GEORGIA DOT RESEARCH PROJECT 18-09

FINAL REPORT

INVESTIGATION ON WATER QUALITY IMPACTS OF BRIDGE STORMWATER RUNOFF FROM SCUPPER DRAINS ON RECEIVING WATERS

Georgia Department of Transportation

OFFICE OF PERFORMANCE-BASED MANAGEMENT AND RESEARCH

600 WEST PEACHTREE STREET NW
ATLANTA, GA 30308

TECHNICAL REPORT DOCUMENTATION PAGE

1. Report No.: FHWA-GA-21-1809	2. Government Accession No.: N/A	3. Recipient's Catalog No.: N/A
4. Title and Subtitle: Investigation on Water Quality Impacts of Bridge Stormwater Runoff from Scupper Drains on Receiving Waters		5. Report Date: February 2021 6. Performing Organization Code: N/A
7. Author(s): George Yuzhu Fu (PI), Ph.D., P.Eng. (https://orcid.org/0000-0001-7620-1843); Ahmed Bakr, Ph.D., (https://orcid.org/0000-0002-4532-2607)		8. Performing Organization Report No.: 18-09
9. Performing Organization Name and Address: Georgia Southern University 201 COBA Drive, Building 232 Statesboro, GA 30460-8077 Phone: (912) 478-5003 Email: gfu@georgiasouthern.edu		10. Work Unit No.: 0190501 11. Contract or Grant No.: 39G9573 PI#0016326
12. Sponsoring Agency Name and Address: Georgia Department of Transportation Office of Performance-based Management and Research 600 West Peachtree St. NW Atlanta, GA 30308		13. Type of Report and Period Covered: Final; Start September 2018 – End February 2021 14. Sponsoring Agency Code: 000031
15. Supplementary Notes: Prepared in cooperation with the U.S. Department of Transportation, Federal Highway Administration.		

16. Abstract:

The objectives of this research project were to investigate water quality impacts of bridge stormwater runoff from scupper drains on receiving waters, and to develop an efficient simulation tool so that the Georgia Department of Transportation (GDOT) and the resource agencies can accurately anticipate potential effects and better evaluate whether scupper drains would adversely affect waters and the protected species. This 29-month project is concerned with the investigation of potential impacts and accumulation of water quality parameters including heavy metals (lead, zinc and copper), polycyclic aromatic hydrocarbons (PAHs), chemical oxygen demand (COD), nutrients, oil and grease, solids, pH, dissolved oxygen, and conductivity in bridge deck scupper drain runoff of six (6) bridge sites and the potential impact on sensitive major rivers in the State of Georgia. Four (4) of the sites are Southeast the State of Georgia, which are SR24, Rocky Ford (RF), US80 and SR297, crossing two (2) major rivers, which are Ogeechee River and Ohoopee River. The other two (2) sites are located North the State of Georgia, which are US255 and SR197, crossing two (2) other major rivers, which are Chattahoochee River and Soque River, respectively. The impact of the elevated parameters due to introduction from bridge deck scupper drains was observable on downstream water quality of all sites, except for lead, PAHs, phosphorous, and oil and grease that were only detected at the four (4) Southeast sites. It was observed that upon discharge of bridge deck scupper drain runoff, instant impairment of downstream water quality took place. Seasonal analyses showed variation in the parameters concentrations that was highly dependant on many factors, such as dry periods, rain intensity, traffic activity and stream discharge. Time intervals analyses showed that highest concentrations of contaminants were in the first 30-60 minutes' samples, and that parameters concentration decrease with continuous rainfall due to dilution. The Stochastic Empirical Loading and Dilution Model (SELDM) was used for modeling potential projected impacts of some water quality parameters of concern on the downstream water quality at each Site location effectively.

17. Keywords: Water Quality Impacts; Bridge Deck Stormwater Runoff; Scupper Drains; Stochastic Empirical Loading and Dilution Model (SELDM)	18. Distribution Statement: No Restriction		
9. Security Classification (of this report): Unclassified	20. Security Classification (of this page): Unclassified	21. No. of Pages: 178	22. Price: Free

Form DOT 1700.7 (8-72) Reproduction of completed page authorized

GDOT Research Project No. 18-09

Final Report

INVESTIGATION ON WATER QUALITY OF BRIDGE STORMWATER RUNOFF FROM SCUPPER DRAINS ON RECEIVING WATERS

By
George Yuzhu Fu, Ph.D., P.Eng.
Professor – Department of Civil Engineering and Construction

Ahmed Bakr, Ph.D.
Postdoctoral Associate

Georgia Southern University Research & Service Foundation, Inc.

Contract with
Georgia Department of Transportation

In cooperation with
U.S. Department of Transportation Federal Highway Administration

February 2021

SI* (MODERN METRIC) CONVERSION FACTORS				
APPROXIMATE CONVERSIONS TO SI UNITS				
Symbol	**When You Know**	**Multiply By**	**To Find**	**Symbol**
		LENGTH		
in	inches	25.4	millimeters	mm
ft	feet	0.305	meters	m
yd	yards	0.914	meters	m
mi	miles	1.61	kilometers	km
		AREA		
in^2	square inches	645.2	square millimeters	mm^2
ft^2	square feet	0.093	square meters	m^2
yd^2	square yard	0.836	square meters	m^2
ac	acres	0.405	hectares	ha
mi^2	square miles	2.59	square kilometers	km^2
		VOLUME		
fl oz	fluid ounces	29.57	milliliters	mL
gal	gallons	3.785	liters	L
ft^3	cubic feet	0.028	cubic meters	m^3
yd^3	cubic yards	0.765	cubic meters	m^3
		NOTE: volumes greater than 1000 L shall be shown in m^3		
		MASS		
oz	ounces	28.35	grams	g
lb	pounds	0.454	kilograms	kg
T	short tons (2000 lb)	0.907	megagrams (or "metric ton")	Mg (or "t")
		TEMPERATURE (exact degrees)		
^oF	Fahrenheit	5 (F-32)/9 or (F-32)/1.8	Celsius	^oC
		ILLUMINATION		
fc	foot-candles	10.76	lux	lx
fl	foot-Lamberts	3.426	candela/m^2	cd/m^2
		FORCE and PRESSURE or STRESS		
lbf	poundforce	4.45	newtons	N
lbf/in^2	poundforce per square inch	6.89	kilopascals	kPa
APPROXIMATE CONVERSIONS FROM SI UNITS				
Symbol	**When You Know**	**Multiply By**	**To Find**	**Symbol**
		LENGTH		
mm	millimeters	0.039	inches	in
m	meters	3.28	feet	ft
m	meters	1.09	yards	yd
km	kilometers	0.621	miles	mi
		AREA		
mm^2	square millimeters	0.0016	square inches	in^2
m^2	square meters	10.764	square feet	ft^2
m^2	square meters	1.195	square yards	yd^2
ha	hectares	2.47	acres	ac
km^2	square kilometers	0.386	square miles	mi^2
		VOLUME		
mL	milliliters	0.034	fluid ounces	fl oz
L	liters	0.264	gallons	gal
m^3	cubic meters	35.314	cubic feet	ft^3
m^3	cubic meters	1.307	cubic yards	yd^3
		MASS		
g	grams	0.035	ounces	oz
kg	kilograms	2.202	pounds	lb
Mg (or "t")	megagrams (or "metric ton")	1.103	short tons (2000 lb)	T
		TEMPERATURE (exact degrees)		
^oC	Celsius	1.8C+32	Fahrenheit	^oF
		ILLUMINATION		
lx	lux	0.0929	foot-candles	fc
cd/m^2	candela/m^2	0.2919	foot-Lamberts	fl
		FORCE and PRESSURE or STRESS		
N	newtons	0.225	poundforce	lbf
kPa	kilopascals	0.145	poundforce per square inch	lbf/in^2

* SI is the symbol for the International System of Units. Appropriate rounding should be made to comply with Section 4 of ASTM E380. (Revised March 2003)

TABLE OF CONTENTS

LIST OF FIGURES

LIST OF TABLES

LIST OF ABBREVIATIONS

AADT	Annual Average Daily Traffic
ADT	Average Daily Traffic
AN	Ammonia Nitrogen
AWRC	Arkansas Water Resource Center
BAM	Biosorption-activated Media
BDF	Basin Development Factor
BOD	Biological Oxygen Demand
BMPs	Best Management Practices
BTEX	Benzene, Toluene, Ethyl Benzene, and Xylene
cfs	Cubic Feet per Second
COD	Chemical Oxygen Demand
DO	Dissolved Oxygen
DTV	Daily Traffic Volume
EPA	Environmental Protection Agency
FHWA	Federal Highway Administration
GCMS	Gas Chromatograph Mass Spectrometer
ISA	Ionic Strength Adjustor
ISE	Ion Selective Electrode
MCTT	Multi-Chambered Treatment Train
NHMRC	National Health and Medical Research Council

ON	Organic Nitrogen
OP	Orthophosphorus
PAHs	Polycyclic Aromatic Hydrocarbons
PBDEs	Polybrominated Diphenyl Ethers
PCBs	Polychlorinated Biphenyls
PEC	Predicted Environmental Concentration
PFAAs	Perfluoroalkyl Acids
PFCs	Perfluorinated Compounds
PFDA	Long-Chain-Length PFCs
PFOS	Perfluorooctane Sulfonate
PM	Particulate Matter
PNEC	Predicted No Effect Concentration
ppm	Parts Per Million
ppb	Parts Per Billion
ppt	Parts Per Trillion
PVC	Polyvinyl Chloride
RF	Rocky Ford
SCMs	Stormwater Control Measures
SELDM	Stochastic Empirical Loading and Dilution Model
SS	Suspended Solids
TDS	Total Dissolved Solids
TH	Total Hardness

TKN	Total Kjeldahl Nitrogen
TMDL	Total Maximum Daily Load
TN	Total Nitrogen
TP	Total Phosphorus
TS	Total Solids
UNEP	United Nations Environmental Programme
UNESCO	United Nations Educational, Scientific and Cultural Organization
USGS	United States Geological Survey
UV	Ultraviolet
VOC	Volatile Organic Compound
VPD	Vehicles Per Day
WERL	Water and Environmental Research Laboratory
W_f	Final Weight
W_i	Initial Weight
WHO	World Health Organization

EXECUTIVE SUMMARY

The proposed objectives of this research project are to investigate water quality impacts of bridge stormwater runoff from scupper drains on receiving waters, and to develop an efficient simulation tool so that the Georgia Department of Transportation (GDOT) and the resource agencies can accurately anticipate potential effects and better evaluate whether scupper drains would adversely affect waters and the protected species.

This 29-month research project aimed at investigating the impact of bridge deck stormwater runoff scupper drains in the Southeast and North regions of the State of Georgia on stream water quality. Several studies have investigated the water quality impacts of highway stormwater runoff in other states with a few that focused on bridge deck runoff scupper drain impacts on sensitive streams and rivers. However, the water quality impacts of bridge deck runoff from scupper drains on sensitive habitats and essential streams are lacking for the State of Georgia. Six bridge locations were selected, of which four are in Southeast Georgia and two are in North Georgia. This research project aimed at covering Southeast and North Georgia to follow the bridge deck stormwater runoff scupper drains impacts at the selected locations through a collaboration between the Georgia Southern University (GSU) Research Team and GDOT.

In this research project, field sampling and lab analyses were the primary focus in order to obtain actual measurements of several water quality parameters of potential impact on stream water quality in Georgia. The water quality parameters of concern were heavy metals (i.e., lead [Pb], zinc [Zn], and copper [Cu]), polycyclic aromatic hydrocarbons (PAHs), chemical oxygen demand (COD), nutrients (i.e., nitrogen and phosphorus), solids, oil and grease, pH, dissolved oxygen (DO), and conductivity. These water quality parameters were selected based on the knowledge that the major sources of stream contamination from bridge deck runoff scupper drains

are traffic and vehicles leakages, atmospheric fallout, bridge maintenance and bridge deck material leaching, after a thorough literature review. Since bridges are direct crossings above rivers and sensitive streams, unlike normal highways and roads, bridge scupper drains are considered direct points of discharge of several contaminants and substances of hazardous concern into underlying water bodies that do not have prior natural or constructed means of filtration or conveyance.

The potential impact of bridge deck scupper drain contamination is considered minor with respect to quantitative evaluation or mass loading of pollutants (i.e., the product of flow rate from bridge deck storm water runoff through scupper drains and the concentrations of water quality parameters of concern). However, the accumulative effect of the several pollutants is of serious concern on the long-term exposure to humans and animals. Moreover, sensitive habitats of rivers and streams can be severely impacted after the exposure to even small amounts of toxic substances. In fact, this research project successfully shed light on the substantial potential impacts of several of the contaminants in bridge deck runoff scupper drains on instream water quality through the insightful investigation of both bridge deck runoff and the instream water quality. Sampling of bridge deck runoff took place at scupper drains using automated and manual means.

The examination of water quality in this research project included extensive upstream, scupper drain, and downstream water analyses that involved observation and effective comparison between the instream and scupper drain water quality. The impact of bridge deck scupper drains on downstream water quality was observed for different parameters at all six (6) bridge sites, and it was observed that the locations that are more urbanized or close to urban locations have the higher impact of impaired water quality. Further examination of bridge deck scupper drains was done for extended periods of times to observe concentrations in the first flush of rainfall and the impact of further rainfall on concentrations of all water quality parameters. Rainfall intensity had a substantial effect on parameter concentrations in scupper drain runoff, and it was found that for most parameters except for solids, higher rainfall intensity led to a dilution effect for parameter concentrations with time. For heavy metals, Pb, Zn, and Cu were found to be of higher concentrations in scupper drain runoff and, thus, could impair instream water quality and elevated downstream concentrations in all six (6) sites, except for Pb, which was only detected in the four Southeast Georgia sites. Bridge deck scupper drains were also investigated on different subsequent dates to determine the impact of dry periods versus rainfall events on the mass loading and concentrations of the studied parameters. Other factors of rainfall intensity and stream discharge were also observed through obtaining data from the United States Geological Survey (USGS) gage stations at each site to have a better understanding of the effects associated with rainfall intensity on bridge deck runoff scupper drain water quality and the effect of stream discharge on instream water quality. This research project showed that bridge deck scupper drain runoff impairs downstream water quality for all parameters of the study, including heavy metals (Pb, Zn, and Cu), PAHs, nitrogen and phosphorus, COD, oil and grease, pH, DO, and conductivity. It was found that the introduction of bridge deck scupper drain runoff elevated the concentration of these parameters downstream in all six streams of the study, both instantly and with time intervals. The lab analyses also showed that the concentrations of parameters varied with different sampling dates, but randomly.

Modeling water quality impact for each site was an important part of the study to project potential long-term impacts of the study sites and to create the opportunity for GDOT and other government agencies to prepare the necessary best management practices (BMPs) for such long-term impacts. The Stochastic Empirical Loading and Dilution Model (SELDM) was used for modeling potential projected impacts of some water quality parameters of concern on the downstream water quality at each site location. The model was deemed effective for projecting potential long-term impacts of pH, phosphorus, sediment, and hardness. The model could successfully predict impacts along 30 years of time and 1,657 storm events, and it was observed to be a very efficient comparative tool to actual field and lab outcomes. The results from running SELDM showed that the North Georgia sites have the highest concentrations of phosphorus and sediment and that the accumulation effect is minor. However, lab data showed that the North Georgia sites have the lowest concentration of all parameters from bridge scupper drains. Thus, a contradiction was found in SELDM outcome in comparison to lab data, indicating that SELDM could also employ other factors through watershed delineation to produce results, which was not the case with instant or short-term lab analyses.

The major takeaways from this project can be summarized as follows:

- Scupper drains without any prior pretreatment or conveyance can directly impact water quality of underlying water bodies in the short-term.
- The impact of scupper drains on downstream water quality is observable for all water quality parameters.
- BMPs are challenging to implement for bridge stormwater runoff from scupper drains because they are crossings over water bodies.
- Continuous monitoring of water quality impact from bridge scupper drain runoff needs to be ongoing for observing long-term accumulative impacts.

SELDM needs to be applied for most water quality parameters of concern instead of only for the four default parameters: pH, total phosphorus, total hardness, and suspended solids.

CHAPTER 1. INTRODUCTION

Bridges have a direct polluting impact on underlying water bodies since bridges are built as passages over surface water bodies. Bridge deck runoff is a platform for various pollutants and, unlike roads and highways, represents a special case since there are no obstructions such as roadside swales and vegetated filter strips between the bridge deck and underlying water bodies.[1-4] Bridge deck runoff contains a medium to wide range of contaminants produced from different sources, including organic contaminants, heavy metals, solids, salts, and nutrients. Sources of pollutants and chemicals in bridge deck runoff can be traffic, atmospheric fallout, degraded bridge deck and road components, construction activities, and industrial emissions.[1,2-15] Several factors make the structural control and treatment of bridge deck runoff challenging, including administrative, design, and maintenance issues and economic concerns.[1,4,16-18] Such factors make the control over bridge deck runoff contamination unpractical.

Pollutants accumulate on bridge deck surfaces during dry periods, and the longer the dry period, the higher the accumulation of pollutants from different sources, such as crossing vehicles, atmospheric fallout in the form of particulate matter and toxins, degradation of bridge surfaces, wear of vehicle components, and road maintenance operations. During rainfall events, stormwater sweeps the deck surface and transports the accumulated contaminants to underlying water bodies by the discharge of runoff through scupper drains and gutter systems, which are engineered discharge points constructed in the deck surface. Extremely toxic substances can be introduced to the bridge deck when leaks or spills occur from vehicles transporting hazardous materials and toxic waste, and subsequently will be conveyed to water bodies with bridge deck runoff discharge.

The more frequent winter maintenance of bridge decks as compared to highways, and the pollutant buildup along the bridge curb wall can lead to the accumulation of more contaminants as compared to highways, such as total nitrogen (TN), total phosphorus (TP), and total suspended solids (TSS), heavy metals, and chemical oxygen demand (COD).[1,10,15] Traffic is a major source of bridge deck contamination and it alone drives the formation and introduction of many chemicals in bridge stormwater runoff without prior filtration, such as polycyclic aromatic hydrocarbons (PAHs), volatile organic compounds (VOCs) and semi-VOCs, oil and grease, asbestos, lead (Pb), cadmium (Cd), and copper (Cu) (See references 1, 2, and 19–22). Moreover, galvanization of bridge metal components adds the heavy metals load of zinc (Zn), Cu, and iron (Fe) (See references 1, 5, 8, 11, 15, and 23).

Continuous introduction of contaminants from bridge decks can cause acute and chronic toxicity in aquatic life and humans under prolonged and sometimes instant exposure to heavy metals, while other salts contribute to aquatic life toxicity and can impair water quality and render it unsuitable for human consumption.[1,12] The continuous introduction of nitrogen and phosphorus from bridge deck runoff can add to the increased nutrient concentration in water bodies, which subsequently increases the eutrophication effect in water bodies.[1] Particulate matter (PM) is one of the prominent contaminants in bridge deck runoff since PMs have the capacity to adsorb and transport various other pollutants from the atmosphere. PMs can carry contaminants such as carbon monoxide (CO); nitrogen oxides (NOx); VOCs and semi-VOCs; sulfur dioxide (SO_2); methane (CH_4); and toxins, including benzene, toluene, ethyl benzene, and xylene (BTEX), butadiene and formaldehyde, among others. Such contaminants can concentrate on a PM's surface and PMs can introduce these contaminants to the aquatic environment when they settle with precipitation and are swept with bridge deck runoff into water bodies.[1] Moreover, PMs can catalyze the interaction and transform such pollutants into other toxic compounds and by-products that can be introduced to receiving waters via the bridge scupper and gutter drainage systems.[1]

This research project is a collaboration between the Georgia Southern University (GSU) Research Team in the Water and Environmental Research Laboratory (WERL) at GSU and the Georgia Department of Transportation (GDOT). The project aims to investigate the impact of the introduction of bridge deck scupper drain runoff on sensitive receiving streams in the Southeast and North regions of the State of Georgia. The project started in September 2018 and is targeted to end in December 2020, with 2 years' duration, throughout which an intense preparation for field, lab, and modeling works was undertaken to fulfill the study. The project included field installations and sampling, lab analyses, and modeling simulations, which all aimed at obtaining a comprehensive understanding of the potential impacts of the introduction of contaminants in bridge deck scupper drain stormwater runoff on sensitive streams in Georgia.

In this research project, a total of six bridges were selected as study sites in Georgia. Data on the impact of bridge deck scupper drain runoff on water quality for the State of Georgia is scarce, and it is important to advance further studies on the impact of bridge deck scupper drain runoff on fresh water and sensitive streams on more states in the United States. Four study sites were selected in Southeast Georgia and two sites were selected in North Georgia. Different water quality parameters were studied, including heavy metals, PAHs, nutrients, solids, COD, oil and grease, pH, dissolved oxygen (DO), and conductivity, aiming to have a full scope on potential impacts of bridge deck scupper drain runoff in Southeast and North Georgia. Sampling was done manually and automatically for bridge deck scupper drain runoff and instream (i.e., upstream and downstream of each bridge sampling point) for quantifying impacts. Based on field and lab outcomes, it was found that bridge deck stormwater runoff through scupper drains can have a potential and, in some instances, a significant impact on instream water quality. Moreover, the Stochastic Empirical Loading and Dilution Model (SELDM) was employed as an efficient projection and comparative tool to further investigate possible long-term impacts of the six bridge sites on their respective downstream water quality.

This research project shows that bridge deck scupper drain runoff impairs downstream water quality for all parameters of the study, including heavy metals (lead, copper, and zinc); PAHs; nitrogen and phosphorus; COD; oil and grease; pH; DO; and conductivity. It was found that the introduction of bridge deck scupper drain runoff elevated the concentration of these parameters downstream at all six streams of the study, both instantly and with time intervals. The lab analyses also showed that the concentrations of parameters varied with different sampling dates, but randomly. SELDM showed that the North Georgia sites have the highest concentrations of phosphorus and sediment, and that the accumulation effect is minor. However, lab data showed that the North Georgia sites have the lowest concentration of all parameters from bridge scupper drains. Such outcome indicated that SELDM could also consider several watershed factors in making its long-term projections.

CHAPTER 2. LITERATURE REVIEW

SOURCES AND IMPACTS OF STORMWATER POLLUTION

Types of Pollutants, and Their Sources and Impact Classification

Bridge deck and highway stormwater runoff can hold a large array of chemicals due to continuous exposure to various sources of contamination. The common constituents that can be found in bridge deck runoff are heavy metals; inorganic salts; aromatic hydrocarbons; suspended solids; and compounds that are shed by vehicles, such as oil, grease, rust, and rubber particles.[5,14,24] All these components are commonly produced from motorized sources, roadway substances, construction activities, and maintenance activities. There are several factors that can affect the distribution and concentrations of the pollutants on a bridge deck, such as antecedent dry periods, seasonal cumulative rainfall, rainfall intensity, and land use.[5,25] Table 1 lists common bridge and highway stormwater runoff pollutants, their different sources, and their hazardous effects on human health and aquatic biota.

Table 1. Common bridge and highway stormwater runoff pollutants, their sources, and their hazardous effects on human health and aquatic biota.

Pollutant	Source	Effects	Ref.
PAHs	Burning of trash and fuel combustion operations	Cytotoxic and carcinogenic, and can cause reproductive complications	[3, 6, 12, 14, 26–28]
Solids	Construction and maintenance activities, atmospheric fallout, dust	Exposure to dissolved solids constituting toxic metals can have health effects	[3, 5, 6, 12, 14, 29]
BTEXs	Petroleum spills and leaks	Acute toxicity, carcinogenic	[6]
Cadmium	Tire wear, insecticides	Acute toxicity, carcinogenic	URS, [5, 6, 12, 14]

Table 1. Continued.

Pollutant	Source	Effects	Ref.
Chromium	Metal plating, moving engine parts, break lining wear	Acute toxicity at high concentrations or continuous exposure	URS, [5, 6, 12, 14]
Lead	Bearing and tire wear, lubricating oil, and grease	Acute toxicity	URS, [5, 6, 12, 14]
Copper	Metal plating, bearing and bushing wear, moving engine parts, break lining wear, fungicides and insecticides, and maintenance operations	Acute toxicity at high concentrations or continuous exposure	URS, [5, 6, 12, 14]
Zinc	Tire wear, motor oil, and bridge metal galvanization	Acute toxicity at high concentrations or continuous exposure	URS, [5, 6, 12, 14]
Nickel	Diesel fuel and gasoline exhaust, lubricating oil, metal plating, bushing wear, break lining wear, and asphalt paving	Cancer and pulmonary diseases	URS, [5, 6, 12, 14]
Manganese	Moving engine parts	Acute toxicity at high concentrations or continuous exposure	URS, [5, 6, 12, 14]
Iron	Galvanization of automobile and bridge parts and moving engine parts	Nausea and intestinal problems at high doses	URS, [5, 6, 12, 14]
Bromide	Exhausts	Disruption of nervous system	URS, [5, 6, 12, 14]
Chloride	Deicing salts	Hyperchloremia at high concentrations and possible kidney failure with prolonged exposure	URS, [5, 6, 12, 14]
Cyanide	Anti-cake compounds used to keep deicing salts granular	Acute toxicity	URS, [5, 6, 12, 14]
Nitrogen	Atmosphere and fertilizers	Eutrophication of water bodies	URS, [5, 6, 12, 14]
Phosphorus	Atmosphere and fertilizers	Eutrophication of water bodies	URS, [5, 6, 12, 14]

Modeling/Predicting the Impact of Stormwater Pollutants

Using mathematical models to predict the loads and impacts of stormwater pollutants can be a useful approach. Models can afford important insights to engineers and decision-makers about potential effects on water quality. In such cases, using theoretical modeling approaches to calculate the predicted environmental concentration (PEC) and the predicted no effect concentration (PNEC) of pollutants can be helpful.[30–33]

When it comes to water resources management or decision-making related to stormwater treatment, models can be very helpful in predicting the need for such measures when monitoring data are lacking. The fact that monitoring results can be below the detection limit due to factors such as dilution can be frustrating and can lead to poor decision-making.[17,34] Assisting models work by linking characteristics such as pollutant concentrations and site specifications to predict future impacts. Site-specific variables, such as traffic flows, rainfall totals, rainfall intensity, rainfall duration, antecedent dry periods, drainage area, and land use, can be considered in theoretical models, which can result in more accurate predictions and better-informed decisions.[17] Several reports exist on the effectiveness of modeling approaches in evaluating stormwater impacts on receiving waters.[35,36] Theoretical modeling may simplify and rely on assumptions based on previous literature data of other model parameters, which increases the level of uncertainty.[17] One major challenge for modeling is the prediction of the first flush of stormwater runoff, which is a variable quantity for each rainfall event, even for the same location. Thus, the use of a generalized equation that relies on climate, rainfall, and runoff characteristics may not be sufficient for predicting the first flush loading.[17] Moreover, previous evaluations have been made for stormwater models and their capacity to predict water quality parameters and flow effects from best management practices application and conclusions, which were made on the need for continuous improvement efforts for such models.[36]

The uncertainty in prediction models can be a consequence of process variability. Such variability is not adequately considered in uncertainty assessment of stormwater quality model outcomes.[37] Therefore, this practice can cause poor management and planning decisions for stormwater mitigation strategies due to inaccurate interpretation of model outcomes.[37] According to previous reports, there are two specific types of uncertainties that have substantial influence on the interpretation of stormwater quality model outcomes: (1) uncertainty inherent to pollutant buildup and (2) wash-off processes and uncertainty in process modeling.[37] If process variability characteristics are included with the model, the uncertainty in stormwater quality modeling due to inherent process variability can be quantified and measured. Therefore, a careful understanding is required of buildup and wash-off variability and the consideration of process uncertainty associated with stormwater quality predictions.[37]

HIGHWAY STORMWATER RUNOFF AND BRIDGE DECK STORMWATER RUNOFF

Water Quality of Highway and Bridge Deck Stormwater Runoff

It is well established that both highway runoff and bridge deck runoff can transport various chemicals of urgent concern to freshwater bodies that can impact human and aquatic ecosystems with exposure. This implies that the constituents of runoff from highways and bridge decks are essentially similar since the sources are the same.[6] However, in a study on characterization of stormwater runoff from bridge decks in eastern Massachusetts, it was shown that the concentrations of suspended sediment in the bridge deck runoff samples were considerably higher than those in the monitored highways. This was attributed to the collection of runoff downstream from catch basins on highways that causes a portion of suspended sediment to be reduced in highway runoff, specifically coarse sediments,[9] while runoff from bridge scuppers did not pass through a similar mechanism for reduction of suspended sediments within the stormwater flow train.[9]

It was also shown that concentrations of all particle-size fractions of suspended sediments in untreated bridge deck runoff were collectively higher, and in particular for the two larger sediment-size fractions of 0.0625–0.25 mm and ≥0.25 mm compared to suspended sediment concentrations in highways with catch-basin treatment. Not only were suspended sediments shown to be substantially higher in concentration in bridge deck runoff compared to highway, but also other constituents, including total nitrogen and total phosphorus, were higher.[9] Overall, the variation in concentrations of runoff constituents between bridge deck runoff and highway runoff can be clearly related to variable factors including intensity of emissions from vehicles, spills and leaks and the lack of bridge deck runoff conveyance, collection and/or filtration mechanisms compared to highway runoff.

Water Quality of Bridge Deck Stormwater Runoff from Scupper Drains

Scupper drains are vertical openings designed in bridge deck through which stormwater runoff can flow off the bridge deck into beneath water body.[6,14,26] Therefore, the significance of scupper drains is that these structures are direct points of discharge of the bridge deck runoff without any prior pretreatment. Thus, the concentrations of major runoff constituents such as suspended solids and nutrients are detected to be higher in bridge deck runoff from scupper drains compared to highways.[9] Since it is expected that the concentration of suspended solids from the discharge of scupper drains can be high, it is also expected that the first flush concentration of other organic and inorganic constituents that can adsorb to suspended matters can be introduced into water bodies.

FACTORS AFFECTING WATER QUALITY OF BRIDGE DECK STORMWATER RUNOFF AND ITS IMPACT ON RECEIVING WATERS

Factors Affecting Water Quality of Bridge Deck Stormwater Runoff

Rainfall Events and Rainfall Intensities

It is recognized that various types of contaminants are airborne and can be emitted from various sources, such as industrial emissions and traffic. These contaminants can settle with rainfall onto water bodies and surfaces, creating toxic conditions to aquatic and terrestrial organisms.[38–40] Moreover, settling airborne particulate matters can adsorb and catalyze the transformation of different contaminants, such as volatile organic compounds and inorganic oxides into other types of contaminants of even more toxic nature.[38] When rainfall is of high intensity, it can mobilize and carry even more pollutants and solids from the bridge deck surface, leading to an increased pollutant concentration in runoff.[10]

The rainfall intensity can affect water quality. For instance, loading of solids can increase with increased water flow and rain intensity.[8] Variables such as maximum rain intensity can significantly affect the concentration of different runoff contaminants.[8] Moreover, contaminant concentration in bridge deck runoff can dramatically decrease after the first flush, which is defined as the initial 30 to 40 min of the rainfall event, or the first 5 to 10 mm of rain.[10]

Atmospheric depositions are a key factor for contamination in bridge deck runoff. Elements, such as copper, cadmium, phosphorus, and nitrogen, originate predominantly from rainfall as a major source of their existence in runoff. The concentration of dissolved nitrate and dissolved ammonia in rainfall was observed to be high, and rainfall is a significant source of such pollutants in bridge deck runoff.[29] It can be inferred that rainfall has a great influence on bridge deck runoff quality, which can vary significantly depending on the rain season and antecedent dry duration.

Location of Bridges

It is projected that the nature of discharged or emitted pollutants depends on the characteristics of the location and the sources of pollutants. In urbanized regions, it is expected that pollution from residential, commercial, and industrial locations will be high. In rural regions, it is expected that agricultural runoff would be high or that urban pollution would be low. The nature of contaminants will vary depending on their sources. Bridge sites that are located close to areas of the landscape that produce large amounts of pollutants (i.e., stormwater hotspots) are expected to convey larger amounts of contaminants than bridge sites away from stormwater hotspots. Examples of stormwater hotspots can include gas / fueling stations, vehicle maintenance areas, vehicle washing / steam cleaning, auto recycling facilities, outdoor material storage areas, loading and transfer areas, landfills, construction sites, industrial sites, and industrial rooftops.[41]

Size (Number of Lanes) of Bridge

The number of lanes of a bridge is often correlated with traffic count values, such as average daily traffic (ADT). Therefore, runoff from bridges with more lanes is expected to be more contaminated by traffic-related pollutants than bridges with fewer lanes.[42] Because more materials are required to construct bigger bridges, they also present an increased risk of contamination from construction-related materials.[43]

Bridge Deck Materials

PAHs are the most common toxic pollutants produced from asphalt sealants, which are commonly used to coat driveways.[28,44] Asphalt is also known to be a composite of other chemicals of petrogenic and pyrogenic character that are formed of hydrocarbons and their combustion products.[28] Asphalt can be degraded into fine particles of different sizes in the micron range, and can be carried by stormwater runoff to the closest receiving water body(s).

Concrete is a synthesized mixture of cement, water, and aggregate material. There are types of concrete that are made by heating a mixture of limestone and clay containing oxides of calcium, aluminum, silicon, and other heavy metals. It is possible that minerals can leach from concrete materials when stormwater runoff is conveyed by concrete infrastructure. Rainwater quality, pertinent to conductivity, total dissolved solids (TDS), and COD, were observed to change after passing through concrete pipes and down concrete gutters, which is consistent with a likely dissolution of cement and leaching of minerals in concrete materials.[45]

Age of Bridges

Porous friction courses in old bridge deck pavement material can entrap more accumulated pollutants from various sources, such as traffic and atmospheric deposition during antecedent dry periods, than new bridge deck pavement material.[46] However, it is also anticipated that fresh, newly paved bridge deck with asphalt or concrete can be a source of more leaching of the deck material's associated chemicals than old bridges.[28,44,45]

Traffic Volume

There is a correlation between traffic volume and contamination of highway and bridge deck runoff. Traffic volume is also a predictor at sampling sites for COD, heavy metals, and PAHs in water.[8,47] Where traffic volume is high, it is anticipated that contaminant loading relative to the aforementioned parameters will increase. Traffic can be considered an influential factor affecting water quality due to emission of organic contaminants (e.g., PAHs) and heavy metals that can deposit on roads and be washed off by runoff water.[48,49] If PMs are found, it is anticipated that residence of organic contaminants, heavy metals, and inorganic oxides will increase in ambient air and on the ground due to adsorption of these contaminants on PMs. Road traffic is found to contribute to the presence of PAHs and heavy metals, such as zinc, copper, lead, and cadmium, in stormwater runoff.[50–52]

In 1990, the Federal Highway Administration (FHWA) set a threshold of 30,000 vehicles per day (VPD), above which little noticeable impact can be observed from vehicles exhaust, leaks, and spills to surrounding water bodies and below which no noticeable impact can be observed, while a threshold of 180,000 VPD is the number at and above which significant impact could be observed to surrounding water bodies. (See references 4–8, 10, 14, 21, and 53)

Based on observations from previous reports, two classifications of traffic volume—average annual daily traffic (AADT) and VPD—are important with respect to the impact of traffic on surrounding and receiving water bodies. One classification is given to highways and bridges with low congestion and the second is given to congested roads and bridges. Congested roads and bridges would have lower VPD count compared to less congested highways and bridges. Congested roads and bridges should have lower VPD count with respect to water quality due to the longer waiting times of vehicles that results in high loads of pollutants per traffic volume ratio.

The literature reports great variation in AADT numbers with respect to impacts on water quality. Some observed impacts at AADT values above 750 VPD, while others observed no impacts until above 30,000 VPD (See references 4, 8, 14, 21, 25, and 54–57). Malina et al. (2005)

reported the average concentrations of constituents in bridge deck runoff from three locations in Texas: Austin (Central), Lubbock (High Plains), and Houston (Coastal Zone). The ADT count was 58,000 VPD at the Loop 360 bridge crossing Barton Creek in Austin, 10,000 VPD at the Loop 289 bridge over the North Fork Double Mountain Fork Brazos River in Lubbock, and 15,000 VPD at the bridge on FM 528 crossing Clear Creek near Friendswood, in the Houston area. Although the Loop 289 bridge had the lowest ADT (10,000 VPD) among the three sites, its runoff showed higher concentrations of total copper (18.70 ppb), total and dissolved lead (14.30 and 1.80 ppb), COD (71.70 ppm), total Kjeldahl nitrogen (TKN) (3.40 ppm), total and dissolved phosphorus (1.10 and 0.90 ppm), and volatile suspended solids (24.30 ppm), while the other two sites showed observable concentrations for the same parameters in addition to showing higher concentrations for several other parameters.[8]

Liu et al. (2017) classified traffic volume into high traffic volume (>5,000 VPD) and low traffic volume (DTV <5,000 VPD). Based on this classification, they observed that production of four-ring PAHs was higher at high traffic volume, and production of two- to three-ring PAHs was higher at low traffic volume in road dust with concentrations ranging from 0.6 to 1.1 mg/kg. This shows that PAHs can be significantly present in road dust at VPD below and above 5,000.[54]

Karlsson et al. (2010) studied water quality from stormwater ponds and found no correlation between increased traffic volume and toxicity, as it was observed that toxicity at a pond site with the lowest AADT (4,700 VPD) was higher than two other sites of AADT 71,000 and 130,000 VPD, based on the presence of toxic organic content. The lowest AADT site also showed varying concentrations of heavy metals, ranging from 0.02 ppb for cadmium, 0.50 ppb for lead, and up to around 20.00 ppb for zinc.[55]

Grant et al. (2003) and Lygren et al. (1984) detected heavy metals, including lead and PAHs, levels in road dust highways of ADT 8,000 VPD.[21,56] Taylor et al. (2014) listed median concentrations of various water constituents with significant concentrations in ppm, including TSS, nutrients, and coliforms in highway runoff with AADT below 25,000 VPD. They also listed several traffic characteristics other than vehicle counts that can greatly impact water quality, including speed, vehicular mix (cars/ trucks), congestion factor, and State regulations controlling exhaust emissions, based on earlier reports by the FHWA.[4,14,57]

Kayhanian et al. (2003) stated, "Due to large variations in urban AADT values (AADT >30,000 VPD), a single classification for urban highways was found to be impractical to properly assess any correlations that may exist between pollutant concentrations and AADT."[58] This statement agrees with the aforementioned reports that showed no clear correlation between water quality and AADT values. The authors elaborated, "Average concentrations of some pollutants (COD, TSS, TDS, turbidity, NH_3, and diazinon) were found to be higher in runoff from nonurban highways than the runoff from urban highways, suggesting sources other than the transportation-related activities."[58] Table 2 shows different AADT values in VPD from the literature.

Table 2. Different AADT values in VPD from the literature and the impact on water quality.

