%)>Summer (28.8%)>Fall (13.1%)>Winter (2.8%)
Elizabeth River	20	>0.01	>0.05	Summer (45.2%)>Fall (20.5%)>Spring (17.4%)>Winter (16.9%)
Rahway River	22	>0.01	<0.05	Summer (62.5%)>Spring (23.9%)>Winter (7.1%)>Fall (6.5%)
Raritan River	25	>0.01	>0.05	Spring (32.1%)>Summer (30.6%)>Winter (21.3%)>Fall (16.0%)
		Difference among Hurricane, normal and Storm		
		ANOVA	Turkey	Loading sequence
Passaic River	1	>0.01	>0.05	Hurricane>Storm>Normal
Saddle River	6	<0.01	<0.05	Storm>Hurricane>Normal
Hackensack River	13	>0.01	<0.05	Hurricane>Normal>Storm
Elizabeth River	20	>0.01	>0.05	Storm>Hurricane>Normal
Rahway River	22	<0.01	<0.05	Storm>Hurricane>Normal
Raritan River	25	>0.01	>0.05	Storm>Normal>Hurricane

Table 5-3 ARIMA model coefficient estimation for TSS loading from six rivers.

Passaic River	Coefficients	ar1	ma1	sar1	sar2	sma2	mean
	ARIMA (1,0,0)	0.5048					2962.766
	s.e.	0.0603					312.6746
	Sigma^2 estimated as 4946362, log likelihood=-1852.09, AIC=3710.19						
Saddle River	ARIMA (0,0,0)						86.5554
	s.e.						12.715
	Sigma^2 estimated as						
Hackensack River	Coefficients	ar1	ma1	sar1	sar2	sma2	mean
	ARIMA (1,0,0) × $(2,0,0)_{12}$	0.1936		-0.0543	-0.083		33.9458
	s.e.	0.0693		0.0822	0.0833		4.3509
	Sigma^2 estimated as 3183, log likelihood=-1093.9, AIC=2197.8						
Elizabeth River	ARIMA (0,0,1)		0.461				60.3269
	s.e.		0.0614				8.7137
	Sigma^2 estimated as 6142, log likelihood=-981.78, AIC=1969.56						
Rahway River	Coefficients	ar1	ma1	sar1	sma1	sma2	mean
	ARIMA (0,0,1) × $(1,0,2)_{12}$		0.3601	-0.3817	0.5247	0.1315	97.5684
	s.e.		0.0606	0.9631	0.9582	0.1398	12.346
	Sigma^2 estimated as 12805, log likelihood=-1313.37, AIC=2638.73						
Raritan River	ARIMA (1,0,0)	0.3393					83.7462
	s.e.	0.0671					15.4854
	Sigma^2 estimated as 20732, log likelihood=-1244.85, AIC=2495.69						