AADT Values in VPD and Impact on Water Quality		
VPD	**Impact on Water Quality**	**Ref.**
<10,000	Impact	[8]
<5,000	Impact	[54]
4,700	Impact	[55]
8,000	Impact	[21]
8,000	Impact	[56]
<25,000	Impact	[4]
<30,000	No impact	[14, 53]
>30,000	Impact	[14, 53]
≥180,000	High impact	[14, 53]
≥3,000	Impact	[52]
≤750	No impact	[52]
≥5,000	Impact	[49]

FACTORS AFFECTING IMPACT ON RECEIVING WATERS

Water Habitat

It is crucial to assess water quality at bridge sites that drain runoff water into sensitive aquatic habitats. Assessing the quality of water at these sites is essential to determine the potential hazardous runoff impact on aquatic biota. If runoff water at these sites is contaminated with heavy metals, PAHs, and other contaminants such as solids and nutrients, this could have hazardous effects on aquatic life, whether through instant or bioaccumulative effects. It is widely known that heavy metals and various forms of organic contaminants can have toxic effects on aquatic life. Contaminants could be airborne and can settle with precipitation onto water bodies and surfaces creating nuisance and toxic conditions to aquatic and terrestrial creatures.[38–40] Moreover, particulate matter can catalyze the transformation of different contaminants, such as volatile organic compounds and inorganic oxides, into other types of contaminants of an even more toxic nature that can substantially impact the surrounding ecosystem.[38]

Stream Size and Flow

Wide and/or rapidly flowing streams are known to have a higher capacity of cleansing by diluting pollutants and reducing their induced accumulative effects. Slower flowing streams have less cleansing capacity and, thus, are more prone to the accumulation effect from different pollutants.[59] Thus, size and flow speed of streams are important site characteristics to be considered in predicting the bridge runoff impact on receiving streams.

The United Nations Educational, Scientific and Cultural Organization (UNESCO), the United Nations Environment Programme (UNEP) and the World Health Organization (WHO) jointly provided an integrated stream classification that adds stream discharge, stream width, and stream order with the drainage area to classify streams (table 3).[60] It is suggested that this classification be used as an integrated classification for stream selection in this GDOT project.

Table 3. River size classification based on average discharge (in cubic feet per second), drainage area (in square miles), river width (in feet), and order.[60]

Classification	Average Discharge (cfs)	Drainage Area (mi^2)	River Width (ft)	Stream Order
Medium Rivers	$3.5\times10^3 - 3.5\times10^4$	$3.9\times10^3 - 3.9\times10^4$	656–2625	6–9
Small Rivers	$3.5\times10^2 - 3.5\times10^3$	$3.9\times10^2 - 3.9\times10^3$	131–656	4–7
Streams	$35.3 - 3.5\times10^2$	$38.6 - 3.9\times10^2$	26–131	3–6

WATER QUALITY PARAMETERS OF BRIDGE DECK STORMWATER RUNOFF

Heavy Metals

Heavy metals were found extensively in the literature to exist in bridge deck and highway stormwater runoff. The sources of heavy metals in bridge deck runoff are reported to be primarily transportation-related and from galvanization of bridge beams. (See references 4–6, 10, 12, 15, and 29) In addition, heavy metals were observed to be released from roadway construction and maintenance activities, roof surfaces, car/vehicle washing, retail/commercial/trading estates, road gully chambers, building misconnections, surface water sewers that carry rainwater directly to local rivers, streams, combined sewer system, cross-connections, and from atmospheric depositions by rainfall.[29] In previous studies, water samples collected from bridges with and without scupper drains were tested.

The concentrations of heavy metals were higher from bridges with scupper drains than those without, specifically in sediments. Thus, reduction in use of scupper drains in new bridges was recommended. Instead, runoff could be directed to roadsides in order to encourage percolation for maximum removal of heavy metals by overland flow detention before runoff reaches water bodies. It was also observed that runoff samples obtained from scupper drains were abundant in particulate matter and only 12 percent of total metals constituents were in dissolved form.[29] Therefore, since most of the heavy metals content exists in sediments in bridge runoff, it was recommended that use of retention/detention ponds can be efficient in removing large percentages of heavy metals from bridge runoff through settling.[29]

Many heavy metals are found to exist in stormwater runoff, such as zinc, copper, lead, cadmium, iron, nickel, chromium, aluminum, and arsenic. However, among these heavy metals, copper, zinc, and lead are most commonly detected.[6,61,62] Many of the heavy metals in stormwater runoff from bridges, such as lead and cadmium, are known to be of hazardous health concern. Table 4 shows the concentrations of the different heavy metals detected in bridge stormwater runoff from previous studies.

Polycyclic Aromatic Hydrocarbons

PAHs are semivolatile organic contaminants that are released from combustion processes. PAHs commonly exist in the atmosphere; thus, the major source of these contaminants is atmospheric depositions with precipitation.[3,12,14] PAHs can be adsorbed on particulate matters and thus can be transported for long distances with stormwater runoff. Transportation (vehicle) emissions are a major source of PAHs, among other pollutants such as VOCs and heavy metals. Although concentrations of PAHs were commonly found to be low, their accumulative and toxic effects on nature are of concern.[61]

Concentrations of PAHs were also found to be above the threshold level specified in guidelines.[52] PAHs were analyzed in stormwater runoff from urban areas in Shenzhen, China, and it was observed that heavy molecular weight PAHs are of high concentrations and can pose harmful effects to the aquatic environment. Wash-off samples were collected from a commercial area in Gold Coast, Australia, and the PAHs were analyzed. Concentrations of benzo[a]pyrene were found to range from 0.13 to 0.77 ppm for different wash-off events, which exceeded to a significant extent the drinking water guideline value of 10.000 ng/L provided by the National Health and Medical Research Council (NHMRC) in Australia, and the Arkansas Water Resource Center (AWRC).[52]

Table 4. Reported concentrations of heavy metals in bridge stormwater runoff.

Reference	Elements								Unit
	Zinc	Copper	Lead	Cadmium	Nickel	Iron	Chromium	Arsenic	
[10]	-	6.00–30.00	-	-	6.00–8.00	1,040.00–5,500.00	6.00–10.00	-	µg/L
[8]	128.00–167.00	16.00–19.00	5.00–14.00	-	-	-	-	-	µg/L
[49]	2,231.00		50.00	10.00	82.00	-	83.00	-	µg/L
[4]	190.00	42.00	44.00	-	-	-	-	-	µg/L
[11]	120.00	16.00	5.00	-	-	-	-	-	µg/L
[6]	555.00	195.00	103.00	-	-	-	-	-	µg/L
[7]	120.00	-	60.00	-	-	9,740.00	-	-	µg/L
[14]	360.00–760.00	130.00–270.00	51.00–160.00	1.00–3.00	15.00–36.00	-	8.00–40.00	-	µg/L
[5]	66.00	10.00	5.00	0.10	2.00	1,420.00	4.00	1.00	µg/L

- : No values found

Both toxicity and adsorption affinity of PAHs were found to vary between different groups of PAHs (by molecular weight). Highly carcinogenic PAHs with high molecular weight were observed to be concentrated in particle size fractions of <0.7 mm. However, significant concentrations of PAHs were also detected in particle sizes ranging from <2 to <500 µm.[37] Therefore, this information sheds light on the affinity of PAHs to adsorb to particles and establishes the context of PAH removal with particles as a stormwater mitigation measure. Table 5 shows average event mean concentrations of PAH values from the literature.

Table 5. Reported PAH event mean concentrations of bridge stormwater runoff values from the literature.

Reference	∑PAHs	Unit
[3]	<1.000 – >2.000	µg/L
[12]	0.015 – 0.500	µg/L
[14]	<0.050 – 0.850	µg/L

Oil and Grease

Oil and grease can exist in stormwater runoff from motor lubricants, spills and leaks, antifreeze and hydraulic fluids and they are commonly measured as collective hexane extractable components. Approximately 2,000 spills per year were recorded to take place on roadways in the U.S.[7] Although oil and grease in general are not classified as emerging priority pollutants, unlike heavy metals and PAHs, their existence can also be of concern to aquatic health due to the creation of nuisance conditions.

Oil and grease in significant quantities can reduce the penetration of oxygen into water bodies which may lead to proliferation of anaerobic conditions. Moreover, lubricating oil and grease can contain toxic lead and can contribute to increased heavy metals concentrations in sediment from bridge deck and highway runoff.[6,11,12] Concentrations of oil and grease can sometimes be significant. Kim et al. reported up to 29.40 mg/L at bridge sites.[63]

Vegetated swales at sides of highways can remove oil and grease in highway runoff. However, the absence of vegetated swales at bridges leads to the high probability of the presence of oil and grease from vehicle leaks and spills. If stormwater runoff from bridges can be directed to dry detention basins, bioretention systems, sand filters and catch basin inserts, this action can lead to significant reduction of oil and grease in runoff water before its discharge to water bodies.[3,4,12] Previous studies observed that most oil and grease concentrations exist where there are irregularities on the road surface.[4] Table 6 shows oil and grease concentration values from the literature (See references 3, 29, 63, and 64).

Table 6. Reported oil and grease concentrations of bridge stormwater runoff values from the literature.

Reference	Oil & grease	Unit
[3]	<0.50 – >1.00	mg/L
[65]	29.40	mg/L
[8]	4.40 – 4.80	mg/L
[64]	6.70	mg/L

Solids

Sources of solids in stormwater runoff are numerous, including atmospheric fallout, dust fall, anthropogenic wastes, rehabilitation work, construction and maintenance activities, and pavement wear.[17,29] Solids loading increases and decreases with hydrologic and climatic factors. It is anticipated that solids loading increases with elevated stream discharge, rainfall intensity, and antecedent dry periods and decreases with foregoing storm intensity. Both rainfall intensity and runoff volume can have a substantial effect on runoff constituent loadings. Storms of high intensity can transport 60 percent of the TSS load in the first 4 min, whereas storms of low intensity can transport about 15 percent of the TSS load. A similar pattern can be observed for loadings of other constituents with rainfall intensity.[29]

Of runoff pollutants measured as COD, 70 percent were found to be associated with suspended or settleable solid content in runoff, and thus can potentially be eliminated through sedimentation and settling.[29] Since more than half of suspended solids were found to be inorganic, a large portion of sediment and its accompanying pollutants in bridge runoff, e.g., heavy metals, can be removed via sedimentation in holding ponds, and an average of 85 percent TSS removal was observed to be achieved in holding ponds.[29] Periodic maintenance and cleaning of holding ponds and the disposal of sediment is highly recommended to retain the treatment process efficacy.

Solids are of significant environmental concern and can be linked to two types of hazardous effects. First, a high concentration of suspended solids can affect the diversity of aquatic organisms of critical importance to stream ecosystems.[6] Sediments are responsible for burying fish eggs and can disrupt the food chain.[26] The survival of benthic organisms was observed to consistently decrease due to the contact of the organisms with sediments that are washed to surface waters from highway stormwater runoff. Second, other pollutants can be conveyed and transported by sediments to other areas, making sediment a delivery mechanism for other pollutants and, thus, a cause for concern. Changes in TSS concentration are linked to storm characteristics. Rainfall intensity can mobilize and flush TSS from bridge decks, and can also cause surges of higher concentrations of other contaminants throughout a storm.[6] Table 7 shows solids concentration values from the literature (See references 4, 6, 10, 11, 29, 63, and 64).

Table 7. Reported solids concentrations of bridge stormwater runoff values from the literature.

Reference	TSS	TS	Unit
[10]	20.00–140.00	300.00–900.00	mg/L
[4]	120.00–178.00	-	mg/L
[8]	53.00–112.00	-	mg/L
[11]	22.00	-	mg/L
[6]	<100.00–>400.00	<100.00–>500.00	mg/L
[65]	155.00	-	mg/L
[64]	65.00	-	mg/L

- : No values found

Nutrients

Many nutrients that are utilized by bacteria, plants, and algae as an energy source are formed of mostly nitrogen and phosphorus. When nutrients increase in water bodies, they lead to proliferation of surface water plants that can consume large amounts of dissolved oxygen, leading to its depletion in subsurface water and, thus, the suffocation and death of various kinds of aquatic organisms, a phenomenon known as *eutrophication*. The primary sources behind nutrients presence in bridge deck stormwater runoff are atmospheric deposition and traffic; however, the contribution of nitrogen and phosphorus from traffic-related sources in runoff appears to be less significant than that from natural sources.[15,66] Nutrients are specified as total nitrogen (e.g., ammonia, organic nitrogen, nitrates and nitrites) and phosphorus, which is predominantly orthophosphates.

The major source of nutrients in bridge deck runoff is atmospheric depositions, while for roadways and highways, soil and agricultural runoff are other major sources. Thus, it is anticipated that nutrient loadings in bridge deck runoff would be generally lower. Concentrations of nutrients in highway runoff in urbanized regions are recorded to be generally lower than those in runoff in undeveloped lands.[29] However, TN and TP from a highway bridge deck (samples obtained at the edge-of-pavement) were found to be substantially larger than those from nearby highways (one rural and one urban).[15] This was caused by the frequent winter maintenance of bridge decks and by pollutant buildup along the curb wall. Nutrients, among other contaminants such as heavy metals, were frequently detected in bridge deck runoff, even more than organic compounds and oil and grease.[10]

A correlation could be proven between concentrations of nitrogen compounds in rainfall to nutrients in runoff by detecting nutrients in both rainfall and runoff. The concentration of nitrate in precipitation was said to account for 50–100 percent of nitrate concentration in bridge deck runoff. Moreover, up to 22 percent of the total phosphorus load observed in highway runoff was attributed directly to phosphorus content in rainfall.[29] Atmospheric chemical processes are known to result in the formation of acid rain, including nitrogen oxide and sulfur oxide compounds in the highway environment. Chemicals that are generated by combustion of fossil fuels by motor vehicles are oxidized into strong acids by ozone and hydrogen peroxide in the presence of water vapor and are finally deposited as sulfuric acid and nitric acid onto the roadway surface. These strong acids have the potential to impair runoff quality and negatively impact aquatic biota in receiving waters.[29] Table 8 shows nutrient concentration values from the literature.

Table 8. Reported nutrient concentrations of bridge stormwater runoff from the literature.

Ref.	TN*	TKN	Nitrate Nitrogen	Nitrite Nitrogen	AN	ON	TP	OP	Unit
[10]	-	0.20–0.23	0.20–9.20	0.02–0.50	-	-	0.10–0.64	-	mg/L
[4]	3.59	2.32	1.06	-	-	-	0.44	0.25	mg/L
[15]	1.18	0.89	0.26	-	0.14	0.78	0.36	0.03	mg/L
[11]	0.97	0.96	-	-	-	-	0.06	0.05	mg/L
[8]	-	1.00–3.40	0.30–1.40	-	-	-	0.10–1.10	0.00–0.90	mg/L

*Note: TN = Total Nitrogen, TKN = Total Kjeldahl Nitrogen, AN = Ammonia Nitrogen, ON = Organic Nitrogen, TP = Total Phosphorus and OP = Orthophosphorus.
- : No values found

pH, Conductivity, and Dissolved Oxygen

Other parameters give a direct indication of water quality, such as pH, conductivity and dissolved oxygen. pH is a measure of the acidic and alkaline magnitude of water through detection of proton mobility; the regular pH of fresh water is neutral to slight alkaline, and the optimum pH for aerobic bacteria is 6.00 to 9.00. The conductivity measurement is an indication of salts, minerals and other dissolved solids in water. The optimum DO concentration for fresh water is 9.00 mg/L. If DO values of fresh water are low, this can reflect an elevated chemical oxygen demand and increased organic contamination, and can be of great concern to the aquatic environment.[38]

Other Hazardous Constituents and Pathogens

Other constituents that are reported to exist in stormwater runoff include halogenated compounds such as polychlorinated biphenyls (PCBs), perfluoroalkyl substances and perfluorinated compounds, and microorganisms. (See references 5, 12, 14, 17, 61, 62, and 66–69). Polybrominated diphenyl ethers (PBDEs) are synthetic chemicals used as flame retardants in electronics, building materials, seat cushions, and clothing. PCBs are very stable chlorinated and nonflammable chemicals that were used as insulators and cooling compounds in electric equipment such as transformers, and have been used in other products like paint, inks, and pesticides. Although the use and application of such halogenated compounds were widely banned long ago, their fingerprints still exist dramatically in nature and water.[70] PCBs and PBDEs have similar chemical structures and very similar toxicity, which can cause problems in marine and freshwater fish ranging from neurotoxicity to hormone disruption.[70] Both PCBs and PBDEs are persistent, hydrophobic compounds that do not degrade or dissolve readily in water and tend to bioaccumulate in fatty tissues and have been detected in soil; air; water; sediment; and bodies of fish, wildlife, and people.[70]

Perfluorinated compounds (PFCs) were used as surface modifiers and surface-active agents for car wax, paint, paper, carpet, and cloth; as emulsifying aids for the polymerization of fluorine resin; and as aqueous-film-forming foams for fire-fighting devices for more than 50 years and were banned by the Montreal protocol due to their adverse effect on stratospheric ozone. However, the pervasive contamination by PFCs is still being detected. PFCs accumulate in wildlife even in the Arctic.[68] Among them, perfluorooctane sulfonate (PFOS) is known to be highly persistent, bioaccumulative, and toxic.

The concentrations of long-chain PFCs such as perfluorodecanoic acid (PFDA) increased markedly in the first flush of stormwater runoff from the Hayabuchi River basin in Japan, which is served by a separated sewerage system.[68] Perfluoroalkyl acids (PFAAs) in stormwater runoff from residential areas were found to be mainly derived from rainfall and ranged from 2.500 to 15.000 ng/L. Moreover, nonatmospheric sources at both industrial and commercial areas contributed to PFAAs in stormwater runoff.[67] In addition, PFAAs on the particles/debris in stormwater runoff were monitored and it was found that high-level PFOS existed on the particulate matter in runoff collected from both industrial and commercial areas; the levels were so high that it could not be explained by the solid-water partitioning and it ranged from 8.700 up to 590.000 ng/L (0.590 µg/L). PFOS on the particulate matter is suspected to have originated from industrial/commercial products, entering the waste stream as PFOS-containing particles.[67]

Microorganisms were detected in stormwater runoff and their concentrations from high-density residential areas were higher than those associated with low-density residential and commercial areas. The major sources of microorganisms in stormwater runoff were most likely the domestic animals and wildlife waste.[69] Concentrations of microorganisms were observed to be significantly affected by seasons. In one study, the lowest concentrations were observed during winter except for staphylococcus aureus, while fecal coliform had a much higher summer count than winter.[69] However, none of the studies of other contaminants and microorganisms referred specifically to bridge deck stormwater runoff.

TREATMENT OF BRIDGE DECK STORMWATER RUNOFF

Several approaches have been implemented for treatment of bridge deck runoff. These commonly include bioretention systems or rain gardens, infiltration systems, filtration techniques, biological treatment and finally disinfection. All these processes can be implemented for bridge deck runoff as well as highway runoff treatment. However, the greatest challenge for bridge deck runoff remains to be the runoff conveyance.

Bioretention

Bioretention media are constructed systems of vegetated shallow gaps that are employed to temporarily store stormwater before introducing stormwater to infiltration, evapotranspiration, or discharge via an underdrain or surface outlet structure. Bioretention systems can be designed to remove different pollutants by filtering stormwater through an engineered soil mix. Removal of contaminants occurs primarily through filtration, shallow sedimentation, sorption, and infiltration in bioretention systems. Several media such as grass buffer strips, organic or mulch layers, sand beds, ponding areas, planting soil, and plants are used for bioretention.[4,5,12] The velocity of stormwater runoff eventually decreases when passing through the bioretention system buffer strip, and consequently the stormwater distributes evenly along a ponding area. Over a period of days, the stored water in the bioretention area planting soil exfiltrates into underlying soils. Physicochemical and biodegradation mechanisms are other pathways of pollutant removal in bioretention systems that take place in planting media through adsorption and microbial action on dissolved pollutants. Bioretention systems can remove several contaminants such as suspended solids, heavy metals, oil and grease, nutrients, and bacteria and can reduce volume and peak flow.[4,12]

Infiltration

Site conditions, such as soil permeability and the potential of clogging, control the performance of infiltration systems.[71] The reasons for clogging of infiltration systems could be the incorrect particle size distribution, excess deposition of sediment, or organic matters that can form a clogging layer on the soil surface, and the rearrangement and settling of the infiltration medium due to the weight of water (hydraulic compaction).[71] The performance of infiltration-based techniques shows very promising ranges, which is not surprising given their observed interaction with different site conditions. The reductions in annual runoff ranged between 40 and 60 percent for swales and filter strips with incorporated infiltration component, 50–90 percent for infiltration trenches and basins, 40–80 percent for unlined bioretention systems (rain gardens), and

45–75 percent for porous pavements. Infiltration-based systems can be beneficial in restoring groundwater levels and base flows.[71]

Concerns were raised about the possibility of groundwater contamination by infiltration systems. However, the potential of groundwater contamination by infiltration output is generally low for most contaminants, except for the uncertainty associated with contamination by pathogens, salts, and thermal fluctuations.[71] Establishing the proper guidelines is necessary to safeguard the consistency of the developed infiltration-based devices with sustainable depths to the groundwater table. Moreover, prediction models capable of projecting potential impacts of infiltration systems on receiving waters, and particularly on base flow processes, are necessary.[71]

Filtration

Filtration is the trapping of runoff when it passes through the engineered media or existing soils where solids and particulate-bound pollutants are physically removed by the media. Dissolved pollutants can be eliminated by trapping in the filtering media if the media possesses adsorptive properties. Treated runoff can recharge groundwater supplies and can reduce surface flow volumes delivered to receiving streams.[4] The media filter design for stormwater is a composition of a two-compartment structure that includes a pretreatment settling basin and a filter bed filled with sand or other absorptive media. When stormwater flows into the first compartment, large particles settle while fine particles and other soluble pollutants are removed as stormwater flows through the filtering media in the second compartment.[12]

Various designs of sand filters include the surface sand filter, underground sand filter, perimeter sand filter, organic media filter, and multi-chambered treatment train (MCTT), which all operate using the same basic principles. Other modifications can be made to traditional sand filters, such as underground and perimeter filters to improve the pollutant removal capacity (e.g., organic media filter).[12]

The open shallow channel design of grassed swales with vegetation covering the side slopes and bottom allows for the collection and transportation of runoff to downstream discharge points. Grassed swales, which can be natural or constructed, are designed to treat runoff through channel vegetation filtration, subsoil matrix filtration, and/or infiltration into the underlying soils. Particulate pollutants (i.e., suspended solids and trace metals) can be trapped in grass swales. Grass swales can also promote infiltration and reduce the flow velocity of stormwater runoff. Vegetated swales can serve as part of a stormwater drainage system and can replace curbs, gutters, and storm sewer systems.[12] A grass drainage channel provides vegetative lining for filtration of runoff. Drainage ditches, roadside ditches, outlets for diversions, and channels at property boundaries are all typical uses.

Biological Treatment

Biological processes can take place in stormwater ponds, wetlands, and during bioretention, and they are a part of the stormwater control measures (SCMs).[4,12] Biosorption-activated media (BAM) can be used in biological treatment and can remove pollutants such as nutrients and heavy metals while taking up less area relative to other treatment options. BAM materials have the dual characteristic of sorption properties as well as suitable active sites for biological growth.[4] However, BAM have limited ability to handle large flows, and it was noted that bridge inlets do not intercept large flow rates. Researchers are looking into being able to raise the flow rate for bridge applications, as in retention areas like swales and pipe-in-pipe wet detention pond harvesting applications.[4]

Complex chemical and biological unit operations may be required to treat bridge runoff pollutants that are dissolved or readily changeable within the aqueous environment as a function of redox, pH, and available partitioning sites (i.e., solids load or media characteristics).[4] The processes of breaking down substances can be spontaneous, chemical, or biological.[3] Even very complicated and toxic molecules can be degraded biologically. In the initial degradation stages, the chemical degradation process is dominating. The biological process requires the substance to be diluted to nontoxic levels that can be consumable by the degrading organisms, which can take a long time if the substances are solid and poorly soluble. Knowledge of the specific substances' biological degradability is, therefore, important to be able to determine the extent of remediation measures. The degradation processes mainly apply to organic substances, while substances which contain toxic elements, such as phosphorus, heavy metals, and arsenic, can still be toxic when degraded and may even change from being toxic to plants to being toxic to animals.[3]

The degradation process is mainly governed by two factors: temperature and pH.[3] The temperature is the most important factor. This is especially true for extreme temperature changes, such as a fire event, where the chemical degradation process is accelerated. Normal temperature changes have an effect, as well, and the degradation is faster during the summer compared to the winter. In a lake, or stream, the difference between the water temperature on the surface and the bottom also affects the degradation rate. At low temperatures, the processes may come to a complete stop, especially the biological processes that are susceptible to low temperatures. In very acidic or basic environments, many stable substances break down, while the pH also affects the processes in a long-term perspective due to the naturally occurring variations. The degradability of a certain substance in water is classified into three categories: readily biodegradable,

biodegradable, and persistent.[3] This classification is mainly designed for treatment plants and for discharges into the nature, where the degradation conditions are less favorable.

Disinfection

Disinfection is applied as the final treatment step following biological treatment, and is necessary to safeguard biologically treated water from microorganisms before discharging treated stormwater to water bodies.[72,73] Microorganisms, including both indicator and pathogenic species, can be removed from runoff water using filtration and chemical removal systems. However, the size of many bacteria and viruses is so small that physical straining cannot be the only removal mechanism. Cyst species, however, are larger and can be removed by physical means. Many bacteria and viruses are removed during sedimentation and filtration due to their surface charges and/or biofilms and exopolymers, such as carbohydrates, polysaccharides, proteins, and nucleic acids, which enable their adsorption to solids that are settleable and/or filterable, or to the filter media itself.[74]

Filtration and sedimentation do not permanently destroy the microorganisms, although they could be destroyed in the media after entrapment through natural predation or desiccation. Desiccation is unlikely in most stormwater filters since these media rarely dry out between storms.[74] The engineering solution to reducing/removing the activity of pathogenic species is disinfection using one of three common methods: chlorine, ozone, or ultraviolet (UV) disinfection.[74] Each of these disinfection techniques has been successfully applied to stormwater runoff and combined sewer overflows.[74]

STOCHASTIC EMPIRICAL LOADING AND DILUTION MODEL

In the absence of field data and the absence of knowledge of concentrations of pollutants that can create harmful impacts, modeling tools can be beneficial in order to assist in the consideration of potential future impacts. With an accurate modeling tool available, resource agencies and government branches can precisely anticipate the potential impacts associated with bridge deck runoff from scuppers on receiving water bodies and sensitive species. The development of the Stochastic Empirical Loading and Dilution Model was a collaboration between the U.S. Geological Survey (USGS) and the FHWA to redesign the FHWA's 1990 highway-runoff quality planning model. Its purpose is to track and reveal the changes in runoff quality and to shed light on the importance of preserving upstream receiving-water quality. The model also has the capacity to project and evaluate potential impacts from runoff on receiving waters. Thus, both entities worked on developing SELDM as a database application so that researchers and other users can efficiently perform runoff simulations.[75]

SELDM is a simulation software that assesses different parameters, such as storm flows, concentrations, and loads, and can provide risk analysis of surpassing water-quality criteria in the presence and absence of user-defined best management practices (BMPs). Using several input parameters, SELDM quantifies annual runoff loads, in addition to annual lake-load analyses. The model incorporates national data sets for water quality, precipitation, stream flow, runoff coefficients, and background water quality. Therefore, users can define the set of data that represents their selected site(s). The software approaches the impacts of different factors, such as precipitation characteristics, stream flow, estimated runoff quantity and quality, and BMPs, via the probability distribution of receiving-water concentrations using the Monte Carlo methods.[75]

SELDM can calculate critical parameters, such as runoff dilution in receiving waters, which include rivers and streams, downstream event mean concentrations, and annual average lake concentrations. Results can then be ranked and plotting positions can be calculated to indicate the level of risk of the adverse effects caused by runoff concentrations, flows, and loads on receiving waters by storm event and by year. SELDM uses site characteristics and representative statistical input values for each hydrologic variable to calibrate; thus, it is different from other deterministic hydrologic models in that it does not require changing of the input variables to match historical record. Thus, SELDM is an empirical model based on input data and statistics rather than traditional theoretical equations. SELDM can also be employed to model runoff from different land covers and land use features by means of the site definition parameters, as long as characteristic water quality and impervious-fraction data are available.[76]

The software identifies each storm using sequence numbers and annual-load for a specific year, and it can integrate several events into one annual-load set if the time between event midpoints of a series of simulated events exceeds a year. SELDM generates each storm randomly; there is no serial correlation, and the order of storms does not reflect seasonal patterns. The annual-load accounting years are random collections of events that are generated with sums of inter-event times less than or equal to a year, and which can be used to generate annual site flows and loads for the total maximum daily load (TMDL) analyses. The population of annual runoff and BMP discharge loads from the selected site indicates the potential contribution from the site, the potential for reducing loads to meet any proposed load allocations, and the uncertainty of such estimates.[77]

SELDM was used in a study to project long-term hazards of TP in highway runoff and to estimate the potential control measures of stormwater and the BMPs to alleviate any risks. In this study, statistics were obtained from the USGS study data in North Carolina. The simulation study was conducted for a 30-year projection, aiming to evaluate the potential long-term impacts of highway runoff. The authors used datasets from 15 selected sites and selected 4 sites for simulating long-term populations of highway bridge runoff, as well. They concluded that the simulated TP population from all 15 sites was a strong indication and a convenient alternative that can be used to estimate the potential hazards of exceeded concentrations in selected sites.[78,79]

CHAPTER 3. MATERIALS AND METHODS

All water samples were collected from scupper drains of bridge sites and the rivers/streams these bridges cross. Different water quality parameters were analyzed that included the following:

- Heavy metals (Pb, Zn, and Cu; *Standard Method* EPA-NERL: 200.9).
- Polycyclic aromatic hydrocarbons (*Standard Method* EPA-NERL: 525.2).
- Chemical oxygen demand (*Standard Method* Hach 8000 [USEPA 5220 D]).
- Nitrogen
- Total nitrogen.
- Total Kjeldahl nitrogen.
- Ammonia nitrogen (NH_3-N; *Standard Method* Hach 10031).
- Nitrate plus nitrite (NO_3^--N + NO_2^--N ; *Standard Method* Hach 102421 [USEPA 40 CFR part 136]).
- Nitrate (NO_3^--N; *Standard Method* Hach 8192).
- Phosphorus (PO_4^{3-}-P; *Standard Method* Hach 8190 [USEPA 4500-P E]).
- Solids (*Standard Method* EPA 1684).
- Oil and grease (*Standard Method* EPA 1664A).
- pH (*Standard Method* EPA 150.2).
- Dissolved oxygen (*Standard Method* EPA-NERL: 360.1).
- Conductivity (*Standard Method* EPA D1125-14 A).

SAMPLING

Sampling was performed using both a Hach Sigma SD900 Portable Sampler and a Teledyne ISCO 6712 Portable Sampler to analyze water collected directly from bridge scupper drains instantly or overnight. Over 80 samples were collected from all six study sites, covering a period of one (1) seasonal year (Spring 2019 to Spring 2020), Table 9 show the number of samples. A total sample volume of 500 mL to 1 liter was taken for analysis, at different sampling dates and time intervals of the same day. Instream sampling was done approximately under the bridge edge at midstream for both upstream and downstream sampling.

Table 9. Number of samples for each parameter and total number of samples.

Parameter	Number of Samples
Heavy Metals	64
PAHs	58
COD	81
Nitrogen	76
Phosphorus	77
Solids	73
Oil and Grease	69
pH, DO, and Conductivity	56
Total	81

Sampling was completed at six selected bridge sites, of which four were located in Southeast Georgia: (1) Site SR 24 on the Ogeechee River near Oliver; (2) Site Rocky Ford (RF) on the Ogeechee River on Rocky Ford Road; (3) Site US 80 on the Ogeechee River near Eden; and (4) Site SR 297 on the Ohoopee River near Swainsboro. The remaining two study sites were located in North Georgia: (1) Site US 255 on the Chattahoochee River near Leaf; and (2) Site SR 197 on the Soque River near Clarkesville. The study sites were selected based on the specific criteria to be crossing major streams, to have a USGS gage station for stream discharge and precipitation information, to have access for on-site sampling equipment installation, and to have circular scupper drains. The major land use for the six study sites' watershed is mainly agriculture. Figure 1 shows the locations of all six bridge sites relative to GSU using Google Maps. Table 10 shows the names, descriptions, and coordinates of the six sites of the study.

Figure 1. Map. All six bridge study sites and their locations using Google Maps (North Georgia sites marked in blue and Southeast Georgia sites marked in black; inset shows the overall locations within the State of Georgia).

Table 10. Six bridge study sites with their names, descriptions, and location coordinates.

Bridge #	Bridge Site Name	Description	Coordinates
1	SR 24	Southeast SR 24 crossing Ogeechee River	North (N) 32.49481, West (W) −81.55564
2	RF	Southeast Rocky Ford crossing Ogeechee River	N 32.649 W −81.841
3	US 80	Southeast US 80 crossing Ogeechee River	N 32.19169 W −81.41518
4	SR 297	Southeast SR 297 crossing Ohoopee River	N 32.44058056 W −82.3819611
5	US 255	North US 255 crossing Chattahoochee River	N 34.627583 W −83.642025
6	SR 197	North SR 197, crossing Soque River	N 34.618839 W −83.528968

Figure 2, Figure 3, Figure 4, Figure 5, Figure 6, and Figure 7 show Google Maps pictures of the six bridge sites with their vertical and horizontal views and the scupper drain images.

Figure 2. Photos. Bridge Site SR 24 from Google Maps showing vertical and horizontal views and bridge deck scupper drains.

Figure 3. Photos. Bridge Site RF from Google Maps showing vertical and horizontal views and bridge deck scupper drains.

Figure 4. Photos. Bridge Site US 80 from Google Maps showing vertical and horizontal views and bridge deck scupper drains.

Figure 5. Photos. Bridge Site SR 297 from Google Maps showing vertical and horizontal views and bridge deck scupper drains.

Figure 6. Photos. Bridge Site US 255 from Google Maps showing vertical and horizontal views and bridge deck scupper drains.

Figure 7. Photos. Bridge Site SR 197 from Google Maps showing vertical and horizontal views and bridge deck scupper drains.

Portable autosamplers were used to obtain scupper drain runoff water either via instant sampling (i.e., grab sample) or at time interval sampling (i.e., programmed sampling). Figure 8 shows the two autosamplers used to obtain scupper drain runoff samples: the Hach Sigma SD900 Portable Sampler and the Teledyne ISCO 6712 Portable Sampler.

Figure 8. Photos. Hach Sigma SD900 Portable Sampler (left) and Teledyne ISCO 6712 Portable Sampler (right).

Autosampling provided efficient sampling of accurate volumes and at accurate time intervals for subsequent sampling. The autosamplers were connected to scupper drains through an on-site sampling apparatus with a polyvinyl chloride (PVC) piping setup, which was fabricated and installed on each of six bridge sites. A clear PVC tubing was extended from each autosampler into an outlet at the end of the horizontal pipe of the on-site sampling apparatus. Figure 9 shows the on-site sampling apparatus for scupper drain sampling.

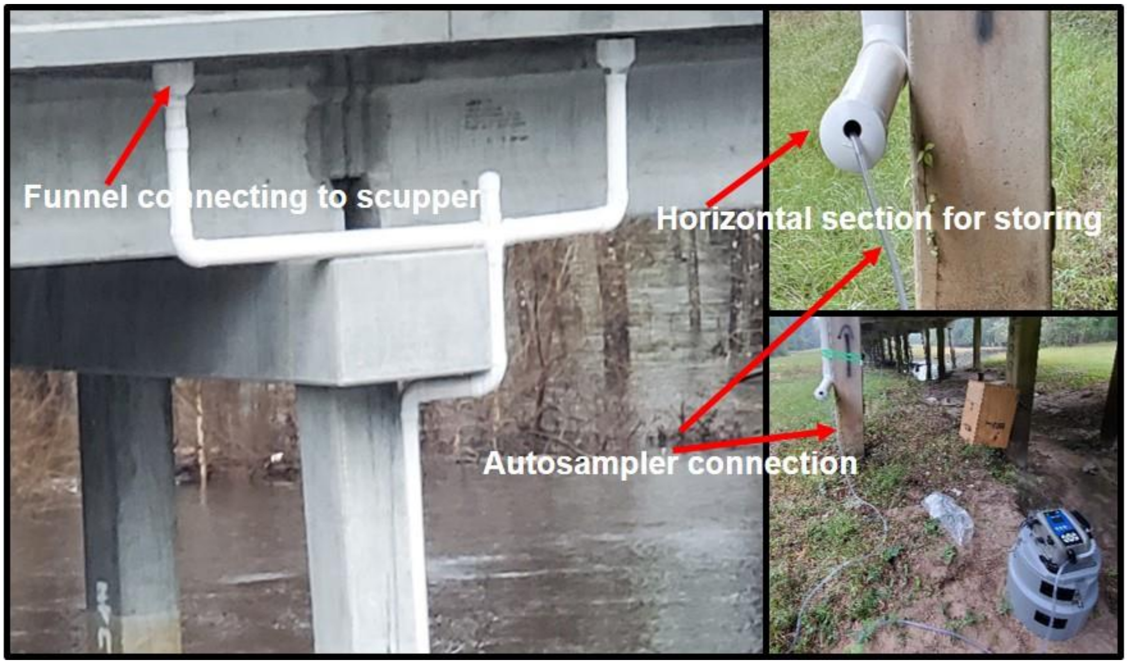

Figure 9. Photos. On-site sampling apparatus setup for scupper drain sampling.

Instream sampling was performed using manual pole sampling under the edge of each bridge, and an upstream sample and a downstream sample were taken at each bridge site.

Figure 10 shows the pole sampling tool used for instream sampling. The pole sampler arm can be extended up to 25 ft in order to obtain instream samples at the proper instream sampling distance from the boundary below the bridge edge, and at the end of the pole arm a 1 L PVC bottle was fixed on a rotating stainless steel cap. Instream sampling during or after scupper drain sample collection during a rainfall event was performed to investigate the impact of scupper drain runoff on the downstream water quality relative to the upstream water quality.

Figure 10. Photo. Pole sampling tool used for instream sampling.

PARAMETERS

Heavy Metals

Three metals of significant concern—Pb, Zn, and Cu— were analyzed in scupper drain samples, and upstream and downstream instream samples. Heavy metals were analyzed by direct injection to a graphite furnace, the Shimadzu AA-7000 Atomic Absorption Spectrophotometer (figure 11) in the Water and Environmental Research Lab at GSU. In each water sample, total and dissolved metals were analyzed. Samples for dissolved metals analysis were prepared by filtering off the obtained water samples using 0.45 µm Pall membrane filters on a vacuum filtration system. Concentration values were directly obtained from the machine output in parts per billion (ppb). A three-point calibration curve was developed using 1000.00 mg/L of Hach standard solution for each element to quantify each element in the samples.

Figure 11. Photo. Shimadzu AA-7000 graphite furnace atomic absorption spectroscopy.

PAHs

Eighteen PAHs were investigated: acenaphthene, acenaphthylene, anthracene, benz[a]anthracene, benzo[b]fluoranthene, benzo[k]fluoranthene, benzo[ghi]perylene, benzo[a]pyrene, chrysene, dibenz[a,h]anthracene, fluoranthene, fluorene, indeno[1,2,3-cd]pyrene, 1-methylnaphthalene, 2-methylnaphthalene, naphthalene, phenanthrene, and pyrene. Water samples were analyzed using a Shimadzu GCMS-QP2010 SE Gas Chromatograph Mass Spectrometer (GCMS) (figure 12) in the WERL at GSU. Prior to analysis, samples were extracted using Sigma GC-grade dichloromethane of 99.9 percent purity by adding dichloromethane to the sample in a 1:1 (25 mL of sample plus 25 mL of dichloromethane) ratio in a 250-mL separatory funnel and shaking for 10 seconds, leaving the sample to settle, and then repeating the extraction process for the oil phase for two more rounds. Roughly 1 mg of calcium chloride was added to each funnel after shaking to enhance separation of the oil phase from the aqueous phase each time. After samples were extracted, around 25 mg of sodium sulfate anhydrous was added to the extracted samples for extra dehydration. Samples were then concentrated from 25 mL to around 5 mL using a Hydrion Scientific Instrument Model R-302 rotary evaporator, and 1 mL was then taken for injection to the GCMS. Three-point calibration curves were obtained using a collective of the 18 compounds' Sigma standard to quantify possible PAHs.

The column oven temperature gradient was 90°C and held for 2 min, then up to 320°C at 5°C/min, and held for 12 min. The vaporizing chamber temperature was 300°C. The inlet mode was splitless and the carrier gas was ultra high purity grade helium. The injection pressure was 21.75 psi (1.5 min) and the constant linear velocity was 43.7 cm/sec. The purge flow was 3 mL/min and the injection volume was 1 µL. The mass spectroscopy interface temperature was 300°C and the ion source temperature was 230°C. The data sampling time was 60 min, the m/z range was 45–450, and the measurement mode was scan.

Figure 12. Photo. Shimadzu GCMS-QP2010 SE Gas Chromatograph Mass Spectrometer.

COD

Chemical oxygen demand was analyzed to quantify the oxidizable content of the water samples. The analysis was done using the Hach photometric method by employing a Hach DR 5000 Spectrophotometer (figure 13) in the WERL at GSU. Prior to measurement, samples were digested using the Hach Method TNT 820 after adding 2 mL of sample to either Hach TNT 821 (low range: 0.00–150.00 mg/L) or TNT 822 (high range: 30.00–1,500.00 mg/L) vials. Samples were digested in a Hach DRB 200 for 2 hours at 150°C and were then measured after 20 min of digestion. To check accuracy, a 100.00 mg/L potassium hydrogen phthalate was used and measured as a standard. The measuring accuracy was found to be 99.9 percent.

Figure 13. Photo. Hach DR 5000 Spectrophotometer.

Nitrogen

TKN

TKN was used to quantify organic plus ammonia nitrogen in the water samples. The analysis was performed using Hach Method TNT 880 in a Hach DR 5000 Spectrophotometer using Hach TNT 880 combined vials. A volume of 1.3 mL of water sample was mixed with both Reagents A and B in a separate vial. The mixed vial was then digested in a Hach DRB 200 for 1 hour at 100°C. The mixture was allowed to cool for 20 min and then a microcap of Reagent C was added and mixed well. A 0.5-mL volume of the mixture was then added to TNT 880 vial 1 and shaken, and 0.2 mL of Reagent D was added to vial 1. In vial 2, 1 mL of undigested water sample was added and shaken. Both vials were left for 15 min and were then measured. Vial 1 represented the total nitrogen and vial 2 represented nitrate plus nitrite. By subtracting the reading of vial 2 from that of vial 1, the TKN value in the sample was obtained.

TN

Total nitrogen values were obtained from the TKN method mentioned above. Vial 1 of the TKN method represented the TN.

NH_3-N

NH_3-N was measured using the Hach direct ion selective electrode (ISE) method. A Hach sensION+MM340 ISE was used. A volume of 25 mL of 10 and 100.00 mg/L NH_3-N standard solutions were used for two-point calibration. One NH_3-N ionic strength adjustor (ISA) powder pillow was added to each standard solution in a beaker that was placed on a stirrer at a moderate stirring rate. The probe was rinsed and placed in the solution beaker for calibration. Sample measurement was done using the same steps of adding 25 mL of raw water sample in a beaker that was placed on the stirrer, adding an ammonia ISA pillow to the sample, and placing the probe in the beaker to read the value. An NH_3-N selective electrode was used for measuring NH_3-N in the sample.

NO_3^--N

NO_3^--N was measured using the Hach direct ISE method. A Hach sensION+MM340 ISE was used. A volume of 25 mL of 100.00 and 500.00 mg/L NO_3^--N standard solutions were used for two-point calibration. One NO_3^--N ISA powder pillow was added to each standard solution in a beaker that was placed on a stirrer at a moderate stirring rate. The probe was rinsed and placed in the solution beaker for calibration. Sample measurement was done using the same steps of adding 25 mL of raw water sample in a beaker that was placed on a stirrer, adding an NO_3^--N ISA pillow to the sample, and placing the probe in the beaker to read the value. An NO_3^--N selective electrode was used for measuring NO_3^--N in the sample.

NO_2^--N

NO_2^--N was calculated by subtracting measured NO_3^--N from measured nitrate plus nitrite nitrogen.

Phosphorus (PO_4^{3-}-P).

Total phosphorus (PO_4^{3-}-P) was measured using the Hach photometric method. A TNT 844 vial was used for measuring PO_4^{3-}-P in the sample. The foil cap of the vial was removed and the DosiCap™ Zip was unscrewed. A sample of 0.5 mL of raw water was added to the vial, the DosiCap Zip was flipped and tightened, and the vial was shaken and placed into a DRB 200 reactor at 100°C for 1 hour. The vial was allowed to cool for 20 min, then 0.2 mL of Reagent B was added to the vial, and the DosiCap Zip was used to cover the vial and was tightened. The sample was shaken, let settle for a few seconds, and then measured using the Hach DR 5000 Spectrophotometer in the WERL at GSU.

Solids

Total suspended solids and total dissolved solids were measured using the standard gravimetric method. A Whatman glass microfiber filter of 1.5 µm was used to filter 50 mL of the raw sample through a vacuum system. After filtration, the filter paper was placed on a previously weighed and tarred aluminum dish in the oven for 30 min at 105°C. The filtrate was placed in a previously weighed and tarred 80-mL beaker and was dried using a hot plate. Both the filter paper and the beaker weights were measured before and after the drying procedure and sample weights were obtained by subtracting the initial weight (W_i) from the final weight (W_f). The weight difference of the filter paper represented the TSS and the weight difference of the beaker represented the TDS. Both weights were divided by the sample volume (i.e., 50 mL) to obtain the solids concentration in mg/L.

Oil and Grease

Oil and grease were measured using the gravimetric procedure. A volume of 100 mL of the raw water sample was transferred to a beaker, and 2.5 mL HCl was added to the sample. The sample was then transferred to a separatory funnel, 20 mL n-hexane was added, and the sample was shaken vigorously for 2 min. The sample was then allowed to separate the oil phase from the aqueous phase for approximately 15 min. The upper layer was transferred to a previously weighed and tarred beaker, and the same extraction step was repeated twice, and each oil portion was added to the beaker. Anhydrous sodium sulfate was added to the beaker for extra dehydration of the oil phase. The oil extract was then dried on a hot plate at around 70°C to evaporate the n-hexane. After another 15 min, the dried residue was allowed to cool for around 20 min and was weighed, and the oil and grease in the sample was obtained by weight after subtracting the weight of the empty beaker from the weight of beaker with residue. The oil and grease concentration in mg/L was obtained by dividing the weight difference by the volume of water sample used, which was 100 mL.

pH

The sample pH was measured using a Thermo Fisher Orion Star A216 dual pH/DO electrode. The pH meter was first calibrated using Thermo Fisher pH standards of pH 4.00, 7.00, and 10.00. After calibration, the probe was directly inserted into the sample to record the pH value.

DO

The sample dissolved oxygen was measured using a Thermo Fisher Orion Star A216 dual pH/DO electrode. The DO meter was first calibrated using a previously sealed biological oxygen demand

(BOD) bottle filled with distilled water. After calibration, the probe was directly inserted into the water sample to record the DO value.

Conductivity

Conductivity was measured using a Fisher Scientific Traceable Conductivity Meter, and the values were obtained directly from the meter reading. Calibration and a sensitivity check were performed using a Thermo Scientific 1413 µS/cm (692.00 ppm as NaCl) Orion application solution.

Stochastic Empirical Loading and Dilution Model

The USGS StreamStats online tool was employed to run watershed delineation of all needed inputs to be defined for upstream and the highway (i.e., bridge site for this research project) in order to obtain downstream outputs, which are the drainage area, main channel length (to the drainage divide), main channel slope, imperviousness, and a basin development factor (BDF).[80] The latitude and longitude were used to provide precipitation outputs from the USGS gage station, pre-storm flow, and planning-level estimates of regional background water quality. SELDM is a mass balance model; it randomly generates stormflow volumes and concentrations from the upstream basin and the highway/BMP, and it adds the flows and loads together to calculate the downstream values. The downstream concentrations are the downstream loads divided by the downstream flows for each event. The longest length of highway (bridge) drainage basin is defined as the length along the main channel from the highway outfall (bridge scupper drain) to the drainage divide (bridge center line). The bridge drainage area was defined as the half cross-sectional area of the bridge to the scupper drain. The StreamStats network trace or raindrop tools were employed to obtain the main channel and then the measure or profile tool was used to find the length of the main channel to the basin divide for upstream. Additional SELDM input information for bridge sites, such as number of lanes, width of lanes, bridge deck material, and AADT, were obtained from GDOT online directories.

CHAPTER 4. RESULTS AND DISCUSSION

In this chapter, the concentrations of each selected water quality parameter from all six bridge sites are shown either in graphical or table format. The results are compared as upstream to scupper drain to downstream concentrations for each site and date, to show different patterns that can display the impact of the scupper drain runoff, where existing, on downstream water quality. Scupper drain runoff water quality from each site is also displayed for different sampling dates and time interval sampling from the scupper drain runoff and are displayed to show impact of time on the scupper drain runoff water quality. Since the impact of bridge scupper drain runoff from all parameters was observable, no statistical analysis was required for this section.

HEAVY METALS

Lead (Pb)

Figure 14 shows lead concentrations from the upstream, bridge scupper drain runoff, and downstream locations for all four Southeast Georgia study sites, which are SR 24 (figure 14-A), Rocky Ford (figure 14-B), Site US 80 (figure 14-C), and Site SR 297 (figure 14-D). From figure 14, it can be observed that scupper drain runoff was higher in lead compared to upstream and it was elevated downstream. For instance, in figure 14-B, -C, and -D for Sites RF, US 80, and SR 297, both the concentrations of total and dissolved lead spiked by the introduction of total and dissolved lead from the scupper drain runoff. Overall, total lead concentration ranges were very low in the range of 0.01–1.02 ppb (μg/L) for all four sites, compared to previously reported values (i.e., 5.00–160.00 ppb), which can be mainly attributed to minor leaks of leaded gasoline and from lubricating oil and grease and tire wear. (See references 1, 2, 6, 12, 21, and 81). According to the U.S. Environmental Protection Agency (EPA), leaded gasoline continued to be used for off-road purposes, such as aircraft, farming equipment, and marine engines.[81] However, lead as a heavy metal can settle in stream sediments and exert an accumulative hazardous effect to sensitive aquatic biota.[6,8,12]

Figure 14 A (Site SR 24) shows an increase in upstream total lead concentration that was not substantially altered by the introduction of lead from the scupper drain runoff. This indicates that lead was previously introduced to instream water from an upstream location to Site SR 24, and with the mean stream discharge (1,760 ft^3/s (cfs) on 11/20/2019 at Site SR 24) it could continue to exist in the Ogeechee River water with potential impacts downstream in the river with flow.

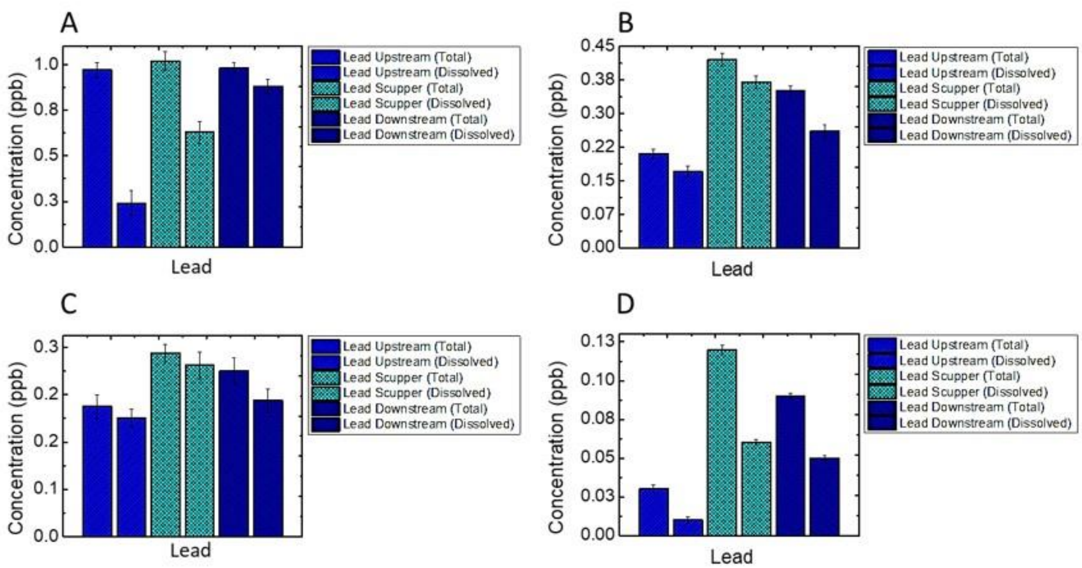

Figure 14. Bar charts. Lead concentrations (total and dissolved) from upstream, scupper drains, and downstream: (A) Site SR 24, (B) Site RF, (C) Site US 80, and (D) Site SR 297. Sample dates were 11/20/2019 for SR 24 and 10/22/2019 for RF, US 80, and SR 297.

From figure 14, lead concentrations in bridge scupper drain runoff ranged from 0.06 ppb for dissolved lead to 1.02 ppb for total lead, while downstream lead concentrations ranged from 0.05 to 0.98 ppb relative to 0.01 to 0.97 ppb for upstream lead concentration for all four Southeast

Georgia sites. This shows an instantaneous increase in lead concentration, ranging from 0.01 to 0.04 ppb in downstream concentration relative to upstream upon introduction of lead from the bridge scupper drain. Considering the potential accumulative impact of lead introduction to aquatic biotas, this increase in lead concentration is a matter of concern. Lead can interfere with the central nervous system; affect metabolism, growth, and reproduction; and damage kidneys and red blood cells, impacting oxygen transfer.[8] Notably, lead (instream and scupper drain) concentrations for Site SR 297 (figure 14-D) are lower than lead concentrations for Sites SR 24 (figure 14-A), Site RF (figure 14-B) and Site US 80 (figure 14-C). Since Sites RF, SR 24, and US 80 are all crossing the Ogeechee River from upstream to downstream, while only Site SR 297 is crossing the Ohoopee River, this indicates that lead concentrations are overall different in the Ohoopee River relative to the Ogeechee River.

Figure 15 shows concentrations of lead (total and dissolved) from scupper drains for all four Southeast Georgia sites from consecutive dates, aiming to follow lead concentration trends from different dates.

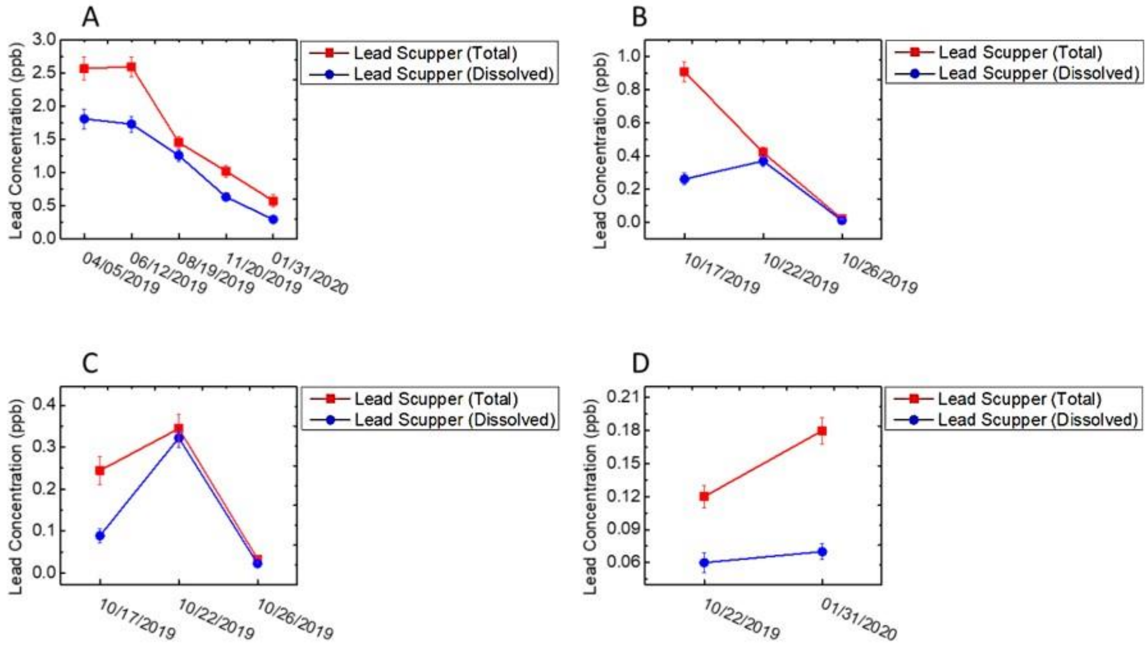

Figure 15. Line graphs. Lead concentrations (total and dissolved) from scupper drains and for different dates: (A) Site SR 24, (B) Site RF, (C) Site US 80, and (D) Site SR 297.

Figure 15-A (SR 24) shows a more extended date range than SR 24, which was the first site worked on, and resulted in more scupper drain samples as compared to the other three Southeast Georgia sites. From figure 14-A, bridge scupper drain runoff followed almost a sharp and consistent declining trend in both total and dissolved lead concentrations that is observed from June 2019 to January 2020. According to the mean river discharge data obtained from USGS 02202190 Ogeechee River at GA 24 (SR 24), near the Oliver, Georgia, gage station, the discharge values were 1,220, 628, 191, 1,760, and 3,630 cfs for 04/05/2019, 06/12/2019, 08/19/2019, 11/20/2019, and 01/31/2020, respectively. According to these values, the decline in lead concentrations starting from 06/12/2019 cannot be clearly attributed to stream discharge except for 11/20/2019 and 01/31/2020 as the stream discharge increased, which can cause a rapid dilution by dissipation of bridge scupper drain runoff upon introduction.

For both North Georgia sites, US 255 and SR 197, lead was undetected. However, on 01/24/2020, samples obtained from Site US 255 showed the presence of 0.01 ppb total lead in bridge scupper drain runoff with undetected lead in both the upstream and downstream samples (see figure 46-F in the appendix). This can be attributed to the high flow of the Chattahoochee River (1,110 cfs on 01/24/2020 at Site US 255) that likely had created a rapid diluting effect of the bridge scupper drain runoff lead upon introduction to the instream water. The daily sum precipitation followed an inconsistent trend with a drop in lead concentrations, as well, for these dates; the values were 0.00, 0.92, 0.13, 0.00 and 0.44 inch for 04/05/2019, 06/12/2019, 08/19/2019, 11/20/2019, and 01/31/2020, respectively. A possible reason for the lead decline could be the drop in the number of vehicles per day in the winter as compared to the summer due to the decline in weather quality that subsequently led to less on-road human activity, such as traveling.

Figure 15-B (Site RF) shows a trend that is similar to that of figure 14-A (Site SR 24). This similarity is no coincidence, as these sites are closest to each other (22.8 miles apart) as compared to their respective distances from the other two Southeast Georgia sites. However, the date range was different and less for figure 14-B as compared to figure 14-A, as can be seen from the figures. Thus, the most convenient explanation for the lead decline is the lower VPD and/or less demand on leaded gasoline during the dates of low lead concentrations in the bridge scupper drain runoff. The same attributions can explain the trend in figure 14-C (US 80); however, for figure 14-D (SR 297), lead concentration increased from 10/22/2019 to 01/31/2020.

Figure 16 shows concentrations of lead (total and dissolved) from scupper drains for all four Southeast Georgia sites, aiming to follow lead concentration trends along extended sampling intervals of 30 min of one rainfall event for each site. From figure 16, it is clear that the concentrations of lead decline with sampling time, which is attributed to the wash off of bridge deck contaminants with continuous precipitation. Thus, it is expected that at the early minutes of the rainfall event (first flush), the lead concentration is the highest in the scupper drain runoff from the bridge deck. It is known that bridge deck runoff will have the highest concentrations of dissolved contaminants in the first flush. (See references 1, 10, 17, and 68). The amount of dissolved contaminants, such as heavy metals, in bridge deck runoff is controlled by two critical factors, which are the amount of rain and the dry periods. The larger the amount of rain, the more diluted the contaminants are, and the longer the dry period, the more concentrated the contaminants are.[1,5,25]

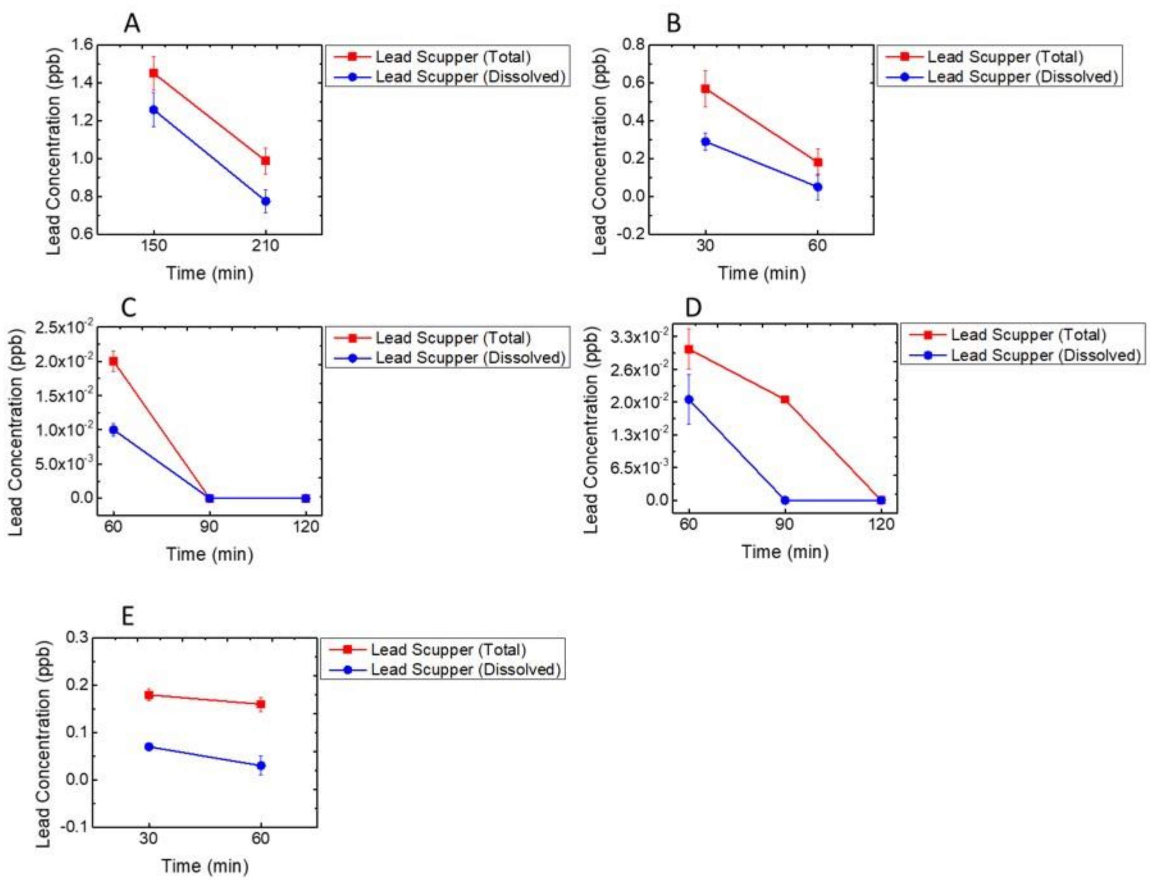

Figure 16. Line graphs. Lead concentrations (total and dissolved) from scupper drains and at time intervals: (A) Site SR 24 at GA 24 on 08/19/2019; (B) Site SR 24 at GA 24 on 01/31/2020; (C) Site RF on 10/26/2019; (D) Site US 80 on 10/26/2019; and (E) Site SR 297 at GA 297 on 01/31/2020.

Figure 17 shows a 1.5-month record for precipitation versus dry periods for each site for the time period before the aforementioned dates for figure 16. Although figure 17 does not show a consistent pattern of dry periods ahead of the precipitation periods on or before the aforementioned sampling dates in figure 16, it can be observed that dry periods prior and close to the sampling dates were frequent. For instance, for figure 17-A, there was a dry period of 10 days from 07/25/2019 to 08/04/2019 followed by another dry period of 4 days from 08/06/2019 to 08/09/2019; for figure 17-B, there was a dry period of 10 days from 01/13/2020 to 01/23/2020.

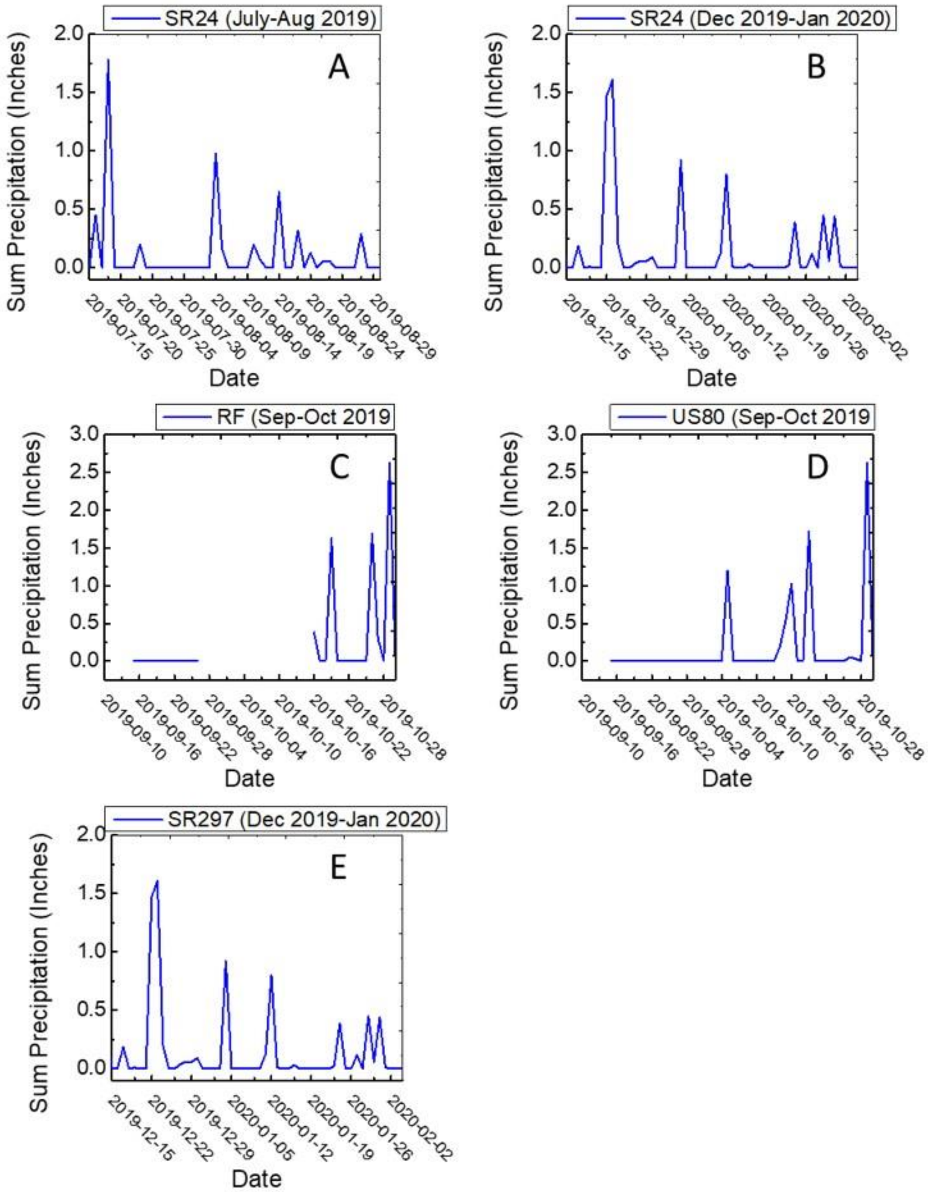

Figure 17. Graphs. Sum precipitation versus 1.5-month record for: (A) USGS 02202190 Ogeechee River at GA 24 for site SR 24 (July–Aug 2019; (B) Site SR 24 (Dec 2019–Jan 2020); (C) USGS 02202040 Ogeechee River at Rocky Ford Rd for Site RF (Sep–Oct 2019); (D) USGS 02202500 Ogeechee River for Site US 80 (Sep–Oct 2019); and (E) USGS 02225270 Ohoopee River at GA 297 for Site SR 297 (Dec 2019 – Jan 2020) (developed from [82–85]).

For figure 17-C (not all data were available), which is expected to follow a similar trend to figure 17-D (same climatic region), there was an extended dry period of 17 days from 09/16/2019 to 10/03/2019 and another dry period of 8 days from 10/06/2019 to 10/13/2019, and for figure 17-E, there were several dry periods—the longest being 10 days, from 01/13/2020 to 01/23/2020. These dry periods allow for the rapid accumulation of different contaminants on the bridge deck surface, some or all of which can be removed by the following rainfall events, depending on the rain intensity and the duration of the rainfall event.[1,5,25] For both North Georgia sites, US 255 and SR 197, there were no time interval data due to undetected lead in the samples.

Zinc (Zn)

Figure 18 shows upstream, bridge scupper drain runoff, and downstream zinc concentrations for all four Southeast Georgia study sites, which are Site SR 24 (figure 18-A), Site Rocky Ford (figure 18-B), Site US 80 (figure 18-C), and Site SR 297 (figure 18-D), and for the two North Georgia study sites, which are Site US 255 (figure 18-E) and Site SR 197 (figure 18-F). As with lead, figure 18 shows an increased total zinc concentration for both bridge scupper drain runoff and, consequently, in downstream water by introduction of zinc from scupper drain runoff, as compared to upstream water for all four sites. It is commonly known that the introduction of both zinc and copper to stream waters can be due to metal plating, bearing and bushing wear, moving engine parts, brake lining wear, maintenance operations, tire wear, motor oil, and bridge metal galvanization (See references 1, 5, 8, 11, 15, and 23).

Although the discharge of the Ogeechee River on 01/31/2020 was relatively high (3630 cfs), the dilution of bridge scupper drain runoff for zinc was insignificant and it could be clearly observed that scupper drain runoff impaired downstream water with zinc introduction for Site SR 24 (figure 18-A). Total zinc concentration increased by 1.25 ppb downstream relative to upstream, and dissolved zinc concentration increased by 0.76 ppb downstream relative to upstream (figure 18-A).

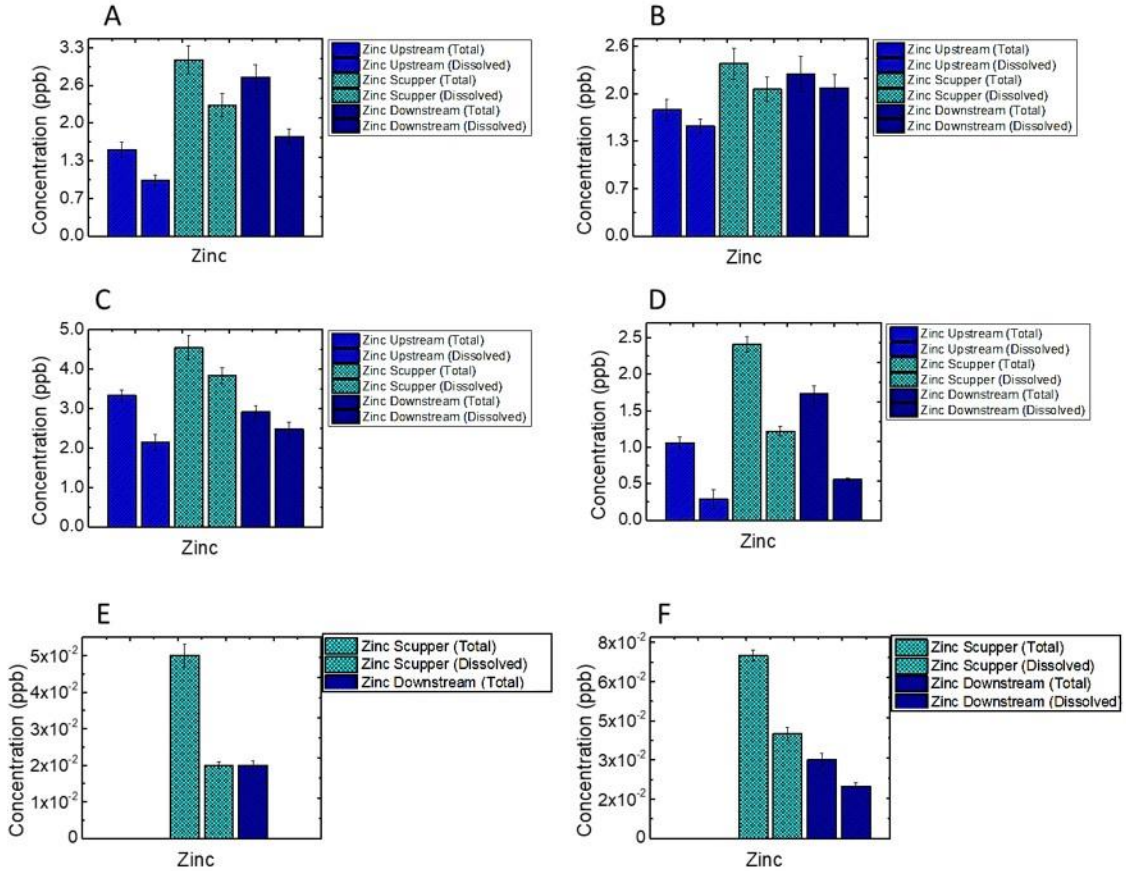

Figure 18. Bar graphs. Zinc concentrations (total and dissolved) from upstream, scupper drains, and downstream for: (A) Site SR 24, (B) Site RF, (C) Site US 80, (D) Site SR 297, (E) Site US 255, and (F) Site SR 197. Sample dates were 01/31/2020 for Sites SR 24 and SR 297, 10/17/2019 for Sites RF and US 80, and 12/10/2019 for Sites US 255 and SR 197.

Although zinc does not pose a huge threat of toxicity to aquatic biota compared to lead, the continued introduction of zinc into instream water can be of hazardous effect to the aquatic ecosystem with prolonged exposure (See references 1, 5, 6, 12, and 14). For the other three sites, the introduction of zinc, whether total and/or dissolved, from bridge scupper drain runoff to the downstream water was also observable (figure 18-B, -C, and -D).

Figure 18-E and figure 18-F show upstream, bridge scupper drain runoff, and downstream zinc concentrations for the two North Georgia sites, which are Site US 255 (figure 18-E) and Site SR 197 (figure 18-F). Generally, the water quality relative to most parameters was observed to be

higher with mostly undetected to low concentrations for the North Georgia sites as compared to the Southeast Georgia sites. This can be attributed to the remote and mostly residential environment of the two North Georgia study sites, which are located close to the city of Clarkesville. However, zinc could be observed to be introduced from scupper drain runoff to the downstream water at both sites.

Figure 18-E (Site US 255) shows the introduction of both total and dissolved zinc from bridge scupper drain runoff to the downstream water. However, only total zinc could be detected in the downstream water samples. According to USGS 02331000 Chattahoochee River near the Leaf, Georgia, gage station, the Chattahoochee River mean discharge on 12/10/2019 was 304 cfs, while that for the Soque River was 146 cfs, according to USGS 023312495 Soque River at GA 197, near the Clarkesville, Georgia, gage station close to Site SR 197. The discharge values for both rivers on 12/10/2019 were not substantially high, yet the discharge for the Chattahoochee River was higher at Site US 255 compared to that of the Soque River at Site SR 197. This difference in the stream discharge value can explain the rapid dilution and disappearance of dissolved zinc at Site US 255 (figure 18-E) in the downstream water samples in comparison to the presence of dissolved zinc at Site SR 197 (figure 18-F) in the downstream water samples.

According to the aforementioned USGS gage stations, on 12/10/2019, the sum precipitation was 0.11 inch for Site US 255, while that for Site SR 197 was not available on the USGS gage station monitoring website for the same date.

Figure 19 shows concentrations of zinc (total and dissolved) from scupper drains for all four Southeast Georgia sites from consecutive dates. For figure 19-A (Site SR 24) and figure 19-D (Site SR 297), there is no consistent trend in increasing or decreasing zinc concentration for the different dates. Total zinc concentrations ranged from 2 to 3.05 ppb for figure 19-A and from 0.08

to 2.68 ppb for figure 19-D, while dissolved zinc concentrations ranged from 1.54 to 3.00 ppb for figure 19-A and from 0.05 to 2.03 ppb for figure 19-D.

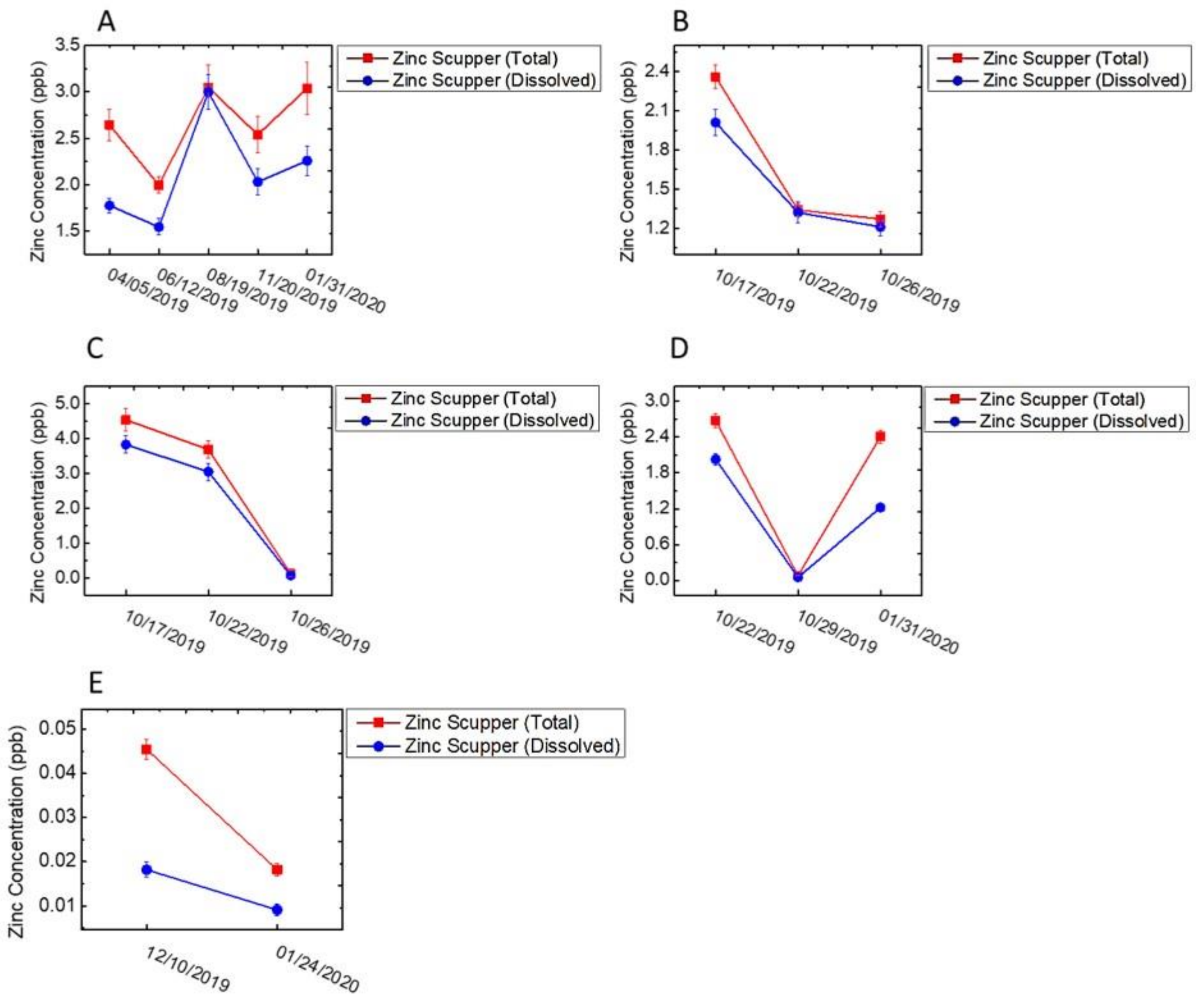

Figure 19. Line graphs. Zinc concentrations (total and dissolved) from scupper drains and for different dates for: (A) Site SR 24, (B) Site RF, (C) Site US 80, (D) Site SR 297, and (E) Site US 255.

Previously, relative to other heavy metals, the major reason for the presence of zinc in bridge deck runoff was reported to be vehicle wear. The highest total zinc concentration (3.05 ppb) was on 08/19/2019 for Site SR 24, and according to USGS 02202190 Ogeechee River at GA 24 (Site SR 24), near Oliver, Georgia, the sum precipitation on that day was 0.13 inch. The highest total zinc concentration (2.68 ppb) for Site SR 297 was on 10/22/2019, and according to USGS 02225270 Ohoopee River at GA 297 (Site SR 297), near Swainsboro, Georgia, the sum precipitation on that day was 0.00 inch. Therefore, the higher precipitation for Site SR 24 on 08/19/2019 as compared to the absence of precipitation for Site SR 297 on 10/22/2019 could have led to higher bridge deck sweeping of zinc from the Site SR 24 bridge and zinc introduction into the downstream water of the Ogeechee River at that location. For Site SR 297, less zinc could have been introduced into the leftover of the collection piping of the on-site sampling apparatus from precipitation on 10/19/2019 (2.06 inches); other reasons could be the varying corrosive effects, such as humidity, durability of structures, and age of structures for both bridges.

Figure 19-B (Site RF) shows a hyperbolic declining trend in zinc concentrations from 10/17/2019 to 10/26/2019. Unfortunately, according to USGS 02202040 Ogeechee River at Rocky Ford Rd, near the Rocky Ford, Georgia, gage station, there were no data available for precipitation in the aforementioned date ranges. Thus, the zinc decline trend cannot be evidenced according to precipitation values, which, however, were observable to be the reason for such a declining trend. For figure 19-C (Site US 80), an exponential declining trend of zinc concentrations from 10/17/2019 to 10/26/2019 was observable. According to USGS 02202500 Ogeechee River near the Eden, Georgia, gage station for Site US 80, the sum precipitation values on 10/16/2019 (a day earlier) was 1.02 inches and was 0.05 inch on 10/26/2019. Thus, it is evident that the higher precipitation on 10/16/2019 as compared to 10/26/2019 led to more sweeping of zinc from the bridge deck surface into the scupper drains on 10/16/2019 and zinc was introduced into the collection piping of the on-site sampling apparatus, as compared to 10/26/2019.

Since Sites SR 24, US 80, and RF are within the same climatic region (43.1 miles apart) and since sampling dates for Sites US 80 and RF were the same, it is expected that the declining trend in zinc concentrations for Site RF (figure 19-B) could be due to similar precipitation conditions as for Site US 80 (figure 19-C). However, the declining trends for both sites are not similar (i.e., hyperbolic decline for figure 19-B [Site RF] and exponential for figure 19-C [Site US 80]). These different declining trends could be attributed to different structural causes of the two bridges.

Zinc concentrations were detected in both North Georgia sites, Sites US 255 and SR 197. However, for sampling days, only Site US 255 scupper drain samples showed zinc concentrations for both days of sampling at both sites, which were 12/10/2019 and 01/24/2020, respectively. For Site SR 197, zinc was detected only on 12/10/2019. Concentrations of zinc for Site SR 197 on 12/10/2019 were found to be 0.07 (±0.00) ppb and 0.04 (±0.00) ppb for total and dissolved zinc, respectively. Figure 19-E shows concentrations of zinc (total and dissolved) from scupper drains for Site US 255 from the two aforementioned consecutive dates.

From figure 19-E, zinc concentrations were seen to linearly decline from 12/10/2019 to 01/24/2020. The sum precipitation value for 12/10/2019 was 0.11 inch, while that for 01/24/2020 was 1.43 inches; thus, it was expected that the zinc concentration trend would follow opposite trends, of increasing in concentration due to higher sweeping of bridge deck contaminants. To have a better explanation for this unexpectedly opposite trend, a 10-day record of precipitation trends was obtained from USGS 02331000 Chattahoochee River near Leaf, Georgia, for Site US 255 (figure 20).

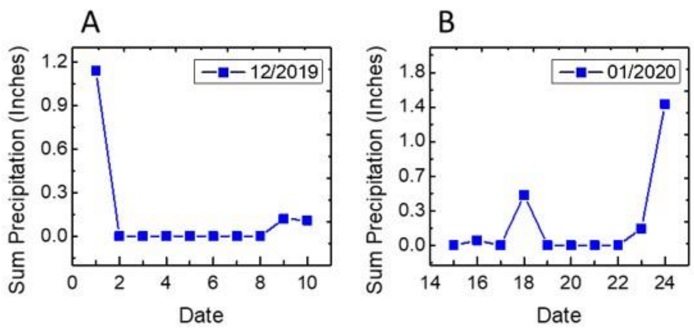

Figure 20. Line graphs. Sum precipitation versus 10-day records from USGS 02331000 Chattahoochee River for Site US 255 for: (A) 12/01/2019 to 12/10/2019, and (B) 01/15/2020 to 01/24/2020 (developed from [86]).

From figure 20, the dry period during the 9 days preceding 12/10/2019 was continuous from 12/02/2019 to 12/08/2019 (figure 20-A). Longer dry periods can lead to, in general, an accumulation of contaminants, whether metals or nonmetals, on the bridge deck, thus, increasing the contaminant amount over an extended period of time (See references 1, 5, 8, and 25). In the case of zinc, the researchers anticipated a higher introduction of zinc from other sources onto the bridge deck, such as vehicle tire wear and motor oil leakage, thus leading to the presence of more amounts of zinc on the bridge deck surface that could then be conveyed by bridge runoff into the scupper drains during the following rainfall event(s). Moreover, humidity can lead to the accumulative corrosion of bridge metal components and the presence of larger amounts of zinc on surfaces that are then carried by rainwater during the following rainfall event(s) and introduced with scupper drain runoff into the downstream water through the scupper drains.

For figure 20-B, it was observed that on 01/18/2020 a rainfall event took place, and that could be a possible explanation for less accumulation of zinc on the bridge deck surface during the following rainfall event on 01/23/2020 and 01/24/2020. As mentioned previously, concentrations of zinc for Site SR 197 on 12/10/2019 were found to be 0.07 and 0.04 ppb for total and dissolved zinc, respectively. However, no precipitation data were available for the aforementioned date.

Thus, the zinc declining trend cannot be evidenced according to precipitation values for Site SR 197, which, however, was observable to be the reason for such a declining trend for Site US 255 (figure 20-A) (i.e., same climatic region, 7.3 miles apart).

Fig 4.8 shows concentrations of zinc (total and dissolved) from scupper drains for all four Southeast Georgia sites and for the two North Georgia sites, aiming to follow zinc concentration trends along extended sampling intervals of 30 min of one rainfall event for each site.

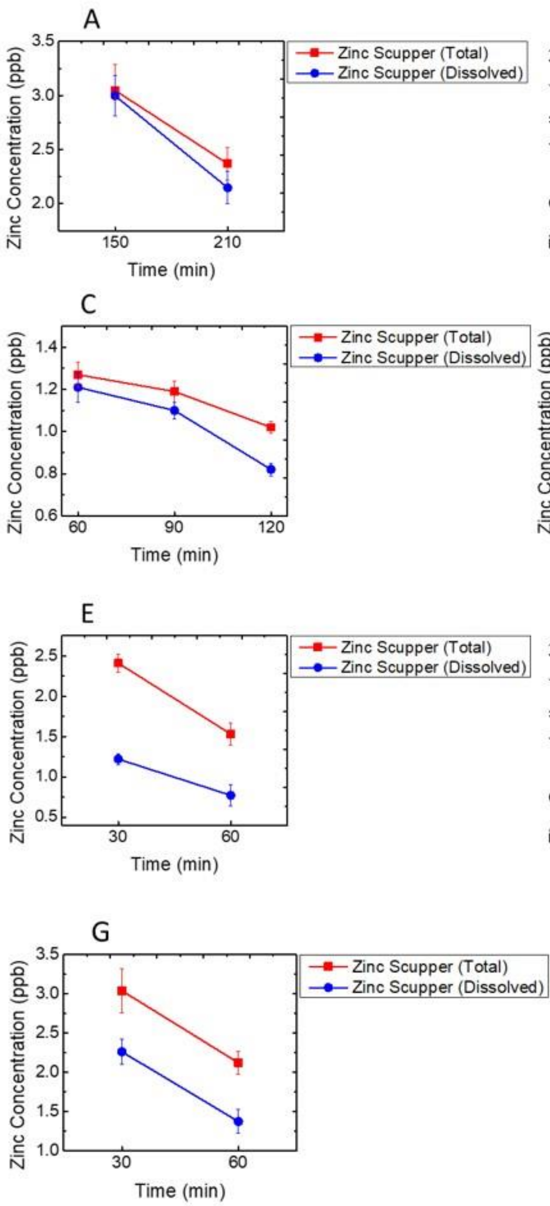

Figure 21. Line graphs. Zinc concentrations (total and dissolved) from scupper drains and at time intervals for: (A) Site SR 24 on 08/19/2019, (B) Site SR 24 on 01/31/2020, (C) Site RF on 10/26/2019, (D) Site US 80 on 10/26/2019, (E) Site SR 297 on 01/31/2020, (F) Site US 255 on 12/10/2019, and (G) Site SR 197 on 12/10/2019.

As with lead (figure 16), figure 21 shows that concentrations of zinc were declining with sampling time, which is also attributed to wash-off bridge deck contaminants with continuous precipitation, and thus, zinc concentrations decreasing with time during each rainfall event.

Figure 21 follows the same sum precipitation time record used in figure 17. Thus, and as mentioned previously, the larger the amount and higher the frequency of rain, the more diluted the contaminants are, and the longer the dry period, the more concentrated the contaminants are.

Figure 22 shows the sum precipitation record trend for Site US 255 for 1.5 months from 11/01/2019 to 12/15/2019. From figure 22, a dry period of 5 days preceded the sampling day (12/10/2019), ranging from 12/03/2019 to 12/07/2019, followed by a minor rainfall event from 12/08/2019 to 12/11/2019. This dry period was the longest during the recorded period in figure 22. Thus, and as mentioned previously, accumulation of zinc and other metals can occur and, therefore, higher concentrations of zinc will appear at the earlier part of the rainfall event and decline with time. For Site SR 197, sum precipitation data were not sufficient to cover the recorded period; however, it is expected that precipitation trends for Site SR 197 would be comparable to Site US 255 (i.e., the same climatic region).

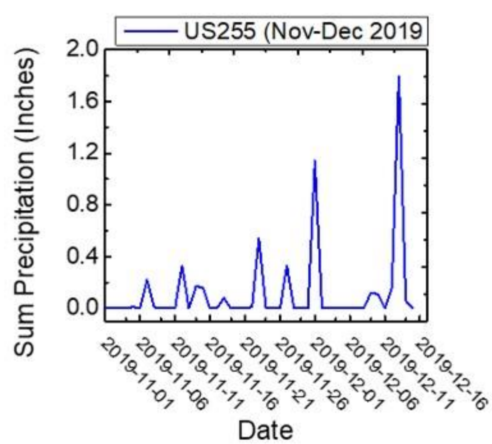

Figure 22. Line graph. Sum precipitation versus 1.5-month record from USGS 02331000 Chattahoochee River for Site US 255 (Nov–Dec 2019) (developed from [86]).

Copper (Cu)

Copper was detected in samples from all four Southeast Georgia sites, specifically in bridge scupper drain runoff and downstream samples. Figure 23 shows upstream, bridge scupper drain runoff, and downstream copper concentrations for all four Southeast Georgia sites and in both North Georgia sites, which are Site SR 24 (figure 23-A), Site Rocky Ford (figure 23-B), Site US 80 (figure 23-C), Site SR 297 (figure 23-D), Site US 255 (figure 23-E), and Site SR 197 (figure 23-F). As shown in figure 23, all scupper drains and downstream samples contained observable amounts of copper, as copper was clearly introduced to downstream water from scupper drain runoff through the scupper drains. As mentioned previously, copper and zinc are both produced from traffic, and are then introduced to instream water in the bridge runoff through the scupper drains.

In figure 23-A (Site SR 24) and Fig 4.10-D (Site SR 297), copper was undetected in upstream samples, while for figure 23-B (Site RF) and 4.10-C (Site US 80), copper was detected in all instream samples, and for all four sites, copper was detected in scupper drain samples. Therefore, it can be inferred that copper was definitely introduced from scupper drain runoff into downstream water through scupper drains, especially for figure 23-B and figure 23-C, copper increased slightly in downstream samples (1.24 ppb of total copper and 0.82 ppb for dissolved copper. Respectively) relative to upstream samples (1.18 ppb of total copper and 0.76 ppb for dissolved copper).

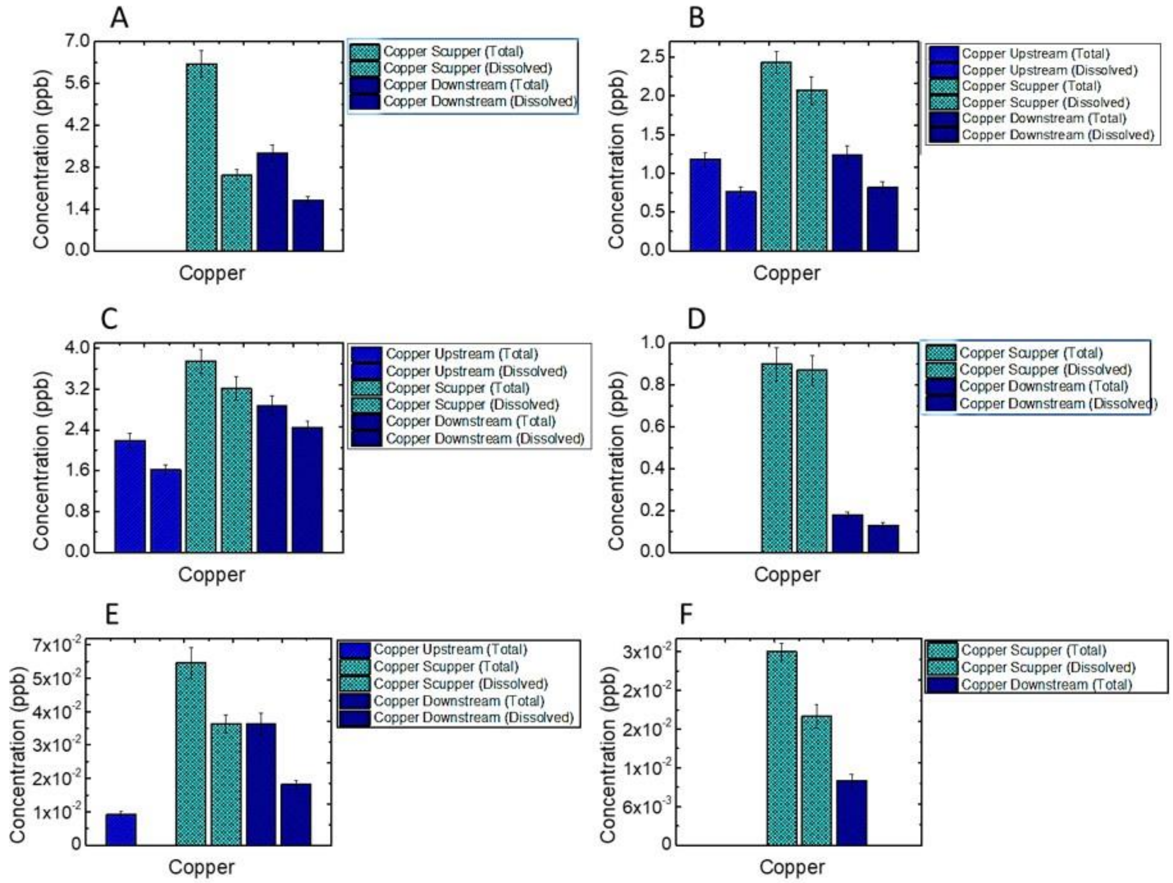

Figure 23. Bar graphs. Copper concentrations (total and dissolved) from upstream, scupper drains, and downstream for: (A) Site SR 24, (B) Site RF, (C) Site US 80, (D) Site SR 297, (E) Site US 255, and (F) Site SR 197. Sample dates were 01/31/2020 for Site SR 24, 10/17/2019 for Sites RF and US 80, 10/29/2019 for Site SR 297, and 12/10/2019 for Sites US 255 and SR 197.

Copper can induce acute toxicity to aquatic biota at high concentrations or prolonged exposure (See references 1, 5, 6, 12, and 14). The high concentrations of copper in scupper drain samples relative to instream samples for figure 23-A and figure 23-D can confidently be related to both factors of precipitation and stream discharge. The average stream discharge value for Site SR 24 (figure 23-A) on 01/31/2020 was 3630 cfs and its sum precipitation was 0.44 inch, while the average stream discharge value for Site SR 297 (figure 23-D) on 10/29/2019 was 52 cfs and its sum precipitation was 2.79 inches. Thus, the combination of both factors of rain and stream discharge can lead to the introduction of relatively large amounts of copper into scupper drain runoff (by rain), while stream discharge acts to dilute the introduced copper from scupper drains into downstream water by rapid dissipation.

For figure 23-B and figure 23-C, the average stream discharge value for Site RF (figure 23-B) on 10/17/2020 was 109 cfs, but no data were available for its sum precipitation value for the aforementioned date, while the average stream discharge value for Site US 80 (figure 23-C) on 10/17/2019 was 119 cfs and its sum precipitation was 0.00 inch. For both sites and although precipitation data were not available for the date, it was observable that precipitation was significant. The low stream discharge values of 109 cfs for Site RF (figure 23-B) and 119 cfs for Site US 80 (figure 23-C) can explain the lower instream dilution of copper in both upstream and downstream samples.

Figure 23-E and figure 23-F show upstream, bridge scupper drain runoff, and downstream copper concentrations for the two North Georgia sites, which are Site US 255 (figure 23-E) and Site SR 197 (figure 23-F). As mentioned previously, the water quality relative to most parameters was observed to be higher at the Southeast Georgia sites compared to the mostly undetected to low concentrations for the North Georgia sites.

At Site US 255 (figure 23-E), total copper was detected in the upstream samples, while it was undetected in the upstream samples for Site SR 197 (figure 23-F). Moreover, concentrations of total and/or dissolved copper in scupper drain runoff and downstream samples were detected to be higher at Site US 255 as compared to Site SR 197. The scupper drain runoff sample total copper concentration for Site US 255 was found to be 0.06 ppb and its dissolved copper was 0.04 ppb, while the scupper drain runoff sample total copper concentration for Site SR 197 was found to be 0.03 ppb and its dissolved copper was 0.02 ppb. Downstream sample total copper concentration for Site US 255 was found to be 0.04 ppb and its dissolved copper was 0.02 ppb, while downstream sample total copper concentration for Site SR 197 was found to be 0.01 ppb and its dissolved copper was undetected. As mentioned in a previous section, according to USGS 02331000 Chattahoochee River near the Leaf, Georgia, gage station, the Chattahoochee River mean discharge on 12/10/2019 was 304 cfs, while that for the Soque River was 146 cfs, according to USGS 023312495 Soque River at GA 197, near the Clarkesville, Georgia, gage station. The discharge values for both rivers on 12/10/2019 were not substantially high, yet the discharge for the Chattahoochee River was higher at Site US 255 compared to that of the Soque River at Site SR 197. Thus, it was expected that the copper concentration would be less in Site US 255 samples as compared to Site SR 197 samples. It is expected that copper was most likely introduced in larger concentrations from a previous source relative to the Site US 255 bridge location, and with the fairly low stream discharge value of 304 cfs for the Chattahoochee River, dissipation of copper was not efficient before reaching the Site US 255 bridge site location. The higher concentrations of copper in scupper drain runoff samples from Site US 255 as compared to Site SR 197 can be attributed to higher traffic or different structural components at Site US 255 as compared to Site SR 197.

Figure 24 shows concentrations of copper (total and dissolved) from scupper drain runoff for all four Southeast Georgia sites and the two North Georgia sites from consecutive dates. Figure 24-A (Site SR 24) and figure 24-C (Site US 80) show an overall declining trend in copper concentration consecutively, while for figure 24-B (Site RF) and figure 24-D (Site SR 297) there is no consistent trend in increasing or decreasing copper concentration for the different dates. Total copper concentrations ranged from 4.27 to 8.08 ppb for figure 24-A and from 2.03 to 3.75 ppb for figure 24-C, while dissolved copper concentrations ranged from 2.54 to 7.03 ppb for figure 24-A and from 1.84 to 3.21 ppb for figure 24-C. To better understand the declining copper concentrations for figure 24-A and figure 24-C, a record of 6 months of precipitation was developed from USGS 02202190 Ogeechee River at GA 24, near Oliver, Georgia, for Site SR 24, and a 1-month record for precipitation was developed from USGS 02202500 Ogeechee River near Eden, Georgia, for Site US 80 (figure 25). Figure 24-E and figure 24-F show total and dissolved copper concentrations in scupper drain samples obtained from both sites, Site US 255 (figure 24-E) and Site SR 197 (figure 24-F), on 12/10/2019 and 01/24/2020. For both sites, it was observed that copper concentrations were higher in samples obtained on 12/10/2019 as compared to those obtained on 01/24/2020.

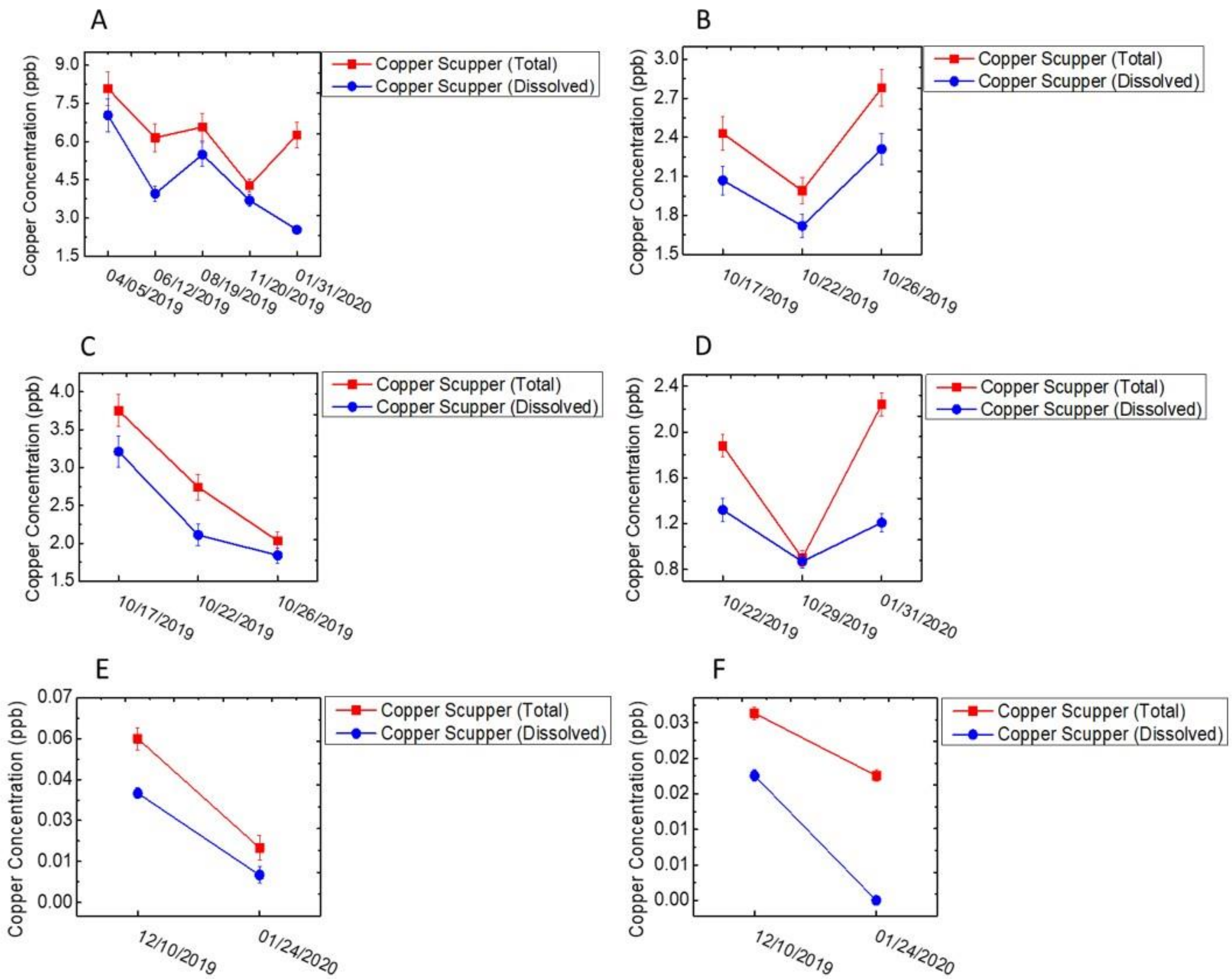

Figure 24. Line graphs. Copper concentrations (total and dissolved) from scupper drains and for different dates for: (A) Site SR 24, (B) Site RF, (C) Site US 80, (D) Site SR 297, (E) Site US 255, and (F) Site SR 197.

Figure 25-A shows that during the period from mid-February to mid-April 2019, precipitation was lower than later months for Site SR 24. Similar previous discussion on dry periods versus precipitation periods, accumulation of copper from different sources is expected to be higher during dry and/or low precipitation periods. Therefore, with the first flushes of rain during early April 2019 and up to mid May 2019, it is anticipated that amounts of copper were higher in scupper drain runoff as compared to later months of higher precipitation frequency and amount. The same interpretation can be given to Site US 80 (figure 25-B), which shows lower

frequency and amount of precipitation during early October 2019 as compared to mid- and late October 2019.

For Site RF (figure 24-B), no precipitation data were found to cover a reasonable range over the month of October 2019. However, as Site RF should be within the same climatic region as Sites SR 24 and US 80, it was expected for Site RF to follow similar precipitation trends to those sites. Therefore, the inconsistent varying copper concentration trend for Site RF (figure 24-B) cannot be interpreted with respect to precipitation trends and could be due to other factors, such as humidity or varying temperature and wind speeds (evaporation). For Site SR 297 (figure 24-D), precipitation versus date was available, and figure 26 shows the sum precipitation versus a 4-month record for Site SR 297.

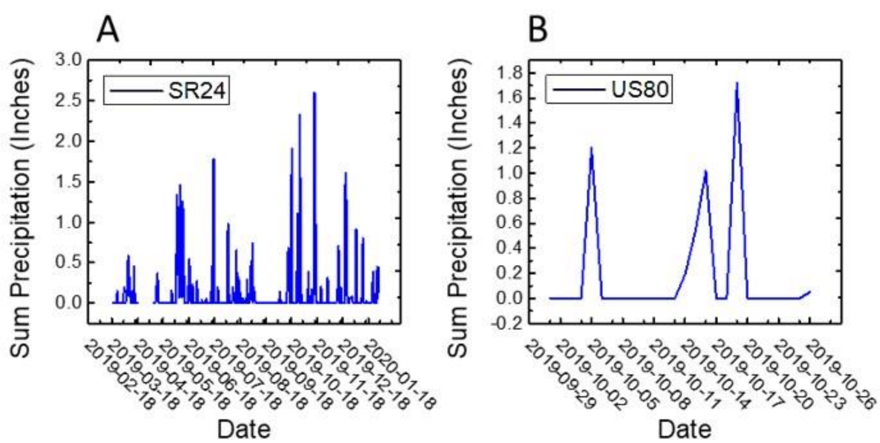

Figure 25. Line graphs. Sum precipitation versus: (A) 6-month record from USGS 02202190 Ogeechee River at GA 24 for Site SR 24, (B) 1-month record from USGS 02202500 Ogeechee River for Site US 80 (developed from [82, 84]).

From figure 26, it can be observed that precipitation was high during the latter half of October 2019 and then a decline in precipitation frequency and amount followed. The decline in precipitation frequency and amount between the period of November 2019 to January 2020 can explain the sharp rise in total copper concentration with precipitation on 01/31/2020 (figure 24-D) due to accumulation during this period. The drop in copper concentrations on 10/29/2019 can be related to high dilution from the intense precipitation amount and frequency during the period of 10/10/2019 to the end of the month.

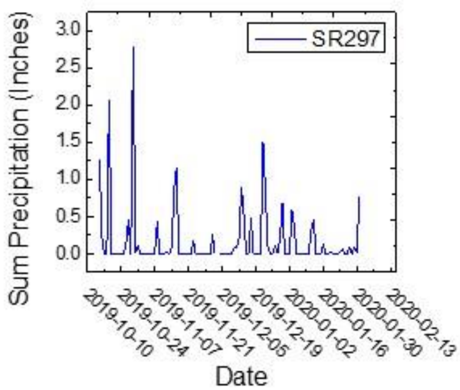

Figure 26. Line graph. Sum precipitation versus 4-month record from USGS 02225270 Ohoopee River at GA 297 for Site SR 297 (developed from [85]).

The sum precipitation at Site US 255 was 0.11 inch on 12/10/2019 and was 1.43 inches on 01/24/2020. According to these precipitation data, it is expected that copper concentrations would follow an opposite trend to that in figure 24. Figure 27 shows a record of precipitation data along 3 months for Site US 255, and from figure 27, it was observed that precipitation was generally abundant for the period from 12/10/2019 to 01/24/2020. However, from figure 27, a dry period of 9 days is observed for the period of 12/01/2019 to 12/09/2019, which can explain the high copper concentration with the precipitation flush on 12/10/2019 as compared to the following month of 01/24/2020 that shows high frequency and amount of precipitation. For Site SR 197, no precipitation data were available for the dates of sampling or to cover a reasonable time period for the site; however, since both North Georgia sites are within the same climatic region, it is expected that precipitation at Site SR 197 would follow the same trends as Site US 255. Therefore, it can be inferred that dry periods play an important role in affecting heavy metals concentrations in bridge deck scupper drain runoff.

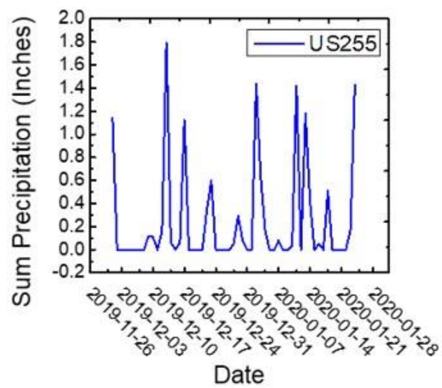

Figure 27. Line graph. Sum precipitation versus 3-month record from USGS 02331000 Chattahoochee River for Site US 255 (developed from [86]).

Figure 28 shows concentrations of copper (total and dissolved) from scupper drains for all four Southeast Georgia sites and two North Georgia sites, aiming to follow copper concentration trends along extended sampling intervals of 30 min of one rainfall event for each site. As with lead (figure 16) and zinc (figure 21), figure 28 shows that concentrations of copper were declining with sampling time, which is also attributed to wash off of bridge deck contaminants with continuous precipitation, and thus, copper concentrations decreased with time during each rainfall event. The concentrations of dissolved copper for figure 28-C (Site RF) showed a sharp decline at 90 min and then an increase at 120 min. However, the overall trend is a decline as compared to the concentration of dissolved copper at 60 min. The rise of copper concentration at 120 min as compared to 90 min could be attributed to more accumulation or sweep of bridge deck contaminants with runoff.

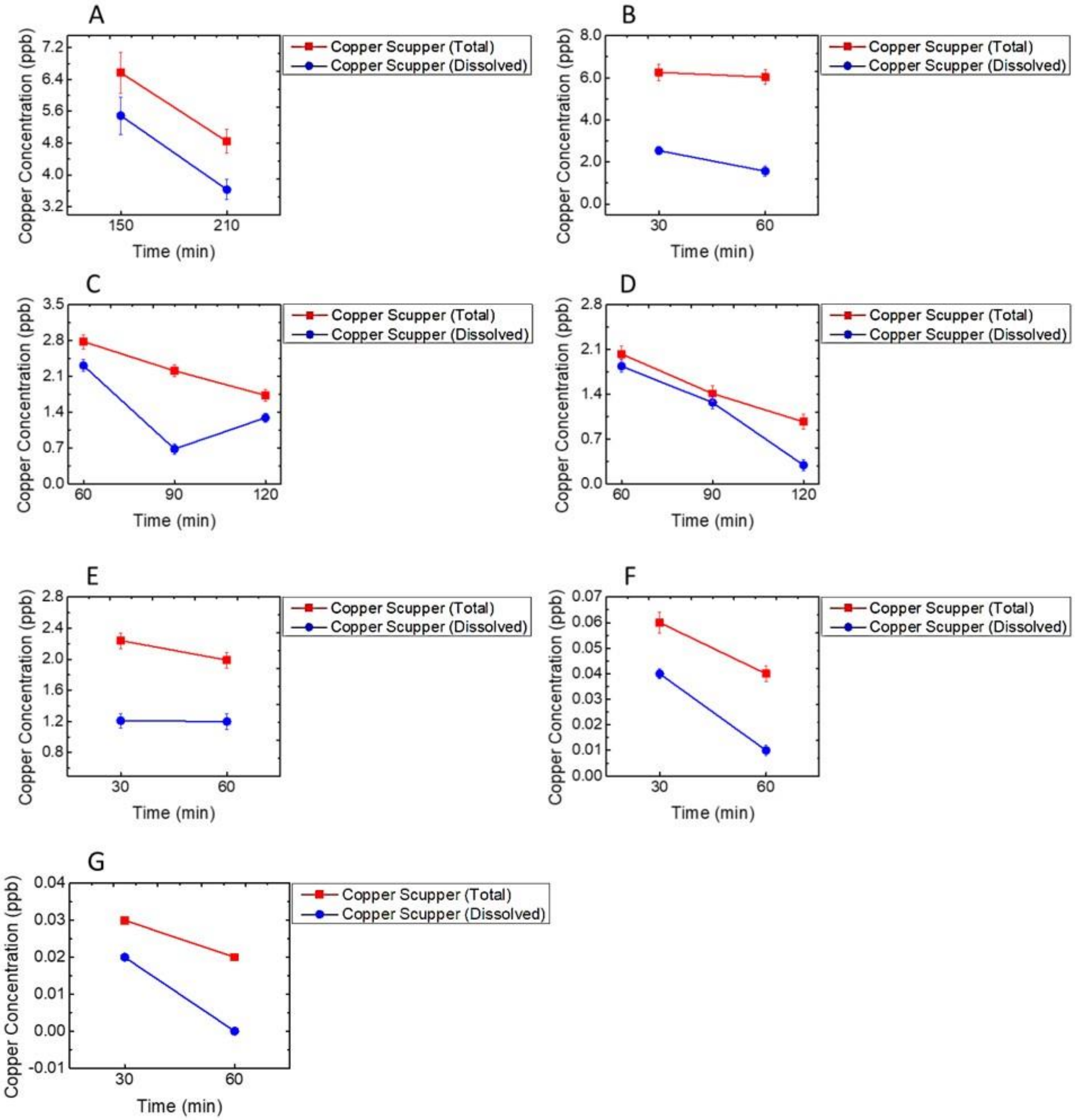

Figure 28. Line graphs. Copper concentrations (total and dissolved) from scupper drains and at time intervals for: (A) Site SR 24 on 08/19/2019, (B) Site SR 24 on 01/31/2020, (C) Site RF on 10/26/2019, (D) Site US 80 on 10/26/2019, (E) Site SR 297 on 01/31/2020, (F) Site US 255 on 12/10/2019, and (G) Site SR 197 on 12/10/2019.

Figure 28 follows the same sum precipitation time records as used in figure 17 for the four Southeast Georgia sites. Thus, the larger the amount and frequency of rains, the more diluted the contaminants are, and the longer the dry period, the more concentrated the contaminants are.

Figure 28-F and figure 28-G show concentrations of copper (total and dissolved) from scupper drains for the two North Georgia sites, aiming to follow copper concentration trends along extended sampling intervals of 30 min of one rainfall event for each site. It is observed that copper concentrations were following the same declining trend as previously reported with lead and zinc, due to dilution of copper with time during the rainfall event.

Figure 28-F and figure 28-G follow the same sum precipitation time record used in figure 20-A for Site US 255. Thus, and as mentioned previously, accumulation of copper, among other heavy metals, can occur and, therefore, higher concentrations of copper will appear earlier in the times of the rainfall event and decline with time. For Site SR 197, sum precipitation data were not sufficient to cover the record period; however, it is expected that precipitation trends for Site SR 197 would be comparable to Site US 255 (same climatic region).

Summary of Heavy Metals Results

In the previous subsections, it was found that the introduction of Pb, Zn, and Cu from scupper drains into instream water was observable in most cases and led to elevated downstream heavy metals concentrations in instream waters. When heavy metals concentrations in scupper drain runoff were followed with subsequent dates, it was found that heavy metals concentrations varied randomly with different dates. These variations were attributed to different factors, such as the preceding dry periods, rainfall intensity, frequency of rains, and traffic volume. A consistent trend of decline in concentrations was observed in most cases in scupper drain runoff concentrations of Pb, Zn, and Cu with sampling time, which was attributed to a decrease of heavy metals concentrations in scupper drain runoff after the first flush and with time due to dilution by rainwater.

POLYCYCLIC AROMATIC HYDROCARBONS (PAHS)

Previous studies on PAHs in stormwater runoff and the existence of PAHs in the environment reported that the sources of PAHs in the environment are mainly atmospheric- and traffic-related, and, although it was reported that concentrations of PAHs are low in nature, their accumulation is of hazardous concern. (See references 1–3, 5, 6, 12, 14, 19–22, and 26–28). In this research project, 18 PAHs were investigated in instream and scupper drain samples from all six Georgia study sites. However, only 7 of the 18 PAHs investigated were detected in scupper drain runoff samples of two Southeast Georgia sites (i.e., Sites SR 24 and US 80), while PAHs were undetected in their instream samples. For the other four sites of the study, PAHs were undetected in scupper drain runoff and instream samples. The seven PAHs that were detected in scupper drain runoff samples from Sites SR 24 and US 80 were benz[a]anthracene, benzo[a]pyrene, fluoranthene, pyrene, chrysene, indeno-[1,2,3-cd]-pyrene, and benzo[b]fluoranthene.

Table 11 shows all data for the PAH analysis for all six study sites with their respective dates. From table 11, it can be seen that benz[a]anthracene and benzo[a]pyrene were detected in scupper drain samples from Site SR 24 on 04/05/2019, and their mean concentrations were found to be 15.000 and 17.000 parts per trillion (ppt) (ng/L), respectively.

Table 11. PAHs data from all six Georgia study sites.

Date	Polycyclic Aromatic Hydrocarbons (PAHs)		
	Upstream	Scupper	Downstream
Site SR 24			
04-05-19	PAHs undetected	Benz[a]anthracene 0.015 ppb (15.000 ppt) Benzo[a]pyrene 0.017 ppb (17.000 ppt)	PAHs undetected
06-12-19	Not sampled	PAHs undetected	Not sampled
08-19-19	Not sampled	Benz[a]anthracene 0.002 ppb (2.000 ppt) Fluoranthene 0.007 ppb (6.700 ppt)	Not sampled
11-20-19	PAHs undetected	PAHs undetected	PAHs undetected
01-31-20	PAHs undetected	PAHs undetected	PAHs undetected
Site Rocky Ford (RF)			
10-07-19	PAHs undetected		PAHs undetected
10-17-19	PAHs undetected	PAHs undetected	PAHs undetected
10-22-19	PAHs undetected	PAHs undetected	PAHs undetected
10-26-19	Not sampled	PAHs undetected	Not sampled
Site US 80			
10-07-19	PAHs undetected		PAHs undetected
10-17-19	PAHs undetected	Pyrene 0.030 ppb (29.500 ppt) Chrysene 0.011 ppb (11.250 ppt) Indeno-[1,2,3-cd]-pyrene 0.007 ppb (7.150 ppt) Benzo[b]fluoranthene at 0.005 ppb (5.200 ppt)	PAHs undetected
10-22-19	PAHs undetected	PAHs undetected	PAHs undetected
10-26-19	Not sampled	PAHs undetected	Not sampled
Site SR 297			
10-17-19	PAHs undetected		PAHs undetected
10-22-19	PAHs undetected	PAHs undetected	PAHs undetected
10-29-19	PAHs undetected	PAHs undetected	PAHs undetected
11-20-19	PAHs undetected		PAHs undetected
1-31-20	PAHs undetected	PAHs undetected	PAHs undetected
Site US 255			
11-22-19	PAHs undetected		PAHs undetected
12-10-19	PAHs undetected	PAHs undetected	PAHs undetected
1-24-20	PAHs undetected	PAHs undetected	PAHs undetected
Site SR 197			
11-22-19	PAHs undetected		PAHs undetected
12-10-19	PAHs undetected	PAHs undetected	PAHs undetected
1-24-20	PAHs undetected	PAHs undetected	PAHs undetected

Thus, the concentrations of benz[a]anthracene and benzo[a]pyrene were found to exceed the drinking water guideline value of 10.000 ppt provided by the National Health and Medical Research Council in Australia and the Arkansas Water Resource Center,[52] which raises the alarm for the essential implementation of mitigation measures. Benz[a]anthracene was detected again in scupper drain samples from Site SR 24 on 08/19/2019 at a concentration of 2.000 ppt, and fluoranthene was detected at a concentration of 6.700 ppt, while benzo[a]pyrene was absent. The lower concentrations of detected PAHs on 08/19/2019 as compared to the concentrations of detected PAHs on 04/05/2019 can be attributed to higher dilution of PAHs by the larger precipitation on 08/19/2019 (0.13 inch) as compared to precipitation on 04/05/2019 (0.06 inch). Several other factors might exist, including the amount of PAHs conveyed in the atmosphere and/or introduced by traffic to the bridge deck and/or bridge deck runoff.

Pyrene, chrysene, indeno-[1,2,3-cd]-pyrene, and benzo[b]fluoranthene were all detected in scupper drain samples from Site US 80 obtained on 10/17/2019 at concentrations of 29.500, 11.250, 7.150, and 5.200 ppt, respectively. The concentrations of both pyrene and chrysene were found to exceed the drinking water guideline value of 10.000 ppt provided by the NHMRC and AWRC, while the concentrations of both indeno-[1,2,3-cd]-pyrene and benzo[b]fluoranthene were found to be below the drinking water guideline value.[52] Both pyrene and chrysene are four-ringed PAHs, while indeno-[1,2,3-cd]-pyrene is six-ringed and benzo[b]fluoranthene is five-ringed. The lower concentrations of indeno-[1,2,3-cd]-pyrene and benzo[b]fluoranthene as compared to pyrene and chrysene can be related to several factors, such as the higher hydrophobicity of indeno-[1,2,3-cd]-pyrene and benzo[b]fluoranthene as compared to pyrene and chrysene, a higher oxidative stress, and/or high biodegradability of the large molecular sized PAHs.

Summary of PAH Results

Of 18 PAHs investigated, only seven were detected in scupper drain samples of two Southeast Georgia sites, which were Sites SR 24 and US 80. The seven PAHs that were detected in scupper drain runoff samples from Sites SR 24 and US 80 were benz[a]anthracene, benzo[a]pyrene, fluoranthene, pyrene, chrysene, indeno-[1,2,3-cd]-pyrene, and benzo[b]fluoranthene. It was found that benz[a]anthracene and benzo[a]pyrene were detected in scupper drain samples from Site SR 24 on 04/05/2019, and their mean concentrations were found to be 15.000 and 17.000 ppt (ng/L), respectively, which exceeded the drinking water guideline value of 10.000 ppt provided by the NHMRC and AWRC. Moreover, the concentrations of both pyrene and chrysene were found to exceed the drinking water guideline value of 10.000 ppt provided by the NHMRC and AWRC. The high concentrations of benz[a]anthracene, benzo[a]pyrene, pyrene, and chrysene raise the concern of potential impact of these four PAHs on human and aquatic life upon exposure. The presence of PAHs in scupper drain runoff can be confidently related to traffic.

CHEMICAL OXYGEN DEMAND (COD)

COD was evaluated to determine the concentration of oxidizable organic and inorganic chemical content in instream and scupper drain samples. COD concentrations were detected from instream and scupper drain samples of all six sites that were subjects of the study, suggesting that COD sources are numerous. Figure 29 shows the impact of the COD-concentrated scupper drain runoff on instream water quality at the four Southeast Georgia study sites and the two North Georgia study sites. Sources of the high bridge deck runoff COD are mainly traffic, bridge deck material, and atmospheric depositions.[1,8,47]

From figure 29, it can be observed that the introduction of scupper drain runoff can increase downstream COD values as compared to upstream values for all four Southeast Georgia sites. COD increased by 5.50, 3.40, 2.30, and 4.90 ppm (mg/L) in downstream as compared to upstream waters for Sites SR 24, RF, US 80 and SR 297, respectively. Traffic and leaching of bridge deck concrete material with precipitation are sources for the introduction of oxidizable content from scupper drain runoff and the increased COD in instream water.[1,45]

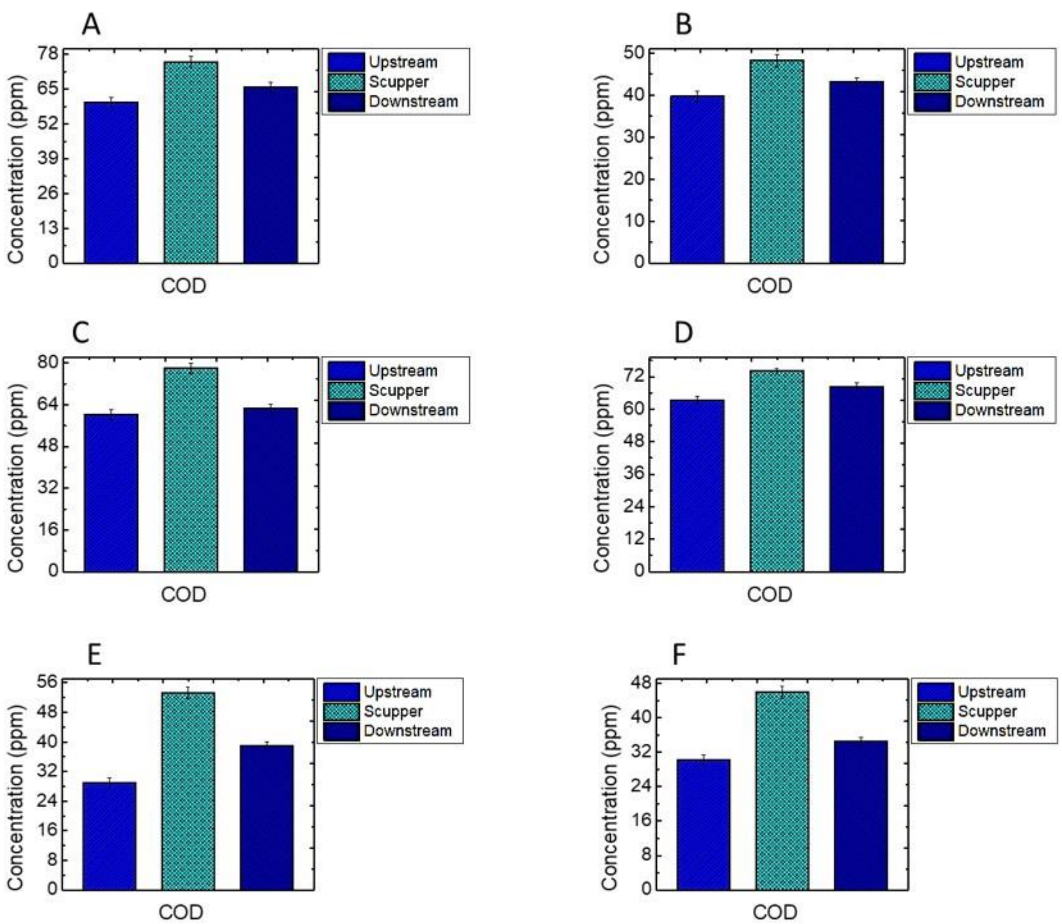

Figure 29. Bar graphs. COD concentrations from upstream, scupper drains, and downstream for: (A) Site SR 24, (B) Site RF, (C) Site US 80, (D) Site SR 297, (E) Site US 255, and (F) Site SR 197. Sample dates were 01/31/2020 for Sites SR 24 and SR 297, 10/22/2019 for Site RF, 10/17/2019 for Site US 80, and 12/10/2019 for Sites US 255 and SR 197.

Traffic volume can also drive the prediction of instream COD concentration, as there is a direct relation between traffic volume and COD in instream water.[1,8,47] COD was found to be higher for nonurban highways compared to urban highways, which shows that some COD sources are not transportation-related.[1,25] The variation of the impact of scupper drain runoff on instream water is close in figure 29, indicating that precipitation impact on varying COD concentration was not of a significant difference between the four Southeast Georgia sites.

For the two North Georgia sites (figure 29-E and -F), the introduction of scupper drain runoff into instream water also elevates the COD concentration downstream as compared to upstream. COD increased by 10.00 and 4.30 ppm (mg/L) in downstream as compared to upstream waters for Site US 255 and Site SR 197, respectively. Since no precipitation data were available for the dates of sampling and no data were available to cover a reasonable time period for Site SR 197, and both North Georgia sites are within the same climatic region, no comparison can be made based on precipitation trends. The average stream discharge was 304 cfs for Site US 255 and 146 cfs for Site SR 197. Thus, the larger difference in COD for Site US 255 (10.00 ppm) as compared to that of Site SR 197 (4.30 ppm) can be related to the structural difference of the two bridges and/or higher traffic volume on the former in dry periods preceding the rainfall event.

Figure 30 shows COD concentrations from scupper drains for all four Southeast Georgia sites and the two North Georgia sites from consecutive dates. Figure 30 shows an increasing trend for COD at Site SR 24 (figure 30-A) and at Site SR 297 (figure 30-D), and a declining COD trend for Site RF (figure 30-B) and Site US 80 (figure 30-C). For figure 30-A and figure 30-D, the increasing trend can be attributed to an increased accumulation of oxidizable content from traffic sources and/or more leaching of bridge deck material prior to the rainfall event for each specified date. Moreover, since large amounts of COD (70 percent) are associated with solids, it is expected that COD can increase with increased rain intensities.[1,8] As evident in figure 25 and figure 26, the rain intensities increased significantly during the periods of October, November, and December 2019, which can explain the elevation of COD in scupper drain samples with time progression for Sites SR 24 and SR 297.

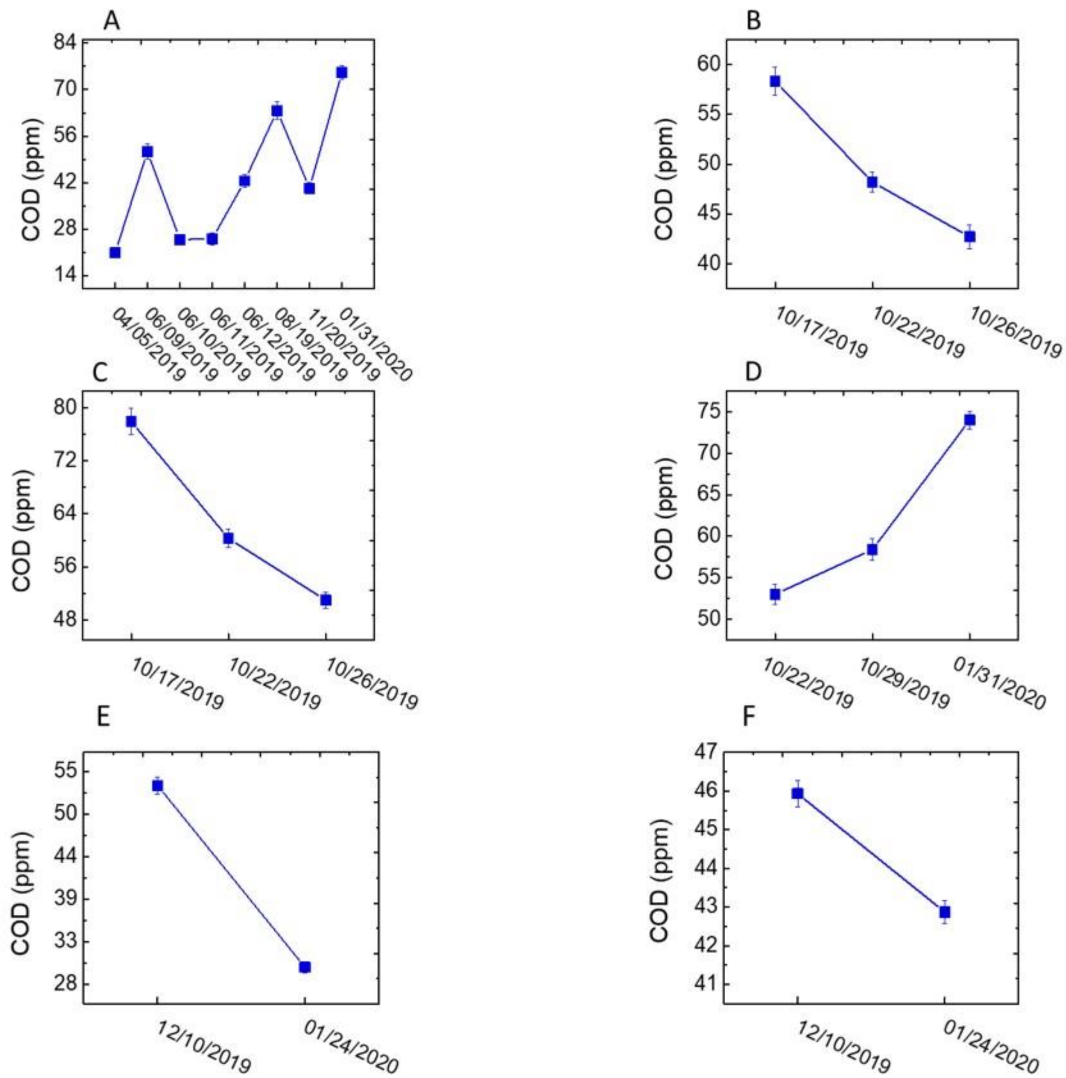

Figure 30. Line graphs. COD concentrations from scupper drains and for different dates for: (A) Site SR 24, (B) Site RF, (C) Site US 80, (D) Site SR 297, (E) Site US 255, and (F) Site SR 197.

For figure 30-B and -C, the decreasing trend can be attributed to a decreased accumulation of oxidizable content from traffic sources and/or more leaching of bridge deck material prior to the rainfall event for each specified date.

Both North Georgia sites, figure 30-E (Site US 255) and F (Site SR 197), show a declining trend of COD concentration, which can be attributed to less traffic volume and/or leaching of bridge deck material on 01/24/2020 as compared to 12/10/2019. The sum precipitation on 12/10/2019 was 0.11 inch and on 01/24/2020 it was 1.43 inches. The drop of COD concentration on 01/24/2020 can also be due to dilution of oxidizable content with the higher rain. It is known that contaminant concentrations, in general, decrease dramatically with increased rain intensity after the first flush of rainfall event.[1,10]

Figure 31 shows COD concentrations at time intervals for specific dates for all four Southeast Georgia sites and the two North Georgia sites. From figure 31, it is observed that COD shows a declining trend with time for all sites, which can be clearly attributed to the decrease of oxidizable content concentration after the first flush and with continuous wash off of the bridge deck surface. Contaminant concentration decreases sharply in scupper drain runoff after the first flush, which is the initial 30–40 min of the rainfall event, or can also be defined as the first 5–10 mm of rain. The drop of contaminant concentration, including COD, in the scupper drain after the first flush is further evidence of the introduction of such content into the instream water.

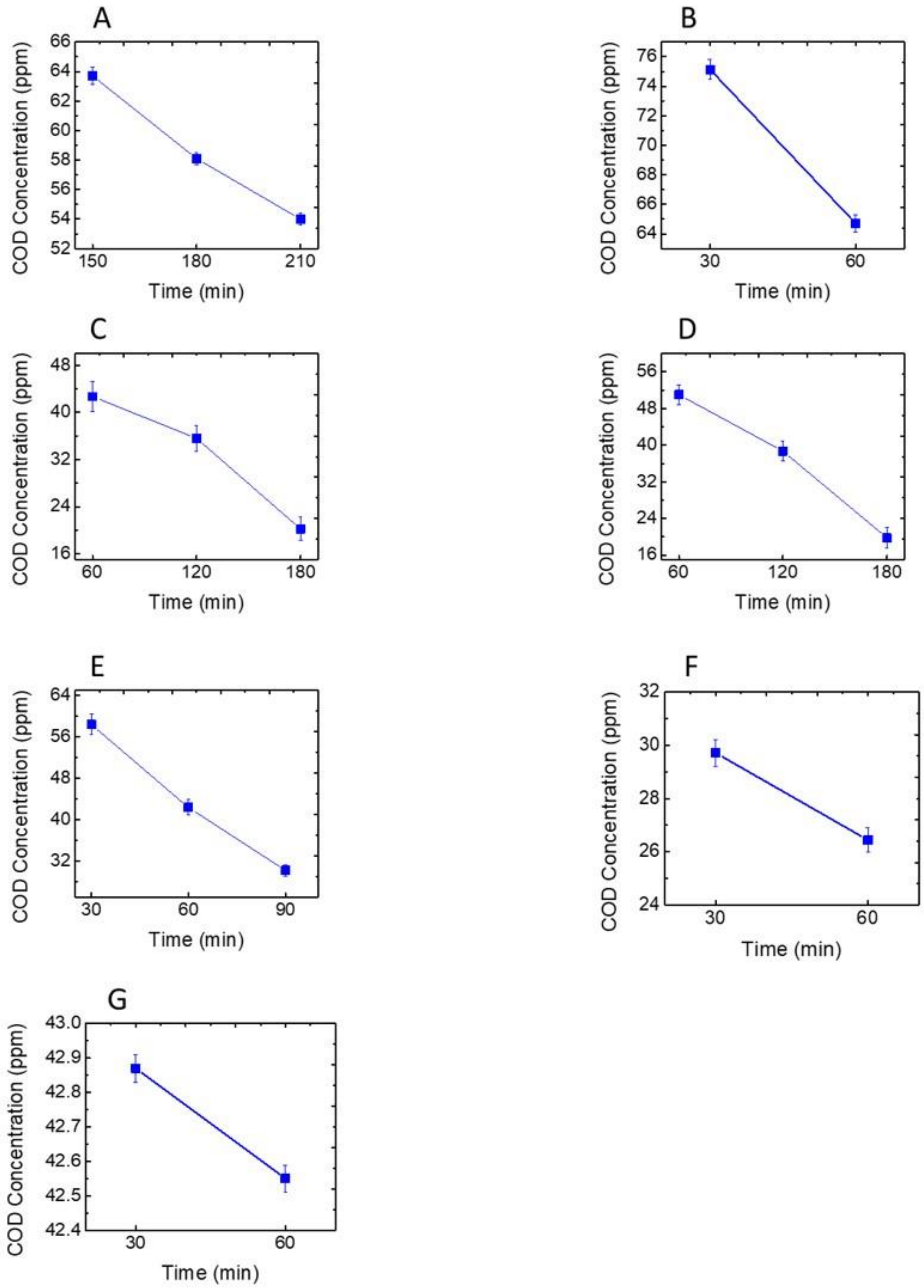

Figure 31. Line graphs. COD concentrations from scupper drains and at time intervals for: (A) Site SR 24 on 08/19/2019, (B) Site SR 24 on 01/31/2020, (C) Site RF on 10/26/2019, (D) Site US 80 on 10/26/2019, (E) Site SR 297 on 10/29/2019, (F) Site US 255 on 01/24/2020, and (G) Site SR 197 on 01/24/2020.

Summary of COD Results

COD levels in scupper drain runoff were found to be high, and the impact on instream water quality was observable for all six study sites. Downstream COD concentrations were elevated by the introduction of scupper drain runoff for all sites. COD concentrations were investigated in scupper drain runoff at different dates, and it was found that COD concentrations varied randomly with the different dates. These variations were attributed to increased humidity prior to rainfall, rainfall intensity, frequency of rains, and traffic volume. A consistent trend of decline in concentrations was observed in all cases in scupper drain runoff COD concentrations with sampling time, which was attributed to decrease of COD concentrations in scupper drain runoff after the first flush and with time due to dilution by rainwater.

NITROGEN

Total Nitrogen (TN)

TN was investigated in scupper drain runoff samples and instream samples, and it was found that nitrogen can sometimes be in excess in the scupper drain runoff. The major source of nitrogen in scupper drain runoff is atmospheric depositions, such as particulate matter and acid rain. A large array of pollutants can settle by adsorption on particulate matter surfaces in the atmosphere or on the ground and can be transported and/or introduced to aquatic environments, such as carbon monoxide; nitrogen oxides; volatile organic compounds and semi-VOCs; sulfur dioxide; methane; and toxins, including BTEX, butadiene, and formaldehyde, among others. Particulate matter can introduce these pollutants and others to the aquatic environment when they settle with precipitation (See references 1, 2, 12, 38, and 67). Figure 32 shows total nitrogen concentrations in scupper drain and instream samples at the four Southeast Georgia sites and the two North Georgia sites.

However, since other sources of nitrogen in instream water can be agricultural-related, such as fertilizers from surrounding boundary lands (see references 1, 5, 6, 8, 12, and 14), it was found that nitrogen concentration in instream water can usually surpass that in the scupper drain runoff. In figure 32, cases of both high nitrogen concentration in scupper drain runoff and in instream water can be observed. From figure 32-A (Site SR 24) and figure 32-D (Site SR 297), concentrations of nitrogen were found to be substantially higher in scupper drain runoff compared to upstream water. This observation confirms that scupper drain runoff can contribute to the introduction of nitrogen into instream water.

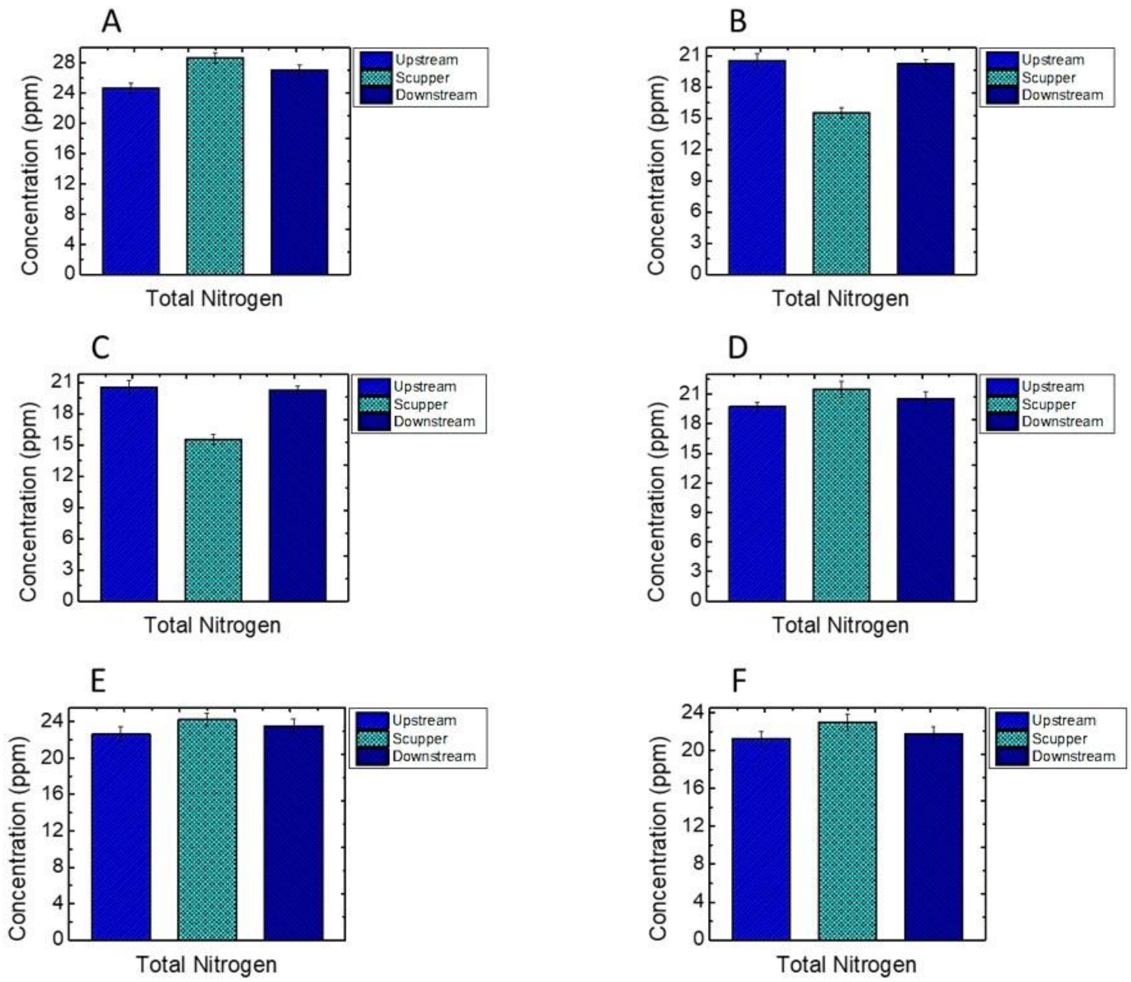

Figure 32. Bar graphs. Total nitrogen concentrations from upstream, scupper drains, and downstream for: (A) Site SR 24, (B) Site RF, (C) Site US 80, (D) Site SR 297, (E) Site US 255, and (F) Site SR 197. Sample dates were 01/31/2020 for Sites SR 24 and SR 297, 10/17/2019 for Site RF, 10/17/2019 for Site US 80, 12/10/2019 for Site US 255, and 01/24/2019 for Site SR 197.

The differences between downstream and upstream nitrogen concentration were found to be 2.36 ppm for SR 24 (figure 32-A) and approximately 1.00 ppm for SR 297 (figure 32-D), respectively. The higher nitrogen concentration in scupper drain runoff at Site SR 24 compared to Site SR 297 can be attributed to more atmospheric nitrogen that can be controlled by factors such as wind and rainfall intensity. However, such factors are variable, and atmospheric nitrogen concentrations can vary between the two sites, as shown in figure 50 in the appendix, which provides total nitrogen concentrations (total and dissolved) from upstream, scupper drains, and downstream for (A) Site SR 24 on 04/05/2019, (B) Site SR 24 on 11/20/2019, (C) Site RF on 10/22/2019, (D) Site US 80 on 10/17/2019, (E) Site SR 297 on 10/22/2019, (F) Site SR 297 on 10/29/2019, (G) Site US 255 on 01/24/2020, and (H) Site SR 197 on 12/10/2020. In figure 32-B and figure 32-C, nitrogen concentrations were found to be higher in upstream samples as compared to scupper drain runoff, which could be attributed to other sources of nitrogen, such as fertilizers from surrounding boundary lands and soil.

For the two North Georgia sites, Site US 255 (figure 32-E) and Site SR 197 (figure 32-F), it can also be seen that nitrogen concentrations were higher in scupper drain runoff as compared to upstream water, and the differences between downstream nitrogen concentration compared to upstream were found to be approximately 1.00 ppm for Site US 255 (figure 32-A) and 0.50 ppm for Site SR 197 (figure 32-B), respectively.

Figure 33 shows total nitrogen concentrations from scupper drains for all four Southeast Georgia sites and the two North Georgia sites from consecutive dates. Figure 33-A (Site SR 24) and figure 33-B (Site RF) show an overall decline of total nitrogen concentration across the different dates, except on 01/31/2020 for figure 33-A, which shows a sharp increase in total nitrogen concentration for Site SR 24. Such behavior reflects that it is hard to predict the nitrogen presence in scupper drain runoff due to uncontrollable factors, such as wind speed and rainfall intensity, that control the deliverance of atmospheric nitrogen to the bridge deck.

The overall declining nitrogen trend in figure 33-A and figure 33-B can be due to continuous wash-off of the bridge deck surface with rain relative to low surface accumulation over days. However, the sharp rise in nitrogen concentration on 01/31/2020 for figure 33-A can be due to an opposite effect of the introduction of large amounts of atmospheric nitrogen with rainfall at that site. Both figure 33-C (Site US 80) and figure 33-D (Site SR 297) show an increase of nitrogen concentration with consecutive days, which indicates increasing introduction of nitrogen to the bridge deck, especially for figure 33-C. For figure 33-D, a plateau can be observed between the months of October 2019 and January 2020, indicating that nitrogen introduction to the bridge deck surface was low during this period. Only a 0.24-ppm (240.00-ppb) nitrogen increase was found during that period in the scupper drain runoff at Site SR 297.

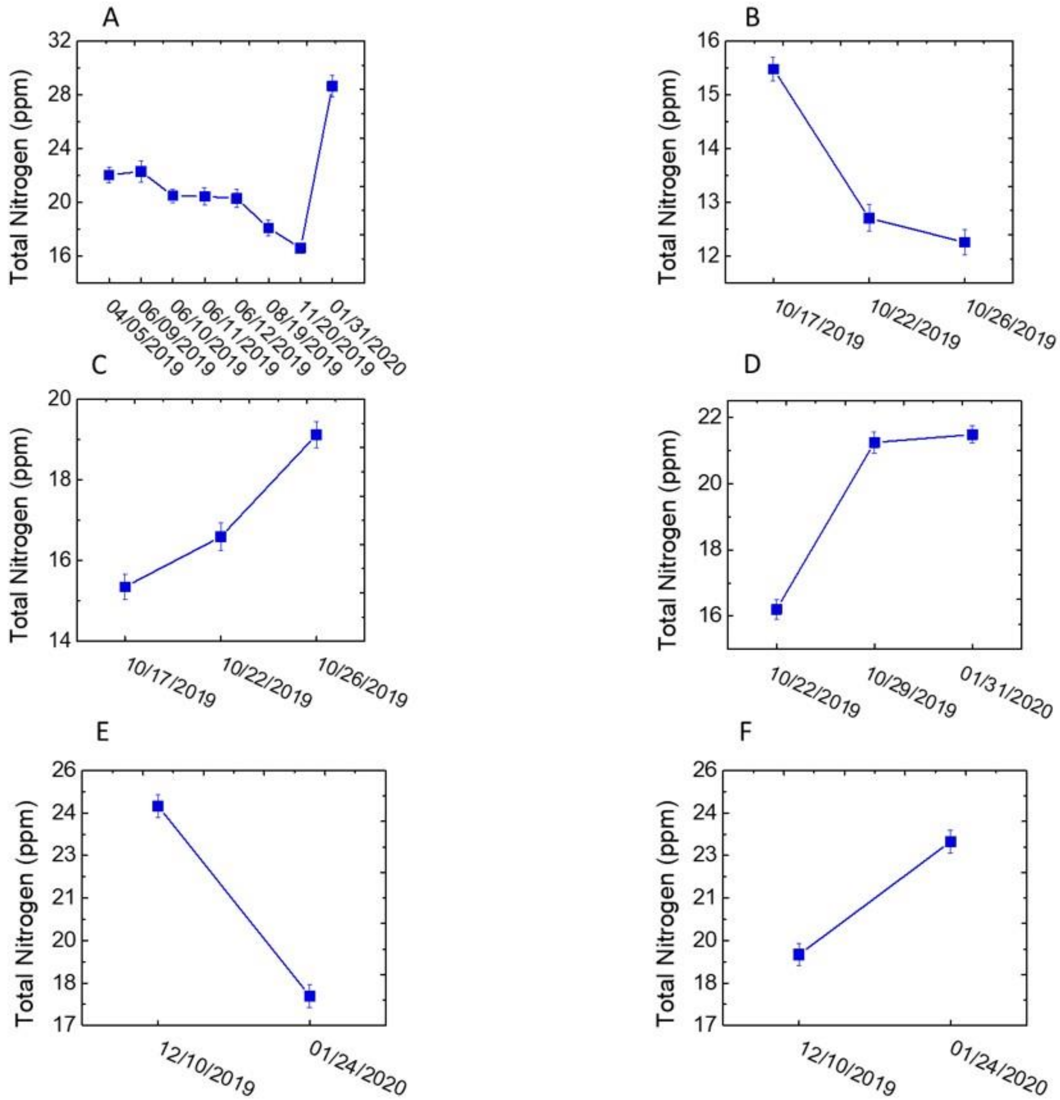

Figure 33. Line graphs. Total nitrogen concentrations from scupper drains and for different dates for: (A) Site SR 24, (B) Site RF, (C) Site US 80, (D) Site SR 297, (E) Site US 255, and (F) Site SR 197.

Figure 33-E and figure 33-F show total nitrogen concentrations from the scupper drains for the two North Georgia sites from consecutive dates. The opposite trends of sharp decline in nitrogen concentrations for figure 33-E compared to the sharp increase for figure 33-F confirm that scupper drain runoff nitrogen trends are unpredictable and are controlled by random factors, such as wind speed and rainfall intensity. Total nitrogen concentrations can be significantly higher

in bridge deck runoff compared to highway runoff, which can be clearly attributed to the absence of bridge deck runoff natural filtration by side vegetation and swales and soil, unlike highway runoff.[1,15]

Figure 34 shows total nitrogen concentrations at time intervals for specific dates for all four Southeast Georgia sites and the two North Georgia sites. From figure 34, it is observed that total nitrogen declined with time for all sites, which can be clearly attributed to the decrease of nitrogen concentration after the first flush and with continuous wash-off of the bridge deck surface. It is noticeable that the total nitrogen trend for the two North Georgia sites follows the same declining trend with time, as well. As mentioned previously, contaminant concentrations decrease sharply in scupper drain runoff after the first flush, which is the initial 30–40 min of the rainfall event, or can also be defined as the first 5–10 mm of rain.[1,10] The drop in contaminant concentration, including total nitrogen, in scupper drain runoff after the first flush is further evidence of the introduction of nitrogen into the instream water.

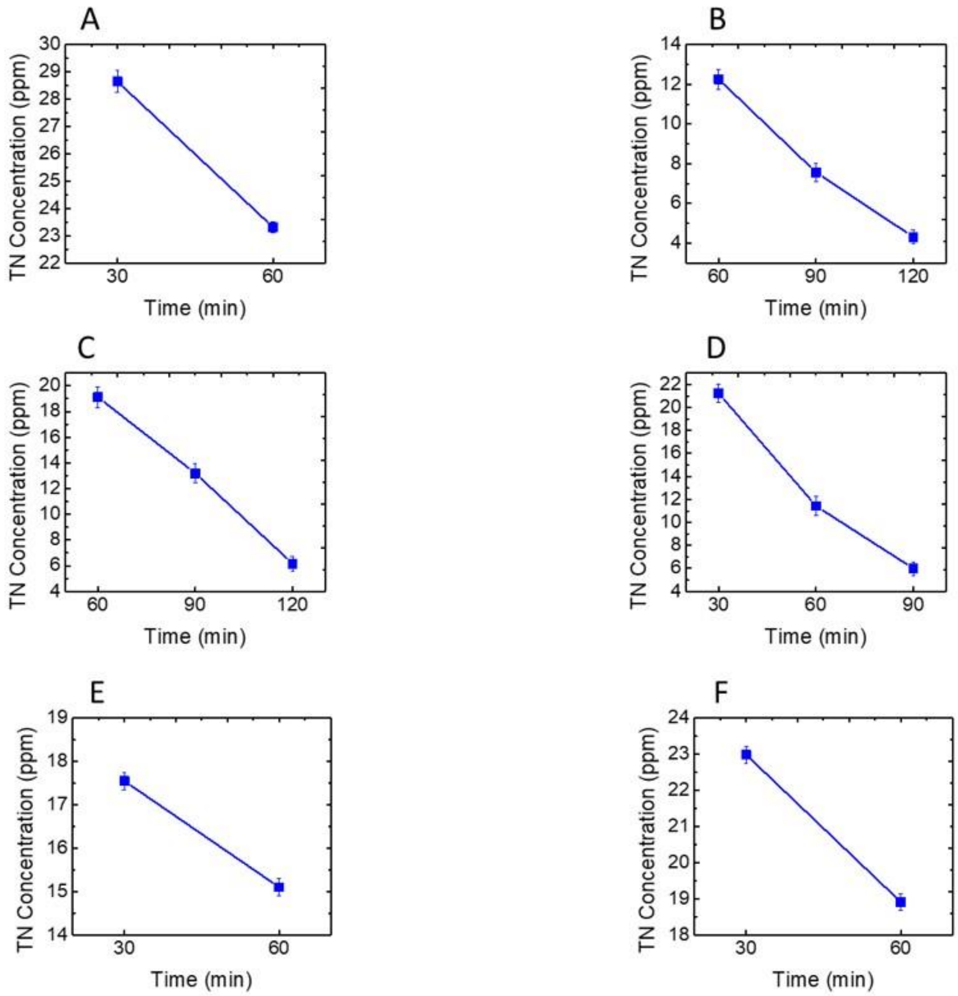

Figure 34. Line graphs. Total nitrogen concentrations from scupper drains and at time intervals for: (A) Site SR 24 on 01/31/2020, (B) Site RF on 10/26/2019, (C) Site US 80 on 10/26/2019, (D) Site SR 297 on 10/29/2019, (E) Site US 255, and (F) Site SR 197 on 01/24/2020.

Total Kjeldahl Nitrogen (TKN)

TKN is a measure of ammonia and organic nitrogen in scupper drain runoff and instream water. Nitrogen, in general, is a nutrient to bacteria, plants, and algae on which these creatures rely for growth and reproduction. The presence of large quantities of nutrients in water bodies can have a harmful effect by creating a nuisance environment and the proliferation of surface water plants, leading to the consumption of large amounts of dissolved oxygen, which subsequently depletes in water, leading to the death of aquatic creatures.[1,38] Thus, it is critically important to monitor the amounts of nitrogen being introduced to instream waters from side boundaries and bridge deck runoff.

In this subsection and the following subsections, average concentrations of different types of nitrogen constituents present in aquatic environments were monitored from instream and scupper drain runoff samples, aiming to observe nitrogen concentrations in its different forms. Table 12 shows TKN concentrations from all six Georgia study sites from different dates and subsequent time intervals, using samples from upstream, scupper drain runoff, and downstream. From table 12, TKN follows trends closely resembling that of total nitrogen reported in the previous subsection. The impact of scupper drain runoff could still be observed by the introduction of higher TKN concentrations relative to the upstream TKN concentrations, which could be introduced from bridge traffic in addition to atmospheric deposition. (See references 1, 5, 6, 8, 12, and 14) However, the unpredictability of TKN concentration is also observed due to other nitrogen sources, such as fertilizers and soil-related sources from roadside ditches along a highway.

The declining TKN concentrations in scupper drain runoff with time are also observed due to the continued wash-off of the bridge deck surface with rainfall, which is the case with contaminant behavior during a rainfall event after the first flush, as mentioned previously.

Table 12. TKN for all six Georgia study sites from instream and scupper drain runoff and at different dates and time intervals.

TKN (Con. Unit ppm [mg N/L])			
Date	**Upstream**	**Scupper**	**Downstream**
Site SR 24			
04-05-19	13.70±0.21	14.25±0.34	14.00±0.36
06-09-19	Not sampled	13.40±0.28	Not sampled
06-10-19	Not sampled	10.60±0.15	Not sampled
06-11-19	Not sampled	10.95±0.20	Not sampled
06-12-19	Not sampled	15.80±0.45	Not sampled
08-19-19	Not sampled	9.80±0.11	Not sampled
11-20-19	12.20±0.19	9.40±0.09	11.80±0.13
1-31-20 (30 min)	14.30±0.30	17.28±0.40	16.44±0.37
1-31-20 (60 min)	Not sampled	15.70±0.35	Not sampled
Average	**13.40±0.23**	**13.02±0.26**	**14.08±0.29**
Site Rocky Ford (RF)			
10-17-19	11.20±0.12	9.74±0.10	10.66±0.15
10-22-19	12.20±0.30	8.14±0.07	9.80±0.10
10-26-19 (60 min)	Not sampled	6.85±0.08	Not sampled
10-26-19 (90 min)	Not sampled	4.95±0.05	Not sampled
10-26-19 (120 min)	Not sampled	3.16±0.04	Not sampled
Average	**11.7±0.21**	**6.57±0.07**	**10.23±0.13**
Site US 80			
10-17-19	15.17±0.35	6.38±0.08	13.74±0.13
10-22-19	13.70±0.24	9.23±0.12	11.42±0.10
10-26-19 (60 min)	Not sampled	10.24±0.12	Not sampled
10-26-19 (90 min)	Not sampled	8.17±0.08	Not sampled
10-26-19 (120 min)	Not sampled	3.77±0.04	Not sampled
Average	**14.44±0.30**	**7.56±0.09**	**12.58±0.12**
Site SR 297			
10-22-19	13.87±0.40	10.74±0.17	12.33±0.20
10-29-19 (30 min)	16.80±0.50	12.81±0.22	13.74±0.30
10-29-19 (60 min)	Not sampled	8.21±0.05	Not sampled
10-29-19 (90 min)	Not sampled	4.87±0.02	Not sampled
01-31-20	11.19±0.15	12.21±0.21	12±0.23
Average	**13.95±0.35**	**9.77±0.13**	**12.69±0.24**
Site US 255			
12-10-19	10.98±0.11	11.24±0.14	11.01±0.12
01-24-20 (30 min)	9.18±0.09	9.47±0.10	9.41±0.10
01-24-20 (60 min)	Not sampled	8.77±0.10	Not sampled
Average	**10.08±0.10**	**9.83±0.11**	**10.21±0.11**
Site SR 197			
12-10-19	9.86±0.10	8.51±0.09	9.4±0.08
01-24-20 (30 min)	12.34±0.51	14.84±0.40	12.98±0.14
01-24-20 (60 min)	Not sampled	12.45±0.14	Not sampled
Average	**11.10±0.31**	**11.93±0.21**	**11.19±0.11**

Ammonia Nitrogen (NH_3-N)

Table 13 shows ammonia nitrogen concentrations from all six study sites from different dates and subsequent time intervals, using samples from upstream, scupper drain runoff, and downstream.

From table 13, ammonia nitrogen followed trends closely resembling that of total nitrogen reported in the previous subsection. The impact of scupper drain runoff could still be observed by the introduction of higher ammonia nitrogen concentrations relative to upstream ammonia nitrogen concentrations that could be introduced from atmospheric deposition. However, the unpredictability of ammonia nitrogen concentration was also observed due to other nitrogen sources, such as fertilizers and soil-related sources from roadside ditches along a highway.

The declining ammonia nitrogen concentrations in scupper drain runoff with time were also observed due to the continued wash-off of the bridge deck surface with rainfall, which was the case with contaminant behavior during a rainfall event after the first flush, as mentioned previously.

Table 13. Ammonia nitrogen for all six Georgia study sites from instream and scupper drain runoff and at different dates and time intervals.

Ammonia Nitrogen (Con. Unit ppm [mg/L-N])			
Date	**Upstream**	**Scupper**	**Downstream**
Site SR 24			
04-05-19	13.85±0.20	13.25±0.23	13.25±0.25
06-09-19	Not sampled	13.40±0.20	Not sampled
06-10-19	Not sampled	10.60±0.10	Not sampled
06-11-19	Not sampled	10.95±0.15	Not sampled
06-12-19	Not sampled	15.8±0.40	Not sampled
08-19-19	Not sampled	8.00±0.06	Not sampled
11-20-19	12.18±0.17	11.70±0.10	11.53±0.12
01-31-20 (30 min)	9.88±0.10	16.25±0.50	13.46±0.14
01-31-20 (60 min)	Not sampled	14.00±0.35	Not sampled
Average	**11.97±0.16**	**12.66±0.23**	**12.75±0.17**
Site Rocky Ford (RF)			
10-17-19	13.10±0.32	13.69±0.16	13.30±0.14
10-22-19	10.80±0.13	11.34±0.10	11.16±0.11
10-26-19 (60 min)	Not sampled	8.79±0.09	Not sampled
10-26-19 (90 min)	Not sampled	6.50±0.07	Not sampled
10-26-19 (120 min)	Not sampled	4.00±0.03	Not sampled
Average	**11.95±0.23**	**8.86±0.09**	**12.23±0.13**
Site US 80			
10-17-19	8.80±0.07	10.00±0.15	9.15±0.10
10-22-19	9.12±0.10	9.22±0.08	9.13±0.08
10-26-19 (60 min)	Not sampled	7.62±0.06	Not sampled
10-26-19 (90 min)	Not sampled	5.14±0.04	Not sampled
10-26-19 (120 min)	Not sampled	3.90±0.02	Not sampled
Average	**8.96±0.09**	**7.18±0.07**	**9.14±0.09**
Site SR 297			
10-22-19	9.89±0.07	11.20±0.13	10.71±0.10
10-29-19 (30 min)	10.39±0.13	12.47±0.14	12.01±0.14
10-29-19 (60 min)	Not sampled	7.16±0.05	Not sampled
10-29-19 (90 min)	Not sampled	3±0.02	Not sampled
01-31-20	9.78	11.5±0.11	11.07±0.12
Average	**10.02±0.12**	**9.07±0.09**	**11.26±0.12**
Site US 255			
12-10-19	6.48±0.04	8.73±0.08	8.22±0.06
01-24-20 (30 min)	8.27±0.07	6.15±0.05	8.46±0.05
01-24-20 (60 min)	Not sampled	5.37±0.03	Not sampled
Average	**7.38±0.06**	**6.75±0.05**	**8.34±0.06**
Site SR 197			
12-10-19	10.27±0.10	12.19±0.15	11.00±0.12
01-24-20 (30 min)	8.19±0.09	6.97±0.07	7.68±0.08
01-24-20 (60 min)	Not sampled	6.22±0.06	Not sampled
Average	**9.23±0.10**	**8.46±0.09**	**9.34±0.10**

Nitrate Nitrogen (NO_3^--N)

Table 14 shows nitrate nitrogen concentrations from all six study sites from different dates and subsequent time intervals, using samples from upstream, scupper drain runoff, and downstream.

From table 14, nitrate nitrogen followed trends closely resembling that of total nitrogen reported in the previous subsection. The impact of scupper drain runoff could still be observed by the introduction of higher nitrate nitrogen concentrations relative to upstream nitrate nitrogen concentrations that could be introduced from atmospheric deposition. However, the unpredictability of nitrate nitrogen concentration was also observed due to other nitrogen sources, such as fertilizers and soil-related sources from roadside ditches along a highway, in addition to oxidative and/or anoxic conditions.

The declining nitrate nitrogen concentrations in scupper drain runoff with time were also observed due to the continued wash-off of the bridge deck surface with rainfall, which was the case with contaminant behavior during a rainfall event after the first flush, as mentioned previously.

Table 14. Nitrate nitrogen for all six Georgia study sites from instream and scupper drain runoff and at different dates and time intervals.

Nitrate Nitrogen (Con. Unit ppm [mg/L-N])			
Date	**Upstream**	**Scupper**	**Downstream**
Site SR 24			
04-05-19	8.85±0.05	7.60±0.03	10.55±0.10
06-09-19	Not sampled	4.85±0.03	Not sampled
06-10-19	Not sampled	5.40±0.02	Not sampled
06-11-19	Not sampled	6.35±0.04	Not sampled
06-12-19	Not sampled	4.38±0.03	Not sampled
08-19-19	Not sampled	7.20±0.04	Not sampled
11-20-19	6.17±0.04	7.88±0.05	7.17±0.06
01-31-20 (30 min)	8.57±0.05	8.94±0.07	8.58±0.07
01-31-20 (60 min)	Not sampled	6.38±0.06	Not sampled
Average	**7.86±0.05**	**6.55±0.04**	**8.77±0.08**
Site Rocky Ford (RF)			
10-17-19	7.52±0.06	7.86±0.08	7.60±0.03
10-22-19	6.63±0.04	7.02±0.07	6.80±0.03
10-26-19 (60 min)	Not sampled	6.27±0.04	Not sampled
10-26-19 (90 min)	Not sampled	4.30±0.03	Not sampled
10-26-19 (120 min)	Not sampled	2.18±0.01	Not sampled
Average	**7.08±0.05**	**5.53±0.05**	**7.20±0.03**
Site US 80			
10-17-19	6.66±0.06	8.05±0.07	7.98±0.06
10-22-19	5.87±0.04	7.82±0.07	6.74±0.05
10-26-19 (60 min)	Not sampled	5.44±0.05	Not sampled
10-26-19 (90 min)	Not sampled	2.19±0.02	Not sampled
10-26-19 (120 min)	Not sampled	0.00±0.00	Not sampled
Average	**6.27±0.05**	**4.70±0.05**	**7.36±0.06**
Site SR 297			
10-22-19	7.29±0.05	3.87±0.03	8.14±0.06
10-29-19 (30 min)	8.08±0.08	6.54±0.04	6.20±0.04
10-29-19 (60 min)	Not sampled	2.50±0.01	Not sampled
10-29-19 (90 min)	Not sampled	0.57±0.002	Not sampled
01-31-20	5.77±0.04	5.54±0.03	5.37±0.04
Average	**7.05±0.06**	**3.80±0.02**	**6.57±0.05**
Site US 255			
12-10-19	6.13±0.04	7.00±0.06	6.90±0.05
01-24-20 (30 min)	4.64±0.02	3.87±0.03	4.52±0.03
01-24-20 (60 min)	Not sampled	3.14±0.02	Not sampled
Average	**5.39±0.03**	**4.67±0.04**	**5.71±0.04**
Site SR 197			
12-10-19	5.39±0.03	7.80±0.07	6.55±0.06
01-24-20 (30 min)	5.20±0.02	4.32±0.03	5.00±0.04
01-24-20 (60 min)	Not sampled	3.59±0.03	Not sampled
Average	**5.30±0.03**	**5.24±0.04**	**5.78±0.05**

Nitrite Nitrogen (NO_2^--N)

Table 15 shows nitrite nitrogen concentrations from all six study sites from different dates and subsequent time intervals, using samples from upstream, scupper drain runoff, and downstream.

From table 15, nitrite nitrogen followed trends closely resembling that of total nitrogen reported in the previous subsection. The impact of scupper drain runoff could still be observed by the introduction of higher nitrite nitrogen concentrations relative to upstream nitrite nitrogen concentrations that could be introduced from atmospheric deposition. However, the unpredictability of nitrite nitrogen concentration was also observed due to other nitrogen sources, such as fertilizers and soil-related sources from roadside ditches along a highway, in addition to oxidative and/or anoxic conditions.

The declining nitrite nitrogen concentrations in scupper drain runoff with time was also observed due to the continued wash-off of the bridge deck surface with rainfall, which was the case with contaminant behavior during rainfall event after the first flush, as mentioned previously.

Table 15. Nitrite nitrogen for all six Georgia study sites from instream and scupper drain runoff and at different dates and time intervals.

Nitrite Nitrogen (Con. Unit ppm [mg/L-N])			
Date	**Upstream**	**Scupper**	**Downstream**
Site SR 24			
04-05-19	0.05±0.00	0.20±0.00	0.20±0.00
06-09-19	Not sampled	4.05±0.03	Not sampled
06-10-19	Not sampled	4.50±0.04	Not sampled
06-11-19	Not sampled	3.15±0.02	Not sampled
06-12-19	Not sampled	0.11±0.01	Not sampled
08-19-19	Not sampled	1.10±0.01	Not sampled
11-20-19	0.00±0.00	0.00±0.00	0.00±0.00
01-31-20 (30 min)	1.76±0.01	2.43±0.02	2.00±0.01
01-31-20 (60 min)	Not sampled	1.23±0.00	Not sampled
Average	**0.60**±0.00	**1.86±0.02**	**0.73±0.01**
Site Rocky Ford (RF)			
10-17-19	1.97±0.01	0.00±0.00	1.79±0.01
10-22-19	1.05±0.01	0.00±0.00	0.17±0.01
10-26-19 (60 min)	Not sampled	0.00±0.00	Not sampled
10-26-19 (90 min)	Not sampled	0.00±0.00	Not sampled
10-26-19 (120 min)	Not sampled	0.00±0.00	Not sampled
Average	**1.51±0.01**	**0.00**±0.00	**0.98±0.01**
Site US 80			
10-17-19	0.00±0.00	0.99±0.00	0.00±0.00
10-22-19	3.32±0.02	0.00±0.00	0.92±0.00
10-26-19 (60 min)	Not sampled	3.44±0.02	Not sampled
10-26-19 (90 min)	Not sampled	2.84±0.01	Not sampled
10-26-19 (120 min)	Not sampled	2.39±0.02	Not sampled
Average	**3.32±0.02**	**1.71±0.02**	**0.92±0.00**
Site SR 297			
10-22-19	2.85±0.01	1.60±0.01	2.49±0.02
10-29-19 (30 min)	2.53±0.01	1.89±0.01	1.87±0.01
10-29-19 (60 min)	Not sampled	0.72±0.01	Not sampled
10-29-19 (90 min)	Not sampled	0.55±0.00	Not sampled
01-31-20	2.76±0.01	3.73±0.02	3.17±0.02
Average	**2.71±0.01**	**1.70±0.01**	**2.51±0.02**
Site US 255			
12-10-19	5.54±0.04	6.00±0.05	5.65±0.04
01-24-20 (30 min)	2.78±0.01	4.21±0.04	4.11±0.03
01-24-20 (60 min)	Not sampled	3.19±0.02	Not sampled
Average	**4.16±0.03**	**4.47±0.04**	**4.88±0.04**
Site SR 197			
12-10-19	4.50±0.03	2.70±0.01	3.65±0.03
01-24-20 (30 min)	3.72±0.01	3.83±0.02	3.74±0.03
01-24-20 (60 min)	Not sampled	2.87±0.02	Not sampled
Average	**4.11±0.02**	**3.13±0.02**	**3.70±0.03**

Summary of TN Results

TN was found to be introduced from scupper drains with observable impact on instream water quality. Downstream TN concentrations were elevated by the introduction of scupper drain runoff for most sites. TN concentrations were investigated in scupper drain runoff at different dates, and it was found that TN concentrations varied randomly with different dates. These variations were attributed to increased nitrogen load from scupper drains due to accumulation on the bridge deck due to atmospheric deposition and by the introduction of bridge deck particulate matter. A consistent trend of decline in TN concentrations was observed in all cases in scupper drain runoff with sampling time, which was attributed to the decrease of TN concentrations in scupper drain runoff after the first flush and with time due to dilution by rainwater.

PHOSPHORUS (PO_4^{3-}-P)

Like nitrogen, phosphorus in its aqueous form (phosphate) is a nutrient to bacteria, aquatic plants, and algae. Therefore, its presence in excess in surface waters can create the same harmful and nuisance conditions that threaten the stability of aquatic biota. Thus, monitoring phosphorus in water bodies is of critical importance. Phosphorus was measured as total phosphate and expressed as PO_4^{3-}-P in water samples from bridge scupper drains and instream water. Figure 35 shows PO_4^{3-}-P concentrations in scupper drain runoff and instream samples at the four Southeast Georgia study sites for upstream, scupper drain runoff, and downstream. For both North Georgia study sites, phosphorus was undetected.

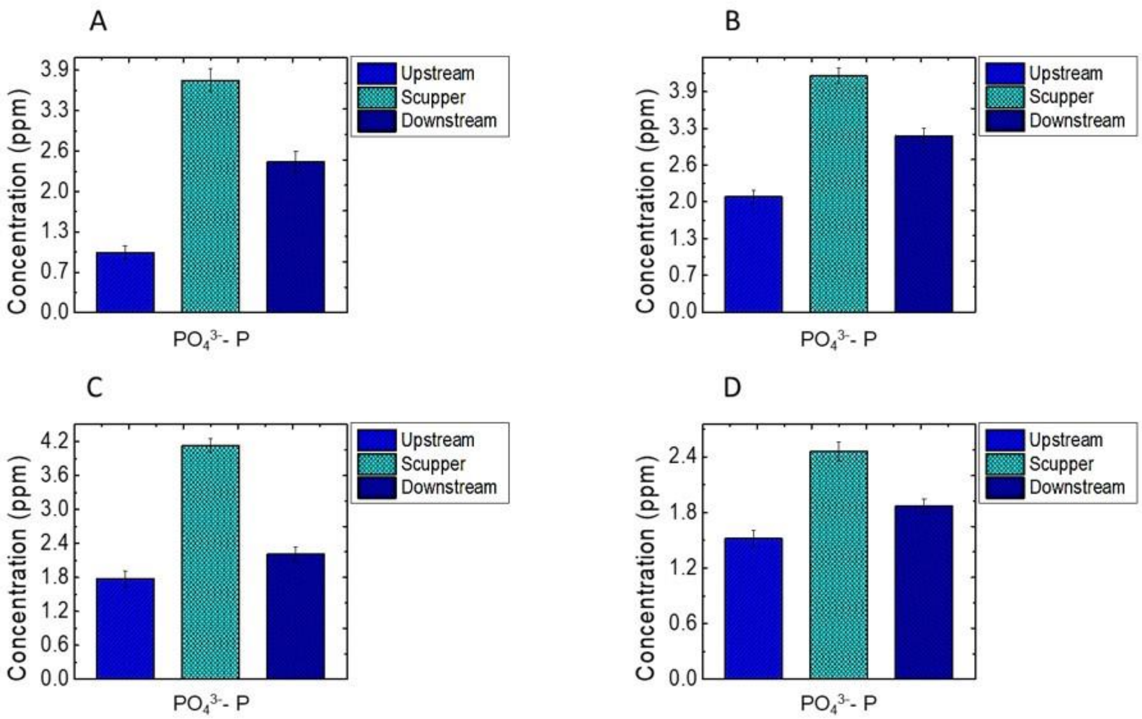

Figure 35. Bar graphs. PO_4^{3-}-P concentrations from upstream, scupper drains, and downstream for: (A) Site SR 24, (B) Site RF, (C) Site US 80, and (D) Site SR 297. Sample dates were 01/31/2020 for Site SR 24 and 10/22/2019 for Sites RF, US 80 and SR 297, respectively.

Major sources of phosphorus are atmospheric depositions and fertilizers, such as nitrogen. Phosphorus resembles nitrogen in its sources and impacts, and, thus, it is expected that phosphorus in scupper drain runoff is from accumulation of atmospheric depositions and the contribution of traffic. (See references 1, 5, 6, 8, 12, and 14) From figure 35, the concentrations of PO_4^{3-}-P in scupper drain runoff are higher than upstream PO_4^{3-}-P concentrations. Why the scupper drain PO_4^{3-}-P concentrations were much higher as compared to upstream is unclear; however, it is of alarming concern. The differences between downstream and upstream phosphate concentrations were found to be 1.50, 1.00, 0.40, and 0.35 ppm for Sites SR 24, RF, US 80, and SR 297, respectively.

Since atmospheric deposition is the key factor for bridge deck contamination with phosphorus, it is expected that the combination of both factors of rainfall (introduction) and dry periods (accumulation) can have a significant effect on the increasing phosphorus amounts on the bridge deck and, thus, its concentration in scupper drain runoff. The longer the dry period, the more phosphorus accumulation on the bridge deck surface from various sources, and the higher the rainfall intensity, the more the atmospheric deposition on the bridge deck surface. Moreover, higher rainfall intensity plays both effects of the introduction of more phosphorus to the bridge deck surface and the dilution of scupper drain runoff contaminants, including phosphorus.

Figure 36 shows PO_4^{3-}-P concentrations in scupper drain runoff for all four Southeast Georgia sites from consecutive dates. From figure 36, the variation of PO_4^{3-}-P concentrations seems random, which confirms previous statements on the unpredictability of nutrient amounts in scupper drain runoff that relies on different variables, including rain frequency, rainfall intensity, dry periods, and traffic volume. The sharp increases in scupper PO_4^{3-}-P concentration in figure 36-A, figure 36-C, and figure 36-D at the later dates could be due to a large bridge deck surface PO_4^{3-}-P accumulation from traffic sources during the preceding dry periods and/or the introduction of large amounts of PO_4^{3-}-P during the rainfall events at the times of sampling.

Figure 37 shows PO_4^{3-}-P concentrations at time intervals for specific dates for all four Southeast Georgia sites. From figure 37, it is observed that phosphate declined with time for all four Southeast Georgia sites, which can be clearly attributed to the decrease of PO_4^{3-}-P concentration after the first flush and with continuous wash-off of the bridge deck surface. This sheds light on the high diluting impact of rain, even though rainfall is known to be the major source of introduction of phosphorus to the bridge deck surface.

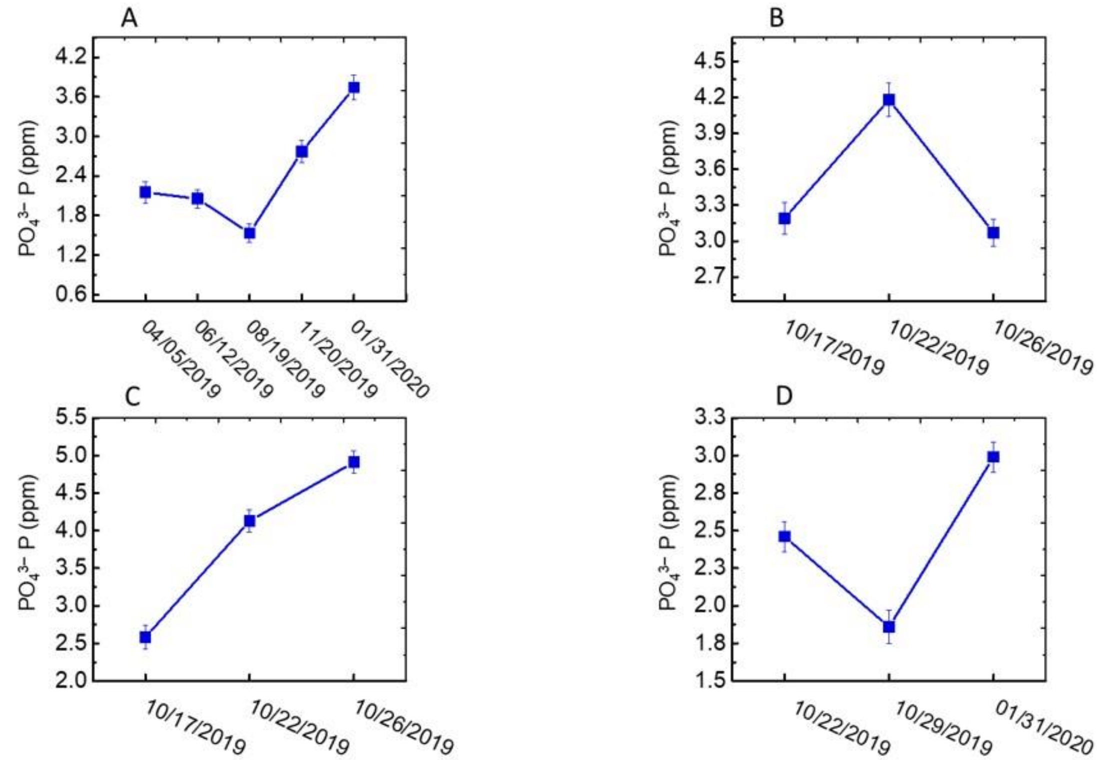

Figure 36. Line graphs. PO_4^{3-}-P concentrations from scupper drains and for different dates for: (A) Site SR 24, (B) Site RF, (C) Site US 80, and (D) Site SR 297.

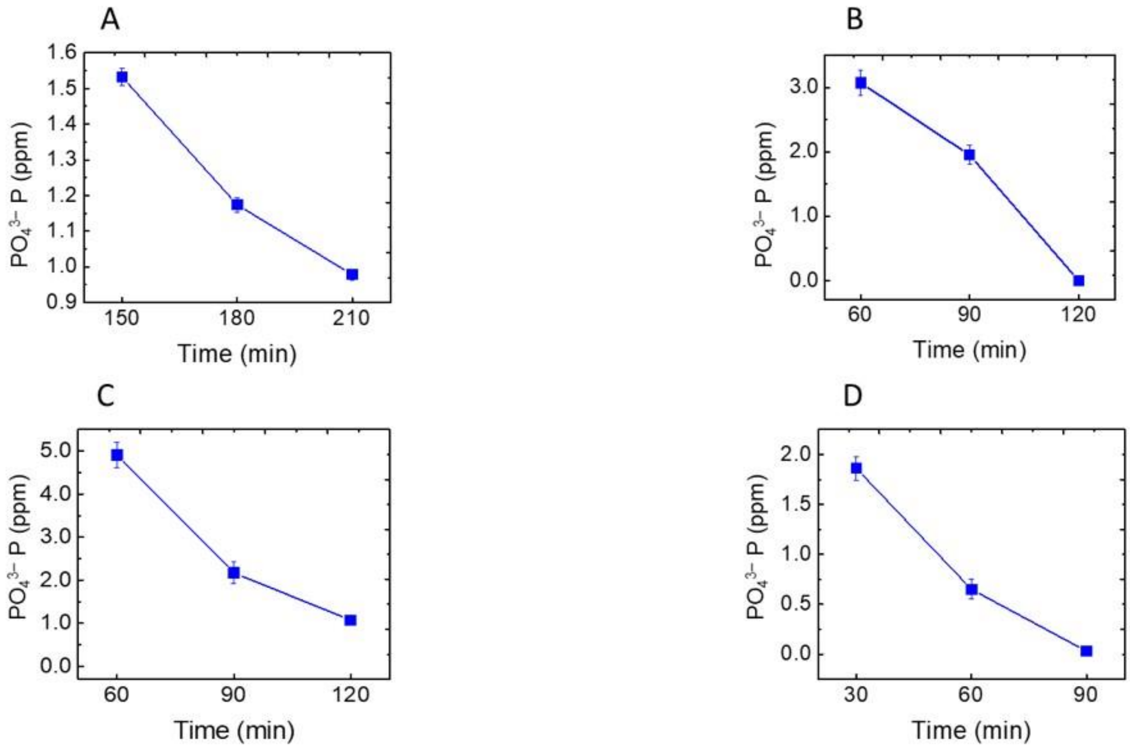

Figure 37. Line graphs. PO_4^{3-}-P concentrations from scupper drains and at time intervals for: (A) Site SR 24 on 08/19/2019, (B) Site RF on 10/26/2019, (C) Site US 80 on 10/26/2019, and (D) Site SR 297 on 10/29/2019.

Summary of PO_4^{3-}-P Results

PO_4^{3-}-P was found to be introduced from scupper drains with observable impact on instream water quality. Downstream PO_4^{3-}-P concentrations were elevated by the introduction of scupper drain runoff for all four Southeast Georgia sites, while PO_4^{3-}-P was undetected at both North Georgia sites. PO_4^{3-}-P concentrations were investigated in scupper drain runoff at different dates, and it was found that PO_4^{3-}-P concentrations varied randomly with different dates. These variations were attributed to the unpredictability of nutrient amounts in scupper drain runoff that rely on different variables, including rain frequency, rainfall intensity, dry periods, and traffic volume. A consistent trend of declines in concentrations were observed in all cases in scupper drain runoff PO_4^{3-}-P concentrations with sampling time, which was attributed to a decrease of PO_4^{3-}-P concentrations in the scupper drain runoff after the first flush and with time due to dilution by rainwater.

SOLIDS

Total Solids (TS)

Solids exist in stormwater runoff, and their sources are numerous. Major sources of solids in bridge stormwater runoff include atmospheric fallout, dust fall, rehabilitation work, construction and maintenance activities, and pavement wear (See references 1, 3, 5, 6, 8–10, 12, 14, and 17). Hydrologic and climatic conditions control the amount of solids loading; for instance, elevated stream discharge, rainfall intensity, and dry periods can increase solids loading, while foregoing storm intensity can decrease solids loading.[1,8] High-intensity storms and high rainfall volume can transport around 60 percent of solids from different sources, including the bridge deck into surface waters, and low-intensity storms can transport around 15 percent in the first 4 min of the storm event.[1,8]

The environmental concern with solids is that they can transport an array of hazardous contaminants into surface waters. Several organic/inorganic contaminants can attach to solids surfaces of different sizes and can be transported into water bodies. Suspended or settleable solid can convey up to 70 percent of COD content in stormwater runoff, and since more than half of suspended solids are constituted of inorganic content, suspended solids can carry and/or contain high concentrations of toxic heavy metals. Therefore, solids can introduce different harmful contaminants to aquatic ecosystems and can increase sediment levels in water bodies, leading to burial of aquatic creatures.[1]

Figure 38 shows total solids concentrations in scupper drain runoff and instream samples at the four Southeast and the two North Georgia sites for upstream, scupper drain runoff, and downstream. From figure 38, it is observed that scupper drain runoff solids can add to instream solids loading, threatening the aquatic ecosystem by the introduction of different pollutants and increasing stream sediment level with time. The differences between downstream and upstream solids concentrations in figure 38-A, -B, -C, and -D were found to be 292.00, 74.00, 114.00, and 89.00 ppm for Sites SR 24, RF, US 80 and SR 297, respectively. These concentration differences are observable and raise concerns for the impact of solids on stream waters. Since suspended solids form the larger portion of total solids, it is expected that the introduction of heavy metals and other inorganic toxins from the bridge deck surface to the instream water is significant; moreover, the sedimentation would be higher at the stream bed.

The introduction of solids into stream waters leads to the transportation of various pollutants downstream as rainfall can mobilize and introduce solids from the bridge deck surface into the instream water.[1,10] Figure 38-E and -F show total solids concentrations in scupper drain runoff and instream samples at the two North Georgia sites for upstream, scupper drain runoff, and downstream. From figure 38-E and -F , it is also evident that introduction of bridge deck solids into the instream increases instream solids loading. The differences between downstream and upstream solids concentrations in figure 38-E and -F were found to be 170.00 and 95.00 ppm for Sites US 255 and SR 197, respectively.

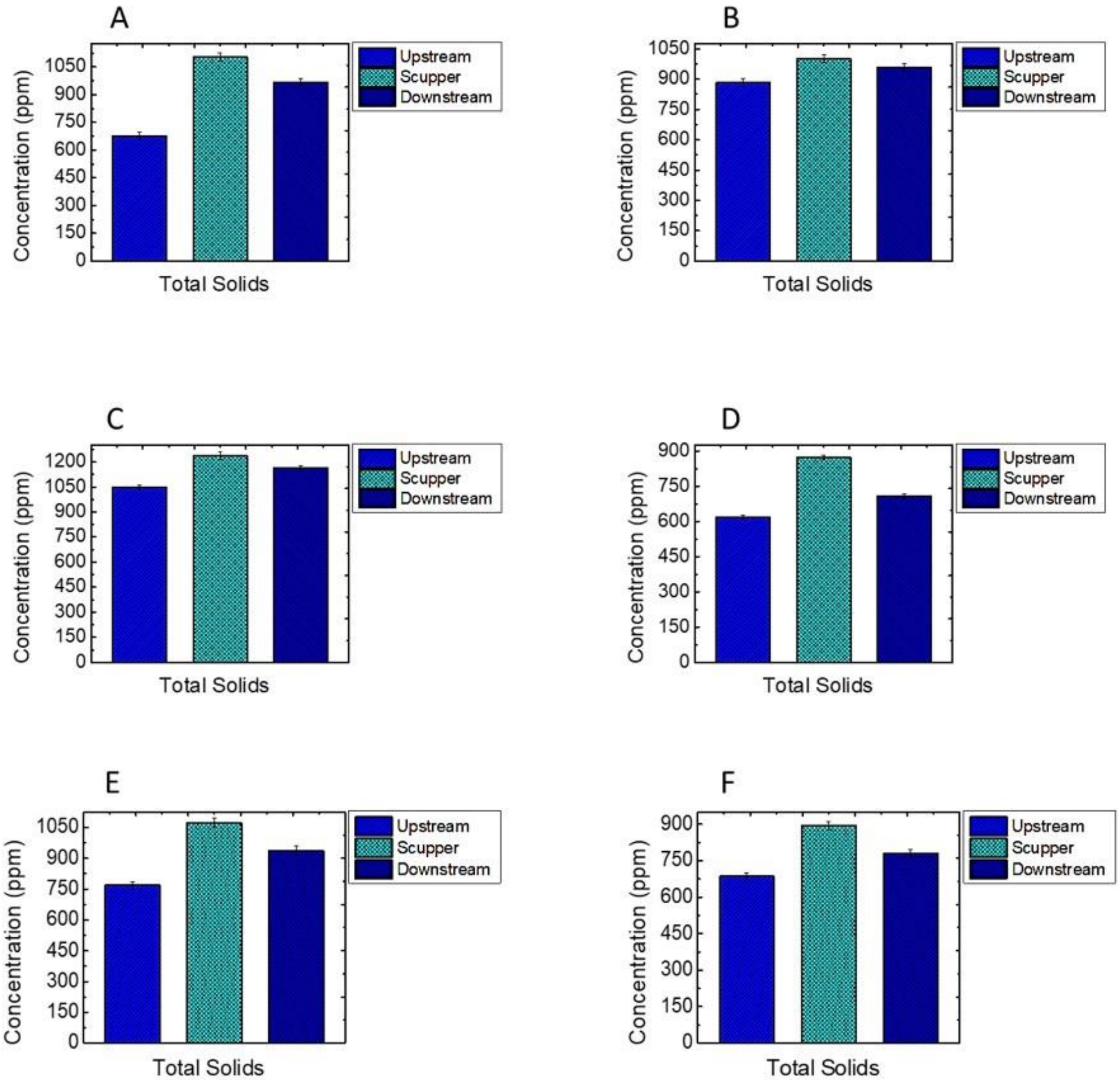

Figure 38. Bar graphs. Total solids concentrations from upstream, scupper drains, and downstream for: (A) Site SR 24, (B) Site RF, (C) Site US 80, (D) Site SR 297, (E) Site US 255, and (F) Site SR 197. Sample dates were 01/31/2020 for Sites SR 24 and SR 297, 10/22/2019 for Sites RF and US 80, and 01/24/2019 for Sites US 255 and SR 197.

Figure 39 shows total solids concentrations in scupper drain runoff for all four Southeast Georgia sites and the two North Georgia sites from consecutive dates. From figure 39, total solids concentrations vary between a decline and an incline, as in figure 39-A (Site SR 24).

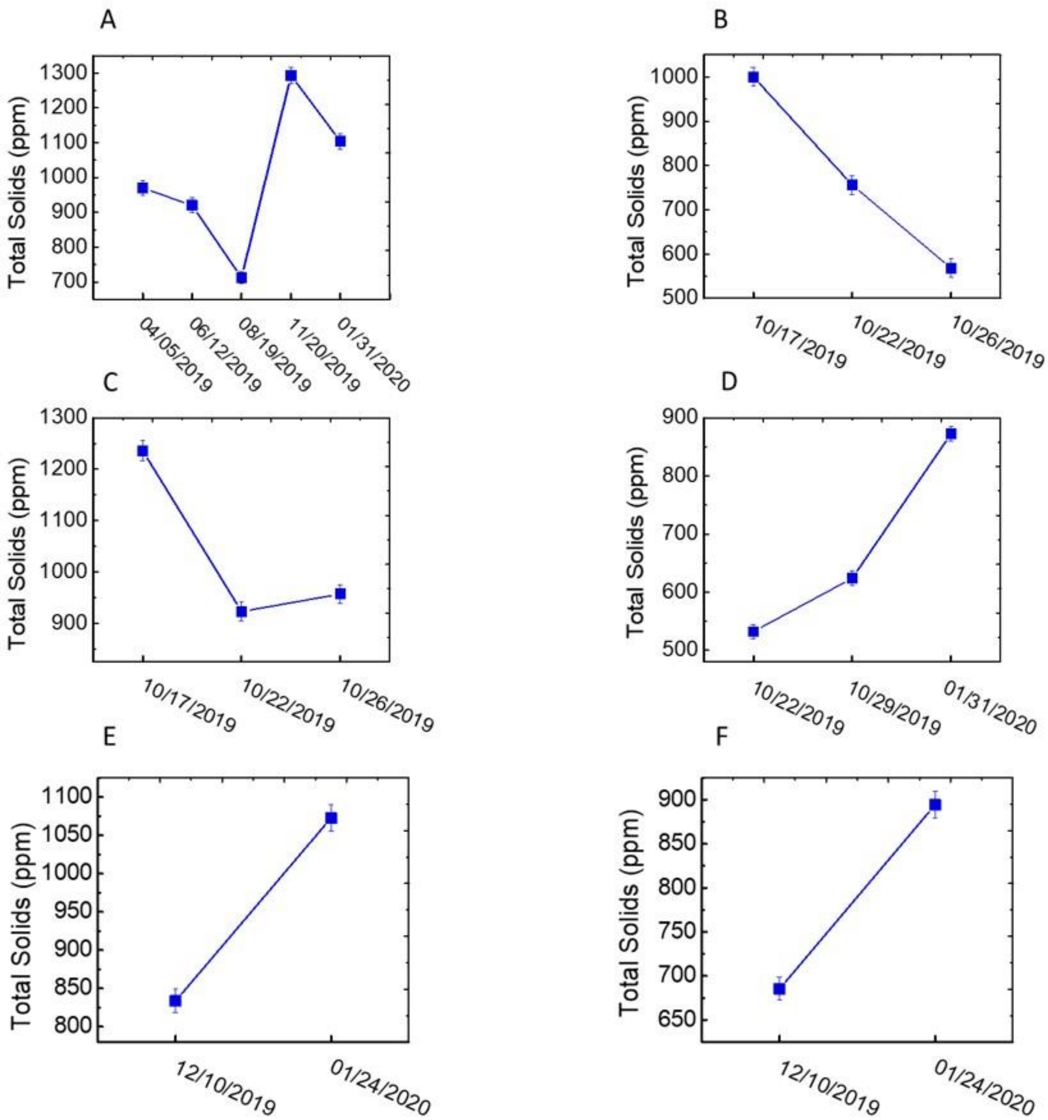

Figure 39. Line graphs. Total solids concentrations from scupper drains and for different dates for: (A) Site SR 24, (B) Site RF, (C) Site US 80, (D) Site SR 297, (E) Site US 255, and (F) Site SR 197.

The lowest solids concentration on 08/19/2019 and the highest on 11/20/2019 for figure 39-A can be explained with respect to rainfall intensity and volume from figure 25-A, which shows lower rainfall intensity and lower rain volume during the month of August 2019 as compared to November 2019. For figure 39-B (Site RF) and figure 39-C (Site US 80), it was clear that rain intensity and rain volume were significantly higher in mid-October as compared to the end of that month, thus increasing solids loading in scupper drain runoff. For figure 39-D (Site SR 297), total solids concentration increases from 10/22/2019 to 01/31/2020; however, it could be seen from figure 26 that rainfall intensity and volume were much higher during the end of October 2019 as compared to the end of January 2020, which contradicts the fact that high-intensity storms and high rainfall volume can transport more volume of solids as compared to low-intensity storms and high rainfall volume. As mentioned earlier, dry periods can increase solids loading on bridge deck surface that accumulates from dust fall, pavement wear, and other sources. Therefore, the higher solids concentration for figure 39-D can be explained with respect to the frequent dry periods and the low cleansing effect of rain preceding late January 2020. Therefore, the combination of both effects of rainfall and dry periods can have a clear impact on increased solids loading.

Figure 39-E and -F show total solids concentrations from scupper drains for the two North Georgia sites from consecutive dates. From figure 20, it is clear that rainfall volume was much lower on and before 12/10/2019 as compared to 01/24/2020, which explains the much lower solids concentration in scupper drain runoff samples of Site US 255 from 12/10/2019, as compared to the higher concentration on 01/24/2020 (figure 39-E). Moreover, it is noticeable that a dry period preceded both 12/10/2019 and 01/24/2020 (figure 20), which indicates that solids accumulation could be taking place prior to rainfall events; however, the higher rain volume on 01/24/2020 led to more solids transportation to scupper drain runoff. As mentioned previously, no precipitation data were available for Site SR 197. Thus, total solids results cannot be evidenced according to precipitation values for Site SR 197 (though it is in the same climatic region as Site US 255, 7.3 miles apart).

Figure 40 shows total solids concentrations at time intervals for specific dates for all four Southeast Georgia sites and the two North Georgia sites.

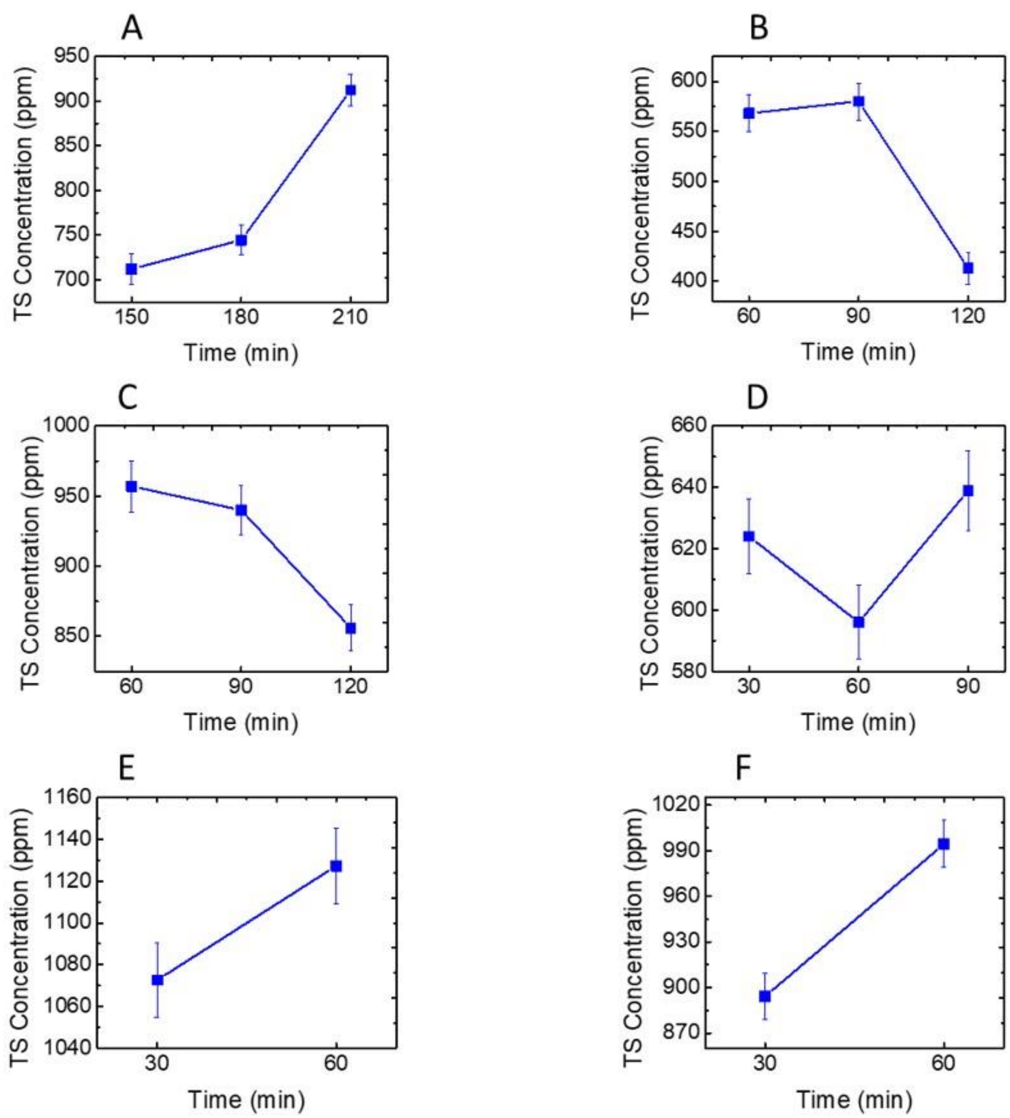

Figure 40. Line graphs. Total solids concentrations from scupper drains and at time intervals for: (A) Site SR 24 on 08/19/2019, (B) Site RF on 10/26/2019, (C) Site US 80 on 10/26/2019, (D) Site SR 297 on 10/29/2019, (E) Site US 255 on 01/24/2020, and (F) Site SR 197 on 01/24/2020.

Figure 40 shows both patterns of increasing and decreasing solids concentrations with time for the Southeast Georgia sites. For instance, figure 40-A (Site SR 24) shows a consistent increase in solids concentration with time, figure 40-B (Site RF) and figure 40-C (Site US 80) show a consistent decrease in solids concentration with time, and figure 40-D (Site SR 297) shows a pattern of decrease then increase in solids concentration with time. These variations in solids concentrations with time can be attributed to most likely varying rainfall intensity during each rainfall event for each site. However, no record of precipitation versus time data was found from any USGS station for all six study sites and for each specified date to support the statement. When rainfall intensity increases, solids mobilization from the bridge deck and its flushing into scupper drains increases, and then decreases when rainfall intensity decreases.[1,10]

Figure 40-E and -F show total solids concentrations at time intervals for the two North Georgia sites. Solids concentration increased with time for both sites, indicating that rainfall intensity increased within the selected sampling time range.

Total Suspended Solids (TSS)

TSS are the insoluble portion of total solids and, in general, are predominantly inorganic in nature. Table 16 shows TSS concentrations for all six Georgia study sites.

Table 16. Total suspended solids concentrations for all six Georgia study sites from instream and scupper drain runoff and at different dates and time intervals.

Total Suspended Solids (ppm)			
Date	**Upstream**	**Scupper**	**Downstream**
Site SR 24			
04-05-19	593.50±12.00	794.00±14.00	640.00±13.00
06-12-19	Not sampled	722.50±13.00	Not sampled
08-19-19 (150 min)	Not sampled	507.00±11.00	Not sampled
08-19-20 (180 min)	Not sampled	532.00±12.00	Not sampled
08-19-21 (210 min)	Not sampled	690.00±14.00	Not sampled
11-20-19	882.00±16.00	1027.00±17.00	1008.00±18.00
01-31-20	547.46±11.00	786.21±15.00	714.00±11.00
Average	**674.00±13.00**	**722.67±13.70**	**787.00±14.00**
Site Rocky Ford (RF)			

Table 16. Cont'd.

Total Suspended Solids (ppm)			
Date	**Upstream**	**Scupper**	**Downstream**
Site Rocky Ford (RF)			
10-17-19	723.00±16.00	842.30±16.00	793.00±15.00
10-22-19	525.00±11.00	578.00±10.00	541.00±12.00
10-26-19 (60 min)	Not sampled	422.00±9.00	Not sampled
10-26-19 (90 min)	Not sampled	431.00±10.00	Not sampled
10-26-19 (120 min)	Not sampled	252.00±5.00	Not sampled
Average	**624.00±14.00**	**505.00±10.00**	**667.00±14.00**
Site US 80			
10-17-19	728.00±14.00	876.00±15.00	820.00±15.00
10-22-19	655.00±13.00	687.00±10.00	654.00±12.00
10-26-19 (60 min)	Not sampled	721±13.00	Not sampled
10-26-19 (90 min)	Not sampled	723±12.00	Not sampled
10-26-19 (120 min)	Not sampled	564±9.00	Not sampled
Average	**691.5±13.00**	**714.00±12.00**	**737.00±13.00**
Site SR 297			
10-22-19	479.00±11.00	389.00±10.00	456.00±10.00
10-29-19 (30 min)	556.00±12.00	413.00±12.00	550.00±10.00
10-29-19 (60 min)	Not sampled	369.00±11.00	Not sampled
10-29-19 (90 min)	Not sampled	420.00±10.00	Not sampled
01-31-20	447.00±0.00	586.79±13.00	493.20±11.00
Average	**494.00±11.50**	**435.50±11.00**	**500.00±10.30**
Site US 255			
12-10-19	502.14±11.00	644.77±12.00	574.60±14.00
01-24-20 (30 min)	607.82±13.00	874.36±15.00	763.73±15.00
01-24-20 (60 min)	Not sampled	925.74±16.00	Not sampled
Average	**555.00±12.00**	**815.00±14.00**	**669.00±15.00**
Site SR 197			
12-10-19	453.00±13.00	509.22±14.00	484.00±11.00
01-24-20 (30 min)	556.00±14.00	741.25±16.00	646.29±13.00
01-24-20 (60 min)	Not sampled	825.37±16.00	Not sampled
Average	**504.50±14.00**	**692.00±15.00**	**565.00±12.00**

Total Dissolved Solids (TDS)

TDS are the soluble portion of total solids and are formed of soluble minerals, salts, and dissolved heavy metals. Table shows TDS concentrations for all six Georgia study sites.

Table 17. Total dissolved solids concentrations for all six Georgia study sites from instream and scupper drain runoff and at different dates and time intervals.

Total Dissolved Solids (ppm)			
Date	**Upstream**	**Scupper**	**Downstream**
Site SR 24			
04-05-19	167.50±4.00	176.00±4.60	146.50±3.50
06-12-19	Not sampled	197.50±4.80	Not sampled
08-19-19 (150 min)	Not sampled	205.00±5.70	Not sampled
08-19-20 (180 min)	Not sampled	212.60±5.00	Not sampled
08-19-21 (210 min)	Not sampled	222.40±6.10	Not sampled
11-20-19	202.13±5.00	266.00±7.00	224.00±6.00
01-31-20	129.00±3.20	317.78±7.40	254.00±6.30
Average	**166.00±4.00**	**228.20±5.80**	**208.00±5.30**
Site Rocky Ford (RF)			
10-17-19	161.00±3.20	158.00±3.00	165.00±4.30
10-22-19	147.00±2.00	178.00±4.00	152.00±3.50
10-26-19 (60 min)	Not sampled	146.00±2.50	Not sampled
10-26-19 (90 min)	Not sampled	149.00±3.00	Not sampled
10-26-19 (120 min)	Not sampled	161.00±4.00	Not sampled
Average	**154.00±2.60**	**158.40±3.30**	**158.50±3.90**
Site US 80			
10-17-19	321.00±7.40	360.00±6.80	343.00±6.60
10-22-19	228.00±5.30	236.00±4.00	227.00±5.00
10-26-19 (60 min)	Not sampled	236.00±5.00	Not sampled
10-26-19 (90 min)	Not sampled	217.00±3.00	Not sampled
10-26-19 (120 min)	Not sampled	292.00±6.60	Not sampled
Average	**274.50±6.30**	**268.00±5.00**	**285.00±5.80**
Site SR 297			
10-22-19	171.00±3.00	143.00±4.00	129.00±2.00
10-29-19 (30 min)	134.00±2.50	211.00±3.40	129.00±1.50
10-29-19 (60 min)	Not sampled	227.00±4.10	Not sampled
10-29-19 (90 min)	Not sampled	21.009±4.00	Not sampled
01-31-20	172.25±3.60	286.00±4.90	215.00±4.00
Average	**159.00±3.00**	**217.20±4.00**	**157.70±2.50**
Site US 255			
12-10-19	130.49±2.30	189.00±3.40	162.34±3.00
01-24-20 (30 min)	160.36±3.00	198.22±4.00	174.20±3.30
01-24-20 (60 min)	Not sampled	201.54±4.00	Not sampled
Average	**145.40±2.60**	**196.20±3.80**	**168.20±3.00**
Site SR 197			
12-10-19	155.11±2.50	176.32±±4.00	178.69±5.00
01-24-20 (30 min)	129.43±2.00	152.94±4.00	134.18±4.00
01-24-20 (60 min)	Not sampled	169.13±4.50	Not sampled
Average	**142.00±2.20**	**166.00±4.10**	**156.40±4.50**

Summary of TS Results

TS was found to be introduced from scupper drains with observable impact on instream water quality. This shows that the bridge deck can increase instream loading of TS, and major sources of solids in bridge stormwater runoff include atmospheric fallout, dust fall, rehabilitation work, construction and maintenance activities, and pavement wear. Downstream TS concentrations were elevated by the introduction of scupper drain runoff for all sites. TS concentrations were investigated in scupper drain runoff on different dates, and it was found that TS concentrations varied randomly with the different dates when TS were investigated at sampling time intervals, which can be attributed to changes in rainfall intensity. When rainfall intensity increases, solids mobilization from the bridge deck and its flushing into scupper drains increases, and then decreases when rainfall intensity decreases.

OIL AND GREASE

The environmental significance of oil and grease to aquatic ecosystems is the capacity of such dense organic components to create nuisance conditions in water bodies, and their effect in blocking oxygen and sunlight. Moreover, oil and grease from lubricating oils contain toxic components, such as lead plus other heavy metals, and thus can increase heavy metals concentrations in scupper drain runoff. Sources of oil and grease in scupper drain runoff are numerous, such as motor lubricants, spills and leaks, antifreeze, and hydraulic fluids. Oil and grease are measured as collective hexane extractable components, and it is estimated that around 2,000 spills take place annually on highways in the U.S.[1,7] Table 1 shows oil and grease concentrations for all six Georgia study sites.

Since oil and grease are insoluble organic content, their concentrations in instream and scupper drain runoff were found to be variable, inconsistent, and mostly low. However, as can be observed from table 118, concentrations of oil and grease were still detected to be higher in scupper drain runoff as compared to instream samples for the four Southeast Georgia sites. Oil and grease were found to be completely undetected in scupper drain runoff and instream samples in the two North Georgia sites, which was clearly attributable to the observable low traffic volume at both North Georgia sites as compared to the Southeast Georgia sites.

Table 18. Oil and grease concentrations for all six Georgia study sites from instream and scupper drain runoff and at different dates and time intervals.

Oil & Grease (ppm)			
Date	**Upstream**	**Scupper**	**Downstream**
Site SR 24			
04-05-19	0.98±0.01	3.50±0.04	0.77±0.01
06-12-19	Not sampled	1.20±0.04	Not sampled
08-19-19 (150 min)	Not sampled	2.30±0.06	Not sampled
08-19-20 (180 min)	Not sampled	1.10±0.08	Not sampled
08-19-21 (210 min)	Not sampled	0.80±0.01	Not sampled
11-20-19	0.00±0.00	0.00±0.00	0.00±0.00
01-31-20	0.00±0.00	0.13±0.00	0.00±0.00
Average	**0.98±0.01**	**1.30±0.04**	**0.77±0.01**
Site Rocky Ford (RF)			
10-17-19	1.44±0.05	4.60±0.03	3.78±0.08
10-22-19	1.22±0.05	1.42±0.01	1.30±0.01
10-26-19 (60 min)	Not sampled	0.37±0.07	Not sampled
10-26-19 (90 min)	Not sampled	0.00±0.00	Not sampled
10-26-19 (120 min)	Not sampled	0.00±0.00	Not sampled
Average	**1.33±0.05**	**2.13±0.04**	**2.54±0.04**
Site US 80			
10-17-19	1.63±0.02	4.37±0.10	2.46±0.08
10-22-19	2.12±0.08	3.39±0.10	2.74±0.09
10-26-19 (60 min)	Not sampled	2.79±0.07	Not sampled
10-26-19 (90 min)	Not sampled	1.03±0.01	Not sampled
10-26-19 (120 min)	Not sampled	1.00±0.01	Not sampled
Average	**1.88±0.05**	**2.52±0.06**	**2.60±0.09**
Site SR 297			
10-22-19	1.76±0.01	2.18±0.05	2.05±0.06
10-29-19 (30 min)	0.00±0.00	0.00±0.00	0.00±0.00
10-29-19 (60 min)	Not sampled	0.00±0.00	Not sampled
10-29-19 (90 min)	Not sampled	0.00±0.00	Not sampled
01-31-20	0.00±0.00	0.09±0.00	0.00±0.00
Average	**1.76±0.01**	**0.45±0.03**	**2.05±0.06**
Site US 255			
12-10-19	0.00±0.00	0.00±0.00	0.00±0.00

Table 18. Cont'd

Oil & Grease (ppm)			
Date	**Upstream**	**Scupper**	**Downstream**
01-24-20 (30 min)	0.00±0.00	0.00±0.00	0.00±0.00
01-24-20 (60 min)	0.00±0.00	0.00±0.00	0.00±0.00
Average	0.00±0.00	0.00±0.00	0.00±0.00
Site SR 197			
12-10-19	0.00±0.00	0.00±0.00	0.00±0.00
01-24-20 (30 min)	0.00±0.00	0.00±0.00	0.00±0.00
01-24-20 (60 min)	0.00±0.00	0.00±0.00	0.00±0.00
Average	0.00±0.00	0.00±0.00	0.00±0.00

PH

Acidic and alkaline conditions can be directly measured through pH, which is a measure of protons' mobility and their concentration in water.[1,38] Natural pH of fresh water can range from neutral to slightly alkaline, and the thriving pH for aerobic bacteria—which is critically important to maintain healthy conditions for aquatic biota through digestion and removal of oxidizable loads—is 6.00 to 9.00[1, 38] If the pH is unbalanced in water bodies, this can lead to the death of aerobic bacteria and, thus, the creation of nuisance conditions that threaten aquatic creatures. Moreover, the pH imbalance makes fresh water unsuitable for human consumption. Table shows pH values for all six Georgia study sites.

From table 19, pH values were found to be within the acceptable range, varying between 5.20 and 7.80 for all six study sites. The slightly acidic pH of 5.20 could be due to the presence of more carbon dioxide in the water with less evaporation during the low-temperature months of October to January, leading to higher formation of carbonic acids as compared to summer months. However, most pH values are found to be within the reasonable range of approximately 6.00 to 8.00.

Table 19. pH values for all six Georgia study sites from instream and scupper drain runoff and at different dates and time intervals.

pH			
Date	**Upstream**	**Scupper**	**Downstream**
Site SR 24			
04-05-19	7.55±0.20	7.75±0.30	7.65±0.20
06-12-19	Not sampled	7.15±0.10	Not sampled
08-19-19 (150 min)	Not sampled	7.82±0.20	Not sampled
08-19-20 (180 min)	Not sampled	7.42±0.10	Not sampled
08-19-21 (210 min)	Not sampled	7.18±0.10	Not sampled
11-20-19	6.63±0.20	5.89±0.10	6.14±0.20
01-31-20	7.47±0.30	5.28±0.10	5.96±0.10
Average	**7.22±0.23**	**6.90±0.14**	**6.60±0.17**
Site Rocky Ford (RF)			
10-17-19	6.97±0.20	6.36±0.10	6.94±0.30
10-22-19	7.12±0.20	6.32±0.10	6.50±0.30
10-26-19 (60 min)	Not sampled	6.59±0.20	Not sampled
10-26-19 (90 min)	Not sampled	6.66±0.20	Not sampled
10-26-19 (120 min)	Not sampled	6.81±0.20	Not sampled
Average	**7.00±0.20**	**6.55±0.16**	**6.72±0.30**
Site US 80			
10-17-19	7.21±0.20	6.88±0.20	6.89±0.20
10-22-19	6.76±0.20	5.22±0.10	6.09±0.10
10-26-19 (60 min)	Not sampled	6.24±0.10	Not sampled
10-26-19 (90 min)	Not sampled	6.38±0.20	Not sampled
10-26-19 (120 min)	Not sampled	7.02±0.20	Not sampled
Average	**7.00±0.2**	**6.35±0.16**	**6.50±0.15**
Site SR 297			
10-22-19	6.26±0.10	6.84±0.20	6.63±0.20
10-29-19 (30 min)	7.24±0.20	7.01±0.20	7.20±0.20
10-29-19 (60 min)	Not sampled	6.87±0.20	Not sampled
10-29-19 (90 min)	Not sampled	7.00±0.20	Not sampled
01-31-20	6.93±0.10	6.15±0.10	6.88±0.20
Average	**6.80±0.15**	**6.77±0.18**	**6.90±0.20**
Site US 255			
12-10-19	7.20±0.20	5.21±0.10	5.90±0.10
01-24-20 (30 min)	6.54±0.10	6.14±0.10	6.24±0.20
01-24-20 (60 min)	Not sampled	6.74±0.10	Not sampled
Average	**6.90±0.15**	**6.00±0.10**	**6.00±0.15**
Site SR 197			
12-10-19	6.89±0.20	5.44±0.10	6.24±0.20
01-24-20 (30 min)	6.49±0.20	6.08±0.10	6.29±0.20
01-24-20 (60 min)	Not sampled	6.43±0.20	Not sampled
Average	**6.70±0.20**	**6.00±0.13**	**6.30±0.20**

DISSOLVED OXYGEN (DO)

DO is a measure of the amount of oxygen present in water. DO is a very important water quality parameter because oxygen is needed by aquatic creatures and aerobic bacteria to survive. The adequate DO concentration in fresh water is 9.00 mg/L; however, DO values can vary by an average of ±1.00 mg/L without harming aquatic systems.[38] DO values of fresh water significantly lower than the optimum values can create very harmful conditions in fresh water bodies, leading to suffocation and death of aquatic creatures, as well as the death of aerobic bacteria and the predominance of nuisance conditions in water bodies, which makes them unfit for human use or consumption. Low DO values also reflect an increase in COD of water bodies due to severe contamination. Table 20 shows DO concentrations for all six Georgia study sites.

Table shows that DO values range from approximately 6.00 to 10.00 mg/L. The fact that many of the low DO values are during the months of winter indicates that COD elevates during these months, leading to decline or imbalance in DO values, which is a matter of concern. Sensitive aquatic habitat can suffer the loss of DO during these times, and, thus, it is important to find means of decreasing the oxygen demand created from the introduction of several contaminants to instream water. In table , DO values are lower in scupper drain runoff due to higher COD and probably higher temperature of runoff, which definitely imbalances the natural DO of fresh water with the continuous introduction of scupper drain runoff into the instream water.

Table 20. Dissolved oxygen concentrations for all six Georgia study sites from instream and scupper drain runoff and at different dates and time intervals.

Dissolved Oxygen (ppm)			
Date	**Upstream**	**Scupper**	**Downstream**
Site SR 24			
04-05-19	10.53±0.20	7.98±0.10	11.34±0.10
06-12-19	Not sampled	9.85±0.10	Not sampled
08-19-19 (150 min)	Not sampled	8.10±0.10	Not sampled
08-19-20 (180 min)	Not sampled	7.90±0.10	Not sampled
08-19-21 (210 min)	Not sampled	7.54±0.10	Not sampled
11-20-19	8.11±0.10	6.39±0.10	7.55±0.10
01-31-20	9.78±0.20	6.27±0.10	7.56±0.10
Average	**9.50±0.17**	**7.70±0.10**	**8.80±0.10**
Site Rocky Ford (RF)			
10-17-19	7.21±0.10	6.11±0.10	6.69±0.10
10-22-19	8.17±0.10	7.87±0.10	7.93±0.10
10-26-19 (60 min)	Not sampled	7.83±0.10	Not sampled
10-26-19 (90 min)	Not sampled	8.11±0.10	Not sampled
10-26-19 (120 min)	Not sampled	7.35±0.10	Not sampled
Average	**7.70±0.10**	**7.45±0.10**	**7.30±0.10**
Site US 80			
10-17-19	7.09±0.10	5.94±0.10	6.33±0.20
10-22-19	8.44±0.10	9.33±0.10	9.71±0.10
10-26-19 (60 min)	Not sampled	6.41±0.10	Not sampled
10-26-19 (90 min)	Not sampled	6.13±0.10	Not sampled
10-26-19 (120 min)	Not sampled	5.97±0.20	Not sampled
Average	**7.80±0.10**	**6.80±0.12**	**8.00±0.15**
Site SR 297			
10-22-19	6.02±0.20	8.13±0.10	6.00±0.10
10-29-19 (30 min)	8.33±0.10	8.76±0.10	8.34±0.20
10-29-19 (60 min)	Not sampled	7.69±0.10	Not sampled
10-29-19 (90 min)	Not sampled	8.19±0.10	Not sampled
01-31-20	10.24±0.20	8.52±0.20	8.96±0.10
Average	**8.20±0.15**	**8.30±0.12**	**7.80±0.15**
Site US 255			
12-10-19	9.06±0.10	7.31±0.20	7.82±0.20
01-24-20 (30 min)	10.17±0.10	8.21±0.10	9.34±0.10
01-24-20 (60 min)	Not sampled	9.24±0.10	Not sampled
Average	**9.60±0.10**	**8.25±0.13**	**8.60±0.15**
Site SR 197			
12-10-19	8.99±0.10	6.60±0.10	7.57±0.20
01-24-20 (30 min)	8.93±0.10	7.55±0.20	7.91±0.10
01-24-20 (60 min)	Not sampled	8.11±0.20	Not sampled
Average	**8.90±0.10**	**7.42±0.17**	**7.74±0.15**

CONDUCTIVITY

Conductivity measurement can give an indirect indication of the amount of salts, minerals, dissolved solids, and other dissolved constituents, such as dissolved heavy metals of concern. Thus, high conductivity values indicate that water can contain elevated amounts of toxic heavy metals. Acceptable conductivity ranges up to 150.00 to 500.00 µS/cm.[87] Table shows conductivity values for all six Georgia study sites. From table 21, conductivity values were found to be within acceptable levels, which indicates that soluble salts, solids, and heavy metals are within an acceptable range for fresh water. Due to instream discharge, soluble components can dissipate with water currents. However, and as can be seen in table 21, scupper drain runoff shows mostly higher conductivity values compared to upstream, and that introduction of runoff elevates the downstream conductivity. This raises the concern that with extended periods of time, the introduction of dissolved content can threaten the instream aquatic ecosystem.

Table 21. Conductivity values for all six Georgia study sites from instream and scupper drain runoff and at different dates and time intervals.

Conductivity (µS/cm)			
Date	**Upstream**	**Scupper**	**Downstream**
Site SR 24			
04-05-19	261.00±2.00	280.00±2.00	277.00±1.00
06-12-19	Not sampled	315.00±2.00	Not sampled
08-19-19 (150 min)	Not sampled	323.00±2.00	Not sampled
08-19-20 (180 min)	Not sampled	329.00±2.00	Not sampled
08-19-21 (210 min)	Not sampled	353.00±2.00	Not sampled
11-20-19	244.00±2.00	429.00±2.00	367.00±2.0
01-31-20	169.00±2.00	263.91±2.00	215.15±2.0
Average	**224.70±2.00**	**327.60±2.00**	**386.40±1.7**
Site Rocky Ford (RF)			
10-17-19	152.00±1.00	273.00±2.00	194.00±1.00
10-22-19	221.00±2.00	305.00±2.00	229.00±1.00
10-26-19 (60 min)	Not sampled	258.00±1.00	Not sampled
10-26-19 (90 min)	Not sampled	229.00±1.00	Not sampled
10-26-19 (120 min)	Not sampled	301.00±1.00	Not sampled
Average	**186.50±1.50**	**265.30±1.40**	**211.50±1.00**

Table 21. Cont'd

Conductivity (μS/cm)			
Date	**Upstream**	**Scupper**	**Downstream**
Site US 80			
10-17-19	136.00±1.00	253.00±1.00	144.00±1.00
10-22-19	232.00±1.00	374.00±2.00	265.00±2.00
10-26-19 (60 min)	Not sampled	469.00±2.00	Not sampled
10-26-19 (90 min)	Not sampled	418.00±2.00	Not sampled
10-26-19 (120 min)	Not sampled	415.00±2.00	Not sampled
Average	**184.00±1.00**	**385.80±1.80**	**204.50±1.50**
Site SR 297			
10-22-19	182.00±1.00	151.00±1.00	170.00±1.00
10-29-19 (30 min)	283.00±2.00	274.00±2.00	280.00±1.00
10-29-19 (60 min)	Not sampled	159.00±1.00	Not sampled
10-29-19 (90 min)	Not sampled	161.00±1.00	Not sampled
01-31-2020	93.87±1.00	137.00±1.00	110.18±1.00
Average	**189.30±1.50**	**176.40±1.20**	**187.00±1.00**
Site US 255			
12-10-19	187.00±1.00	354.00±2.00	321.00±2.00
01-24-20 (30 min)	227.00±1.00	342.00±2.00	289.00±1.00
01-24-20 (60 min)	Not sampled	37.001±2.00	Not sampled
Average	**207.00±1.00**	**355.70±2.00**	**305.00±1.50**
Site SR 197			
12-10-19	98.74±1.00	255.00±1.00	175.00±1.00
01-24-20 (30 min)	110.00±1.00	210.00±1.00	168.00±1.00
01-24-20 (60 min)	Not sampled	258.00±1.00	Not sampled
Average	**104.40±1.00**	**241.00±1.00**	**171.50±1.00**

STOCHASTIC EMPIRICAL LOADING AND DILUTION MODEL (SELDM)

SELDM is a lumped parameters empirical model that relies on various input data (e.g., site location, precipitation characteristics, hydraulics, and geographic and ecoregion information) to calculate critical parameters, such as downstream event mean concentrations in rivers and streams, which makes it different than traditional theoretical equations models.[1,76] SELDM can model runoff quality from different land topographies and land use characteristics employing defined site parameters when impervious-fraction data and water quality characteristics data are available (the

USGS has records of water quality data for different states). SELDM's output of numerical values for water quality is represented versus time or storm events, and the model identifies each storm using sequence numbers and annual load for a specific year. SELDM can integrate several events into one annual-load set if the time between event midpoints of a series of simulated events exceeds a year's time.[1]

SELDM was employed as a comparative tool to evaluate actual lab data in comparison to SELDM simulated data. Since SELDM generates storm events randomly, there is no serial correlation, and the order of storms does not reflect seasonal patterns.[1] Therefore, the model is employed here not to provide information for water quality in accordance with different seasons, but rather to provide an insight on model outcomes versus actual lab outcomes. The software is designed such that to obtain downstream information, water quality parameters need to be defined in highway (i.e., bridge site) and upstream water quality menus based on the USGS database. Since such information for the State of Georgia is not abundant, only defined downstream pairs were modeled (i.e., downstream component paired with bridge runoff component) that were available within the software for the six Georgia study sites of this research project, which are pH, total phosphorus, total hardness, and suspended sediment. Common parameters were relied on for pH and total phosphorus for comparison to our lab data, and SELDM outcomes were provided for all four parameters to help shed light on the potential impacts of these parameters in the long-term.

Figure 41 shows SELDM downstream pH values versus storm number and year number for Site SR 24 in the Southeast Georgia sites. The data are represented versus virtual storm events, which in total are 1,657 events, as well as over 30 years of simulated prolonged time, aiming to provide hypothetical insight on varying downstream impacts of pH. Surprisingly, SELDM predicted the pH for all four Southeast Georgia sites to be similar, and SELDM-generated pH values were well preserved between 5.80 and 8.20 for all four Southeast Georgia sites (see figure 53, figure 54, figure 55, and figure 56 in the appendix). This could be interpreted relative to two factors: (1) the pH for surface waters in the Southeast ecoregion is closely similar, ranging between slightly acidic and slightly alkaline with a perfectly average neutrality of 7.00; and (2) pH changes in surface waters are trivial.

Such outcome is in agreement with our measured pH values (table) that show insignificant pH variabilities for all four Southeast Georgia sites, varying between 5.20 and 7.80. Thus, both lab outcomes and SELDM modeling were in close agreement on pH changes, which was not indicative of potential impacts due to instream contamination. This also indicated that only sharp changes in acidity or alkalinity would reflect in pronounced pH changes, either instantly or over long durations of time. Since most contaminants considered in this study were found to be of relatively low concentrations, and also considering the instant instream dilution effect of stream discharge, detection of sharp pH changes in instream water from any acidic or alkaline contaminants was unlikely. The same outcomes were observed for the two North Georgia sites, Sites US 255 and SR 197, with pH ranging between 5.20 and 7.50.

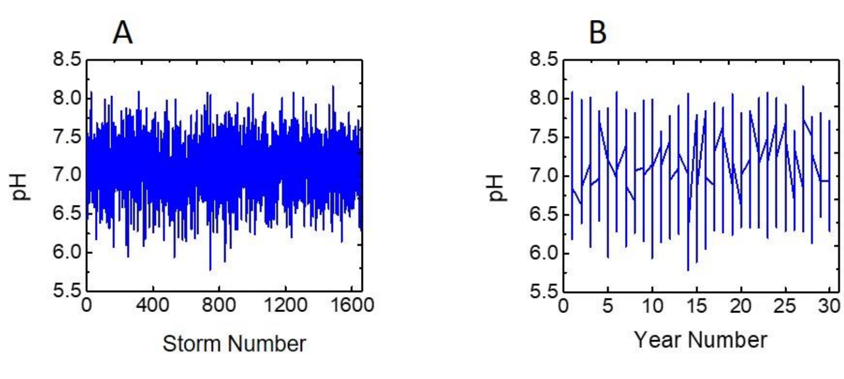

Figure 41. Graphs. SELDM downstream pH versus storm number and year number for: (A) Site SR 24 versus storm number, and (B) Site SR 24 versus year number.

Figure 42 shows minimum, average, and maximum total phosphorus concentrations in ppm (mg/L) developed from SELDM simulations for all six study sites in Southeast and North Georgia. Only minimum and maximum values with their averages are provided from the SELDM outputs. In general, SELDM TP concentrations were in the range of 0.01 ppm to 9.00 ppm for all six study sites. Phosphorus was detected experimentally only in the four Southeast Georgia sites, and our lab results showed that TP was in the range of 0.00 ppm to approximately a maximum of 5.30 ppm (measured for Site SR 297), while the highest SELDM TP of 9.00 ppm was for Site SR 197 and the second highest of 7.00 ppm was for Site US 255 (figure 42).

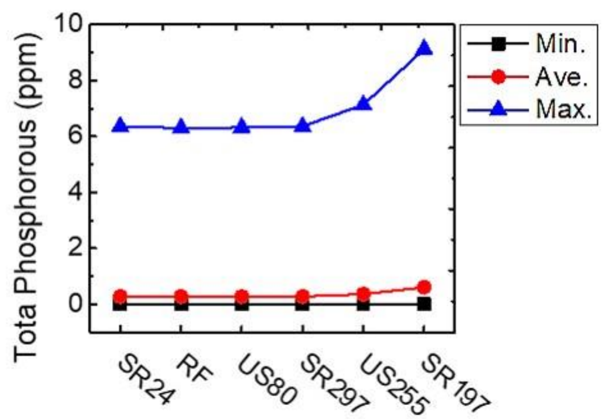

Figure 42. Line graph. Total phosphorus concentrations represented as minimum, average, and maximum for all six Georgia study sites, obtained from the SELDM simulation outputs.

However, the lab measurements for both North Georgia sites, US 255 and SR 197, showed the exact opposite outcomes to lab data of undetected levels of TP for both North Georgia sites. Since major sources of phosphorus are the atmosphere and fertilizers, it is hard to relate other minor factors, such as traffic count, to such outcomes for phosphorus. This unpredictable outcome of SELDM for phosphorus shows how the modeling software relies on major factors rather than instant conditions, as with field measurements. SELDM predicted the highest phosphorus concentration (9.00 ppm) for SR 197 to be existent in 13 years, which can also indicate that phosphorus concentration may spike with time for this location. On the other hand, the highest concentration for Site US 255 was predicted to be present in 1 year, which can indicate that this watershed has a potential of spiking phosphorus concentrations due to several factors, such as atmospheric fallout or soil-related factors. However, since SELDM generates storm events randomly with no serial correlation, the predictions are hypothetical. SELDM predicted the TP concentrations to be almost similar for the four Southeast Georgia sites, ranging from 0.00 to 6.36 ppm. The lab results in this study for the Southeast region showed phosphorus to range from approximately 1.00 to 5.30 ppm.

Although SELDM predicted lower minimum values, the maximum values are in proximity. The lab results showed that most TP concentrations are of similar values for downstream with an average of 2.40 ppm, although it is unclear why SELDM predicted very close TP maximums for all four Southeast Georgia sites (6.30–6.36 ppm). This outcome could most likely indicate that phosphorus concentrations for the watershed are stable and that the potential for spiking TP concentrations over the years is unlikely. The low average concentration values for SELDM TP for all six study sites (0.28–0.61 ppm) infers that 95 percent of TP concentration values for all six study sites are below the theoretical mean for each site, and that the concentration spikes above the calculated average are for only 5 percent of storm/year numbers. This outcome shows that, in most cases, the chance for the presence of high downstream phosphorus concentrations is low and that the impact is accumulative.

Figure 43 shows minimum, average, and maximum suspended solids (SS) concentrations in ppm (mg/L) developed from SELDM simulations for all six study sites. Since SS carries an array of contaminants, including phosphorus, it is not unexpected to find a similar trend as with the case of TP in spiking maximum SS concentrations for the two North Georgia sites of 295.00 ppm for SR 197 and 107.00 ppm for US 255. Moreover, the SS is following a synonymous stable concentrations trend for the four Southeast Georgia sites, ranging from 68.10 to 70.00 ppm.

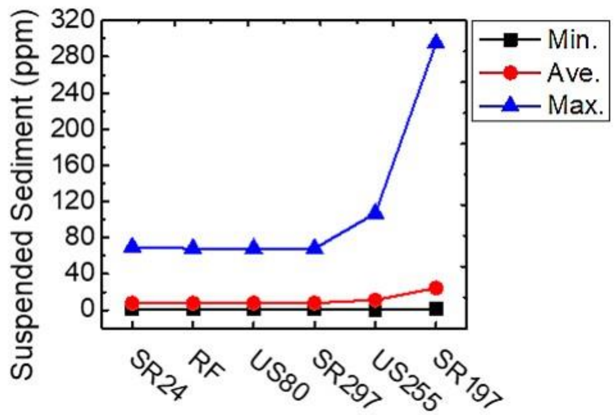

Figure 43. Line graph. Suspended solids concentrations represented as minimum, average, and maximum for all six study sites, obtained from the SELDM simulation outputs.

As SELDM is a lumped parameters model that relies on a wide range of factors from the watershed, including imperviousness and other highway characteristics, its calculations potentially take into account soil conditions and other surrounding factors. The software considers the bridge site as a highway site, although the exact location coordinates and drainage information for each bridge site are provided. Thus, it can be contingent that the unpredictability of its outcomes can be accepted from such a standpoint. SELDM predicted much lower SS concentrations as compared to lab-measured TSS for all six study sites. The maximum predicted value was 295.00 ppm for Site SR 197, compared to the highest lab-measured value of 646.00 ppm for the same site; our lab results showed the overall highest downstream TSS to be 1,008.00 ppm for Site SR 24, as compared to SELDM's highest prediction of 70.00 ppm for the same site. The average SS concentration values for SELDM for all six study sites (7.80–24.50 ppm) infer that 90 percent of SS concentration values for all six study sites are below the theoretical mean for each site, and that the concentration spikes above the calculated average are for only 10 percent of storm/year numbers. As with phosphorus, this outcome showed that, in most cases, the chance for the presence of high downstream SS concentrations is low and that the impact is accumulative.

Figure 44 shows minimum, average, and maximum total hardness (TH) concentrations in ppm (mg/L as $CaCO_3$) developed from SELDM simulations for all six study sites in Southeast and North Georgia.

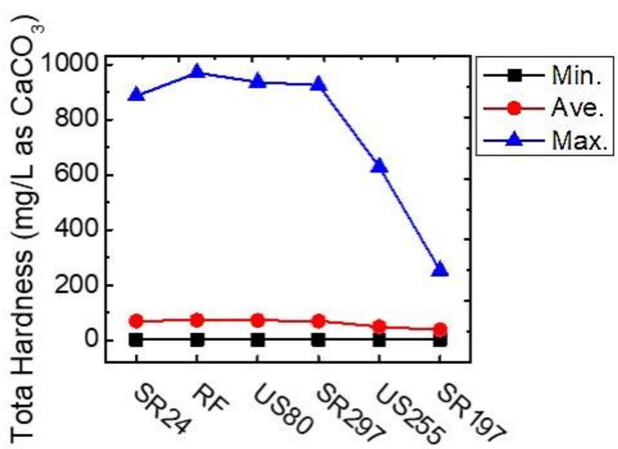

Figure 44. Line graph. Total hardness concentrations represented as minimum, average, and maximum for all six Georgia study sites, obtained from the SELDM simulation outputs.

From figure 44, it is observed that SELDM TH concentrations among the six study sites show opposite trends to TP and SS in figure 42 and figure 43. The highest TH value of 970.00 ppm for Site RF reflects increased hardness at some point along the extended SELDM forecast period of 30 years, which is projected to be in 28 years. As with the previous outcomes for TP and SS, the spiking concentration of TH is for a single storm event taking place in 28 years (figure 45).

This outcome is most likely based on the SELDM estimation given specific unique characteristics in the watershed and might consider specific surrounding factors for that ecoregion.

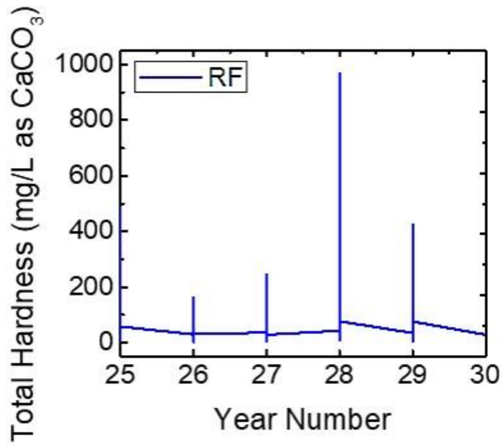

Figure 45. Graph. Maximum total hardness concentrations range (25–30 years projection) for Site RF in Southeast Georgia, obtained from the SELDM simulation outputs.

The lowest maximum TH value was found to be 252.00 ppm for Site SR 197, with a similar trend of a single storm event spike in concentration. The reason Site SR 197 was showing lower SELDM TH concentrations compared to other sites is unclear and not consistent with its opposite trends for TP and SS. However, this outcome clearly reflects that SELDM considers different variables that do not impact TP and SS in calculating TH concentrations, and that TP and SS do not impact stream hardness. Although sediment is responsible for the transmission of various contaminants into the instream, including heavy metals, SELDM is indicating that the likelihood of calcium and magnesium introduction to the instream with the sediment is low.

Likewise, the Site US 255 SELDM TH maximum concentration of 628.00 ppm followed an exact opposite trend to that of its TP and SS. The maximum TH concentrations for all four Southeast Georgia sites ranged in the proximity of 886.00 to 970.00 ppm, also following an opposite trend to their TP and SS trends in figure 42 and figure 43. The average concentrations values (figure 44) for SELDM TH for all six study sites (37.00–72.00 ppm) infer that 92 percent of TH concentration values for all six study sites were below the theoretical mean for each site, and that the concentration spikes above the calculated average were for only 8 percent of storm/year numbers. As with phosphorus and SS, this outcome showed that in most cases, the chance for the presence of high downstream total hardness concentrations was low and that the impact is accumulative.

CHAPTER 5. CONCLUSIONS

In this research project, six bridge sites in the Southeast and North regions of the State of Georgia were studied for the impact of bridge deck scupper drain stormwater runoff on sensitive streams. Four study sites in Southeast Georgia were selected: (1) Site SR 24 on the Ogeechee River, near Oliver, Georgia; (2) Site Rocky Ford on the Ogeechee River on Rocky Ford Road; (3) Site US 80 on the Ogeechee River near Eden, Georgia; and (4) Site SR 297 on the Ohoopee River near Swainsboro, Georgia. The remaining two study sites were located in North Georgia: (1) Site US 255 on the Chattahoochee River near Leaf, Georgia; and (2) Site SR 197 on the Soque River near Clarkesville, Georgia. The impact of different contaminants from bridge scupper drain stormwater runoff in the State of Georgia was not considered in the existing literature; thus, it was important to provide insight on the potential impacts of bridge scupper drain runoff on sensitive streams in Georgia.

The following water quality parameters were studied in this research project:

- Heavy metals (Pb, Zn, and Cu).
- PAHs.
- COD.
- Nitrogen (TN, TKN, NH_3-N, NO_3^--N, and NO_2^--N).
- Phosphorus (PO_4^{3-}-P).
- Solids (TS, TSS, and TDS).
- Oil and grease.
- pH.
- DO.

- Conductivity.

The influence of stream discharge and rainfall intensity were also observed, and it was found that instream discharge can play a significant role on changing downstream concentrations of water quality parameters. Rainfall intensity had a substantial effect on parameter concentrations in scupper drain runoff, and it was found that for most parameters except for solids, higher rainfall intensity and frequency leads to a dilution effect for parameter concentrations with time. For heavy metals, it was found that lead, zinc, and copper were of higher concentrations in scupper drain runoff and, thus, could impair instream water quality and elevate downstream concentrations in all six Georgia study sites, except that Pb was only detected in the four Southeast Georgia sites.

Among the 18 PAHs that were investigated, only seven were detected in scupper drain runoff samples of two of the Southeast Georgia study sites, Site SR 24 and Site US 80, while PAHs were undetected in their instream samples. The seven PAHs that were detected in scupper drain runoff samples from Site SR 24 and Site US 80 were benz[a]anthracene, benzo[a]pyrene, fluoranthene, pyrene, chrysene, indeno-[1,2,3-cd]-pyrene, and benzo[b]fluoranthene. Among the detected PAHs, the concentrations of benz[a]anthracene, benzo[a]pyrene, pyrene, and chrysene were found to exceed the drinking water guideline value of 10.000 ppt provided by the National Health and Medical Research Council and the Arkansas Water Resource Center. However, the excessive concentrations for these compounds were detected in scupper drain runoff of only Sites SR 24 and US 80, without observable impact to downstream water quality.

Nitrogen with all its components represented as TN was found to exist in all samples, with higher concentrations in scupper drain runoff samples compared to instream samples for most sites. Phosphorus as PO_4^{3-}-P was detected in all four Southeast Georgia sites with high concentrations in scupper drain runoff compared to instream samples and observable impact on downstream water quality, while it was undetectable in samples from both North Georgia sites. Solids with all its components represented as TS were found to be introduced from scupper drains into instream waters for all six study sites. However, solids were the only parameters that decreased considerably after the first flush in samples with time intervals. Oil and grease were found to be in low concentrations in all samples from the four Southeast Georgia study sites and undetectable in the North Georgia samples. However, oil and grease in the scupper drain runoff of the Southeast Georgia site samples were in higher concentrations compared to the instream samples and can have an impact on downstream water quality. The pH, DO, and conductivity were found to be in acceptable value ranges for all six study sites.

SELDM was employed in this research project as an effective modeling tool to project potential impacts of bridge deck stormwater runoff on instream water quality. Four downstream water quality parameters were defined in the software's input menu and were used to apply software simulations for the six Georgia study sites in this research project: pH, TP, TH, and SS. Common parameters were relied on, such as pH and total phosphorus, for comparison to our lab data, and SELDM outcomes were provided for all four parameters to help shed light on the potential impacts of these parameters in the long-term. SELDM provided similar values for pH for all six study sites, which indicated insignificant impact of pH and was in agreement with the lab-measured pH values. Surprisingly, SELDM identified the highest TP and SS concentrations to be for the two North Georgia study sites, which contradicted the lab findings of undetected TP in the North Georgia sites and not excessive TSS concentrations for the North Georgia sites compared to the four Southeast Georgia sites. This unpredictable outcome of SELDM projections for TP and SS shows how the modeling software relies on major factors rather than instant conditions, as with field measurements. As SELDM is a lumped parameters model that relies on a wide range of factors from the watershed, including imperviousness and other highway characteristics, it possibly takes soil conditions and other surrounding factors into account in its calculation. SELDM deals with the bridge site as a highway site although the exact location coordinates and drainage information for each bridge site are provided. Thus, it can be considered that the unpredictability of its outcomes can be accepted from such a standpoint. Further studies on more of the numerous bridge sites in the State of Georgia are recommended for further monitoring of impacts of bridge deck scupper drain runoff on sensitive streams and rivers are recommended, and more water quality data for the State is needed.

APPENDIX

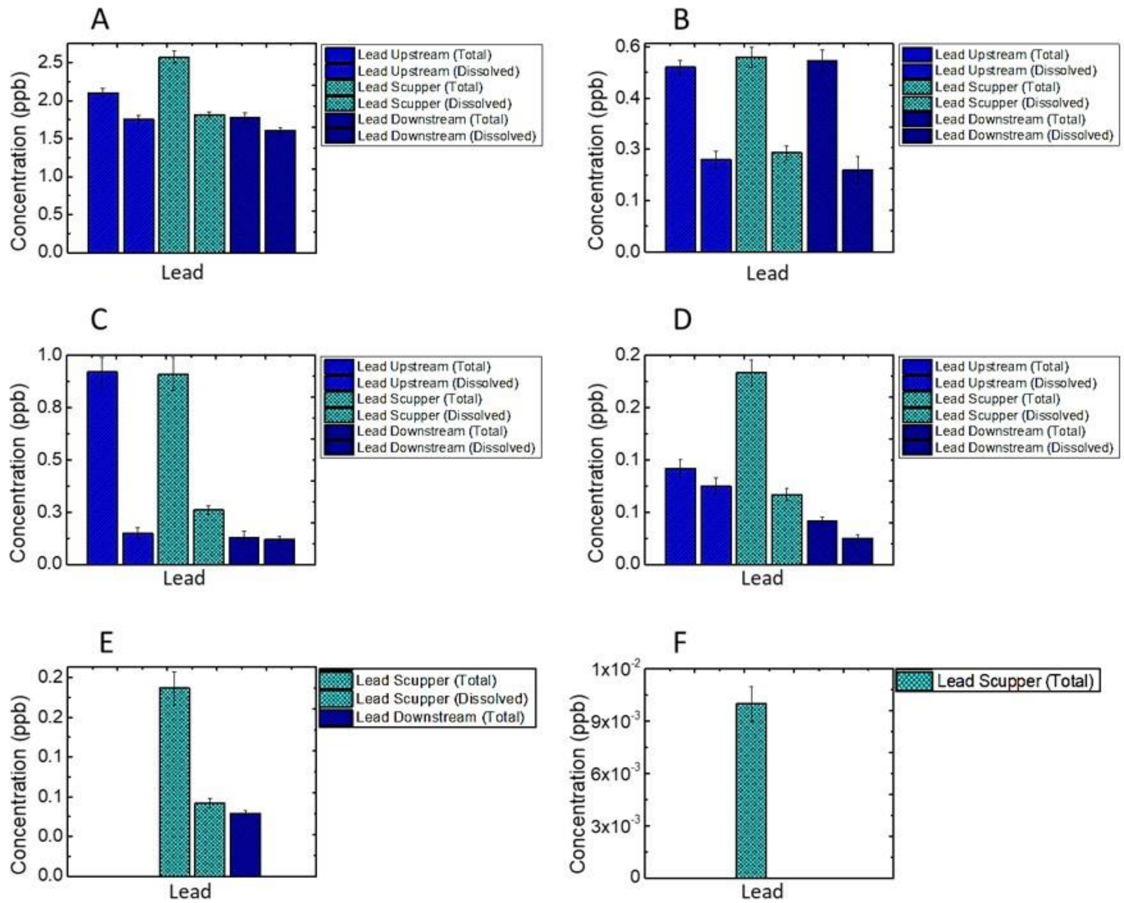

Figure 46. Bar graphs. Lead concentrations (total and dissolved) from upstream, scupper drains, and downstream for: (A) Site SR 24 on 04/05/2019, (B) Site SR 24 on 01/31/2020, (C) Site RF on 10/17/2019, (D) Site US 80 on 10/17/2019, (E) Site SR 297 on 01/31/2020, and (F) Site US 255 on 01/24/2020.

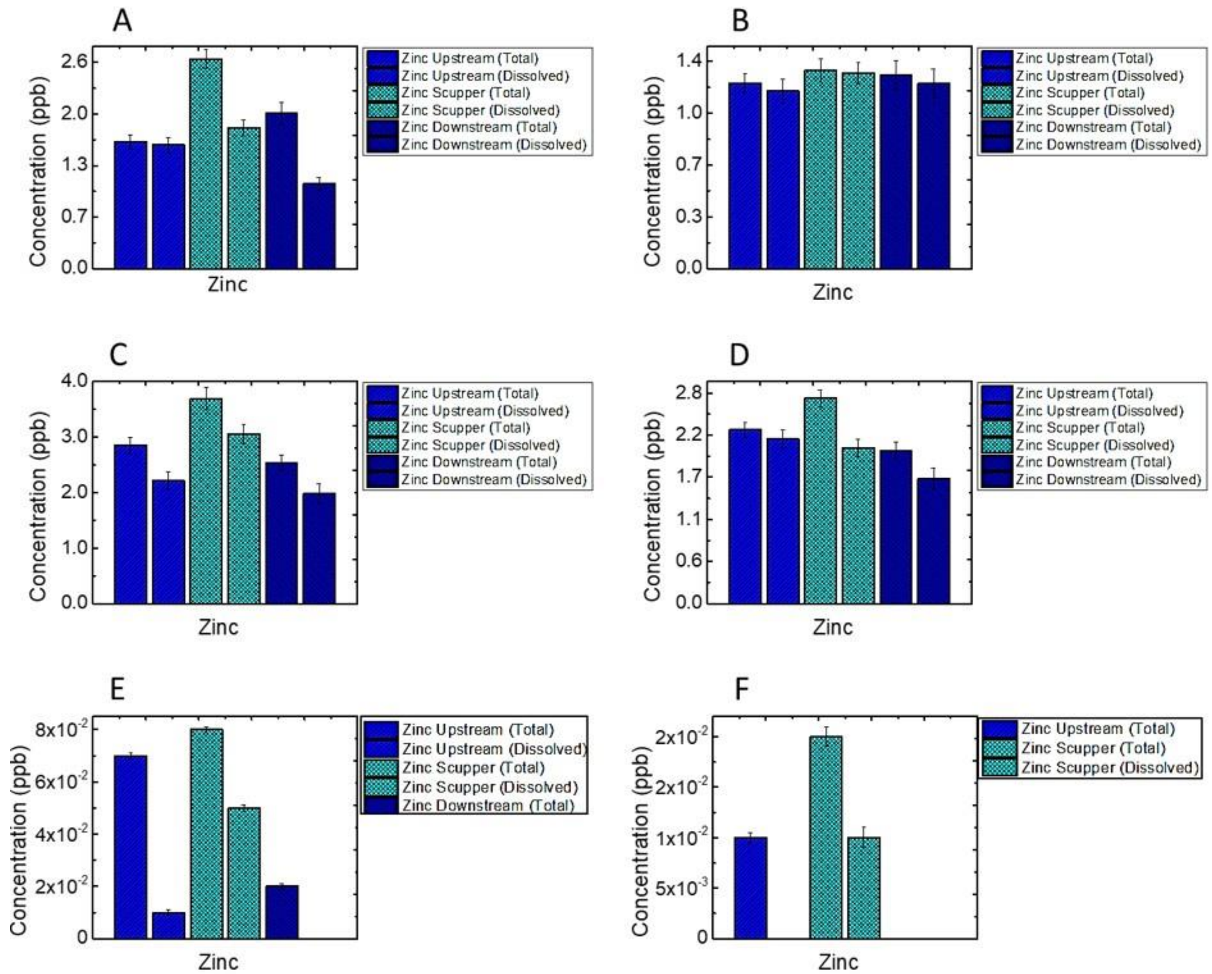

Figure 47. Bar graphs. Zinc concentrations (total and dissolved) from upstream, scupper drains, and downstream for: (A) Site SR 24 on 04/05/2019, (B) Site RF on 10/22/2019, (C) Site US 80 on 10/22/2019, (D) Site SR 297 on 10/22/2020, (E) Site SR 297 on 10/29/2020, and (F) Site US 255 on 01/24/2020.

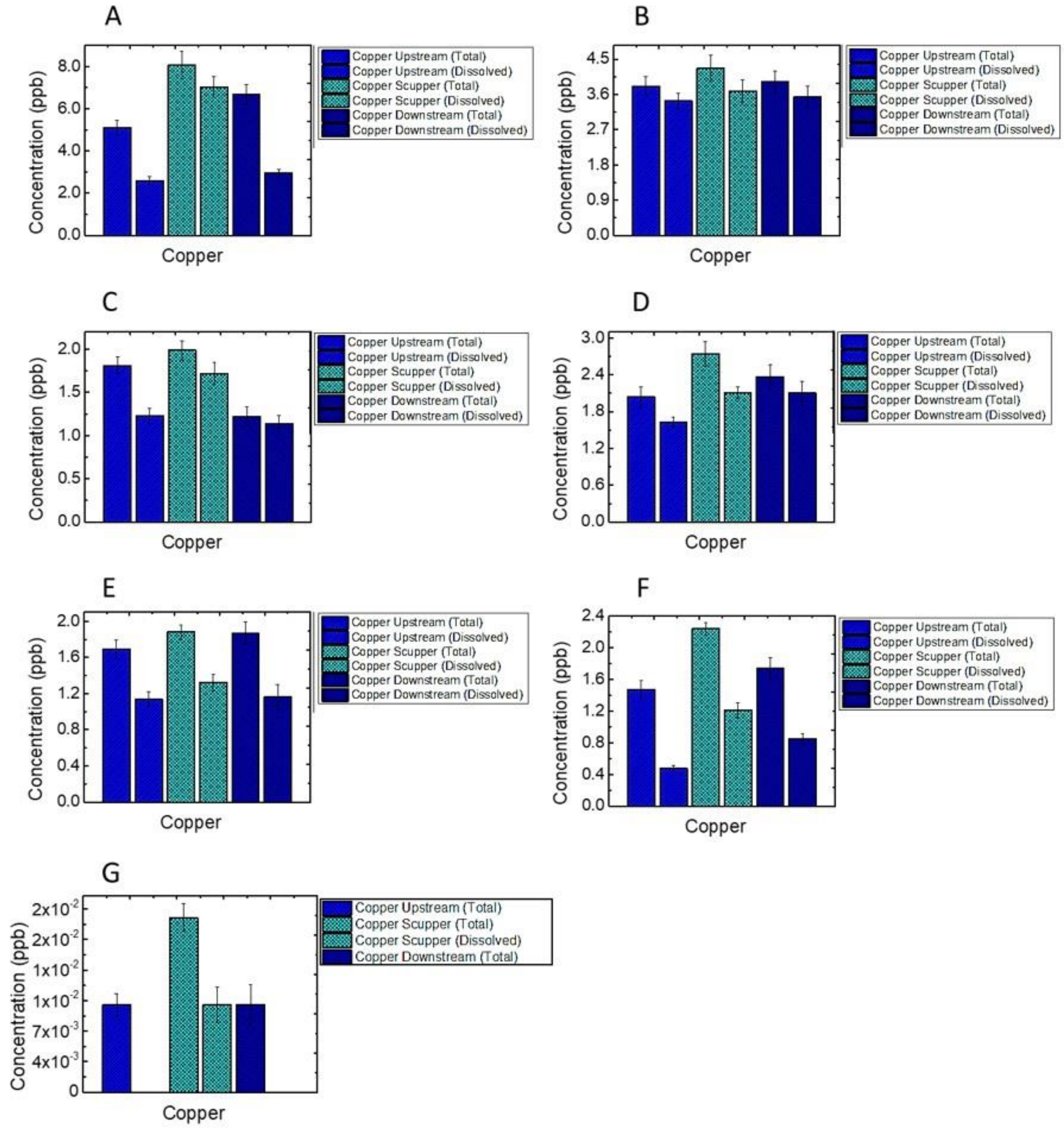

Figure 48. Bar graphs. Copper concentrations (total and dissolved) from upstream, scupper drains, and downstream for: (A) Site SR 24 on 04/05/2019, (B) Site SR 24 on 11/20/2019, (C) Site RF on 10/22/2019, (D) Site US 80 on 10/22/2019, (E) Site SR 297 on 10/22/2019, (F) Site SR 297 on 01/31/2020, and (G) Site US 255 on 01/24/2020.

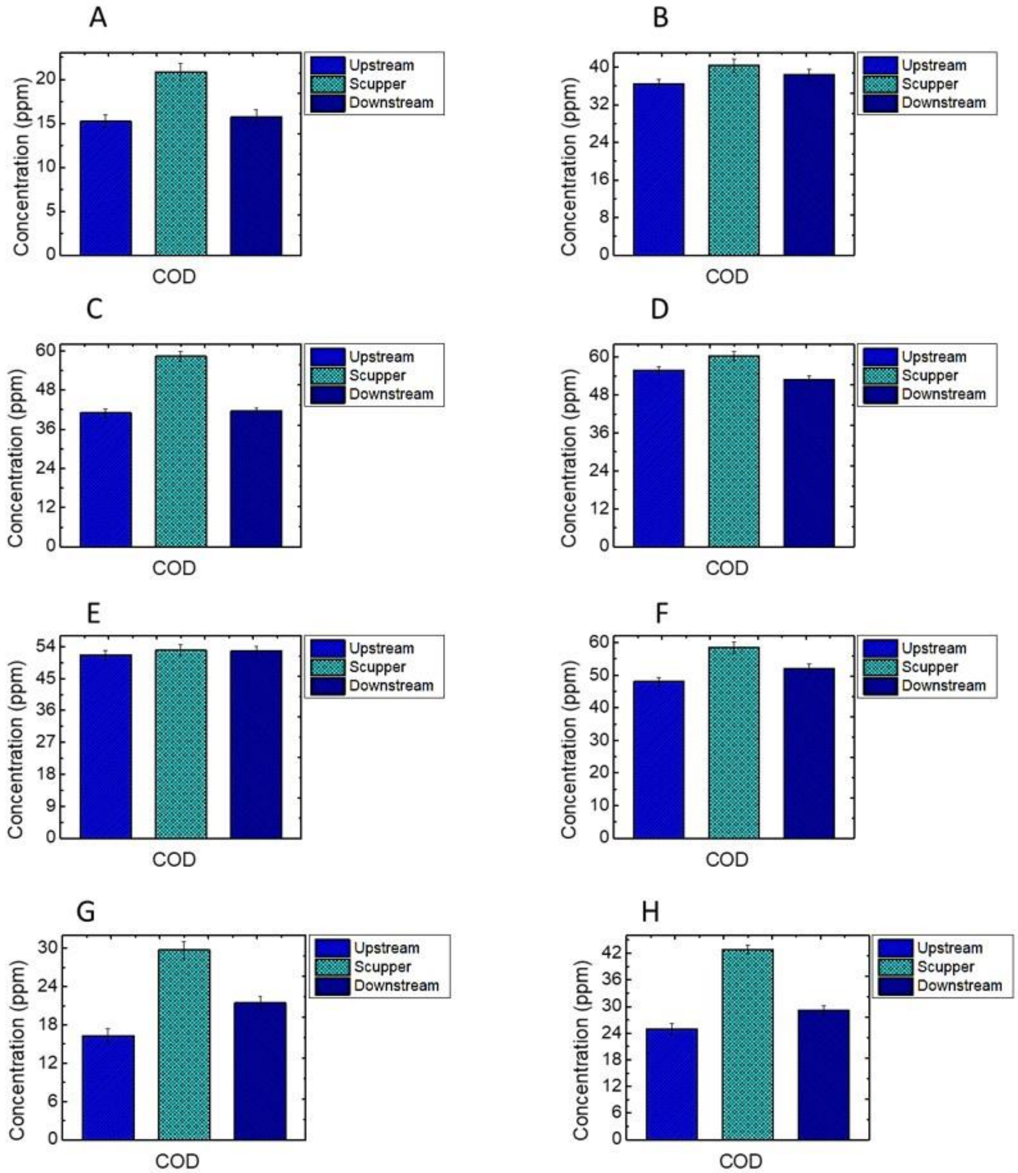

Figure 49. Bar graphs. COD concentrations (total and dissolved) from upstream, scupper drains, and downstream for: (A) Site SR 24 on 04/05/2019, (B) Site SR 24 on 11/20/2019, (C) Site RF on 10/17/2019, (D) Site US 80 on 10/22/2019, (E) Site SR 297 on 10/22/2019, (F) Site SR 297 on 10/29/2019, (G) Site US 255 on 01/24/2020, and (H) Site SR 197 on 01/24/2020.

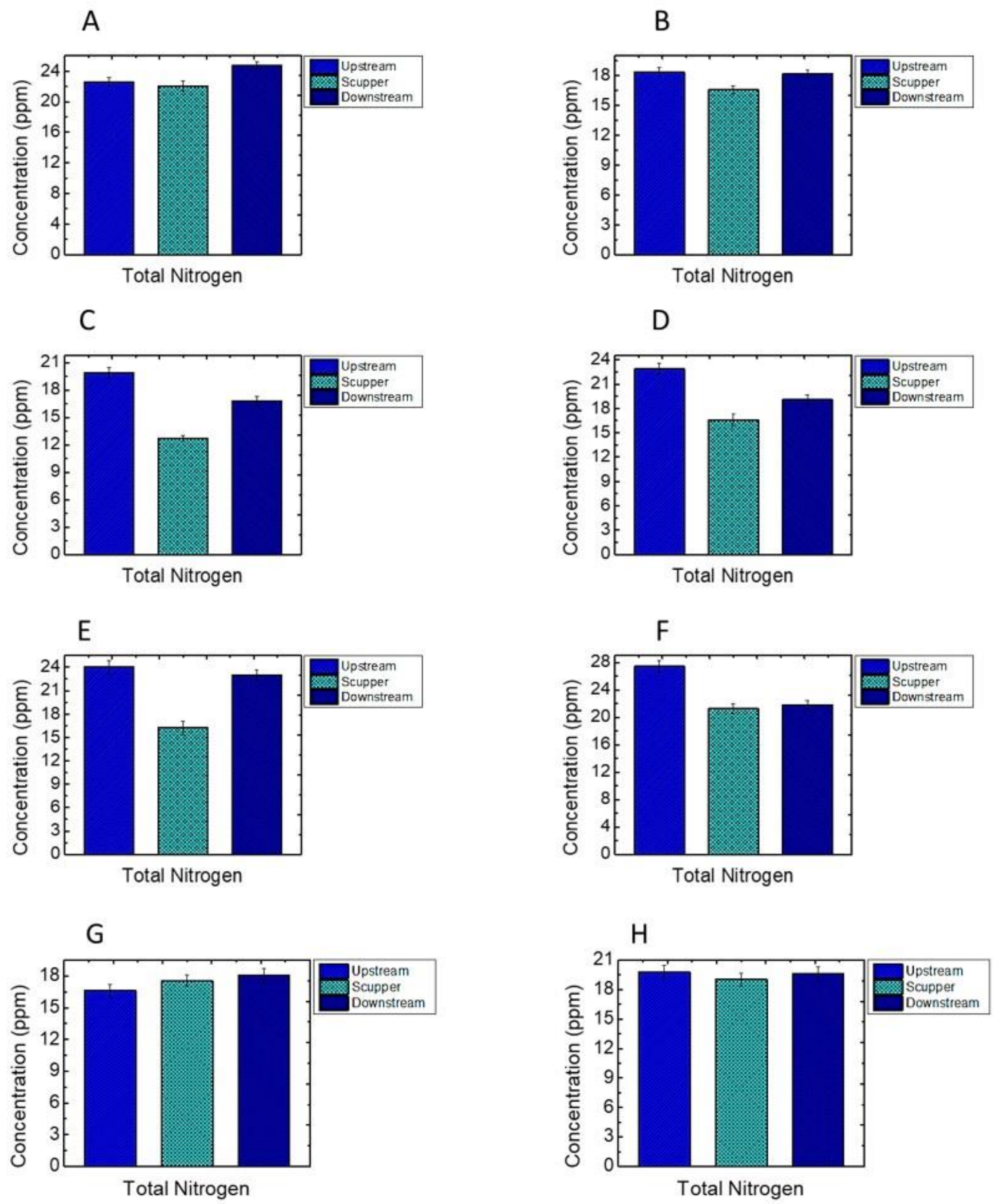

Figure 50. Bar graphs. Total nitrogen concentrations (total and dissolved) from upstream, scupper drains, and downstream for: (A) Site SR 24 on 04/05/2019, (B) Site SR 24 on 11/20/2019, (C) Site RF on 10/22/2019, (D) Site US 80 on 10/17/2019, (E) Site SR 297 on 10/22/2019, (F) Site SR 297 on 10/29/2019, (G) Site US 255 on 01/24/2020, and (H) Site SR 197 on 12/10/2020.

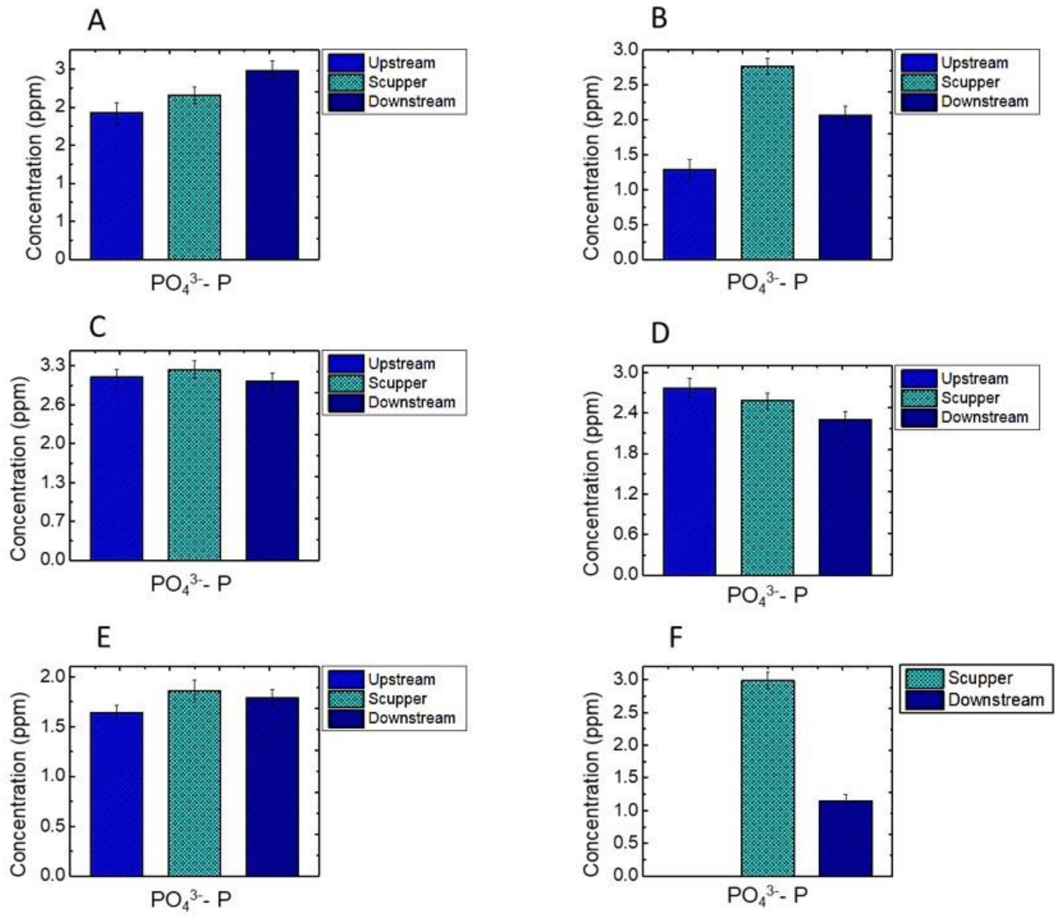

Figure 51. Bar graphs. PO_4^{3-}-P concentrations (total and dissolved) from upstream, scupper drains, and downstream for: (A) Site SR 24 on 04/05/2019, (B) Site SR 24 on 11/20/2019, (C) Site RF on 10/17/2019, (D) Site US 80 on 10/17/2019, (E) Site SR 297 on 10/29/2019, (F) Site SR 297 on 10/29/2019.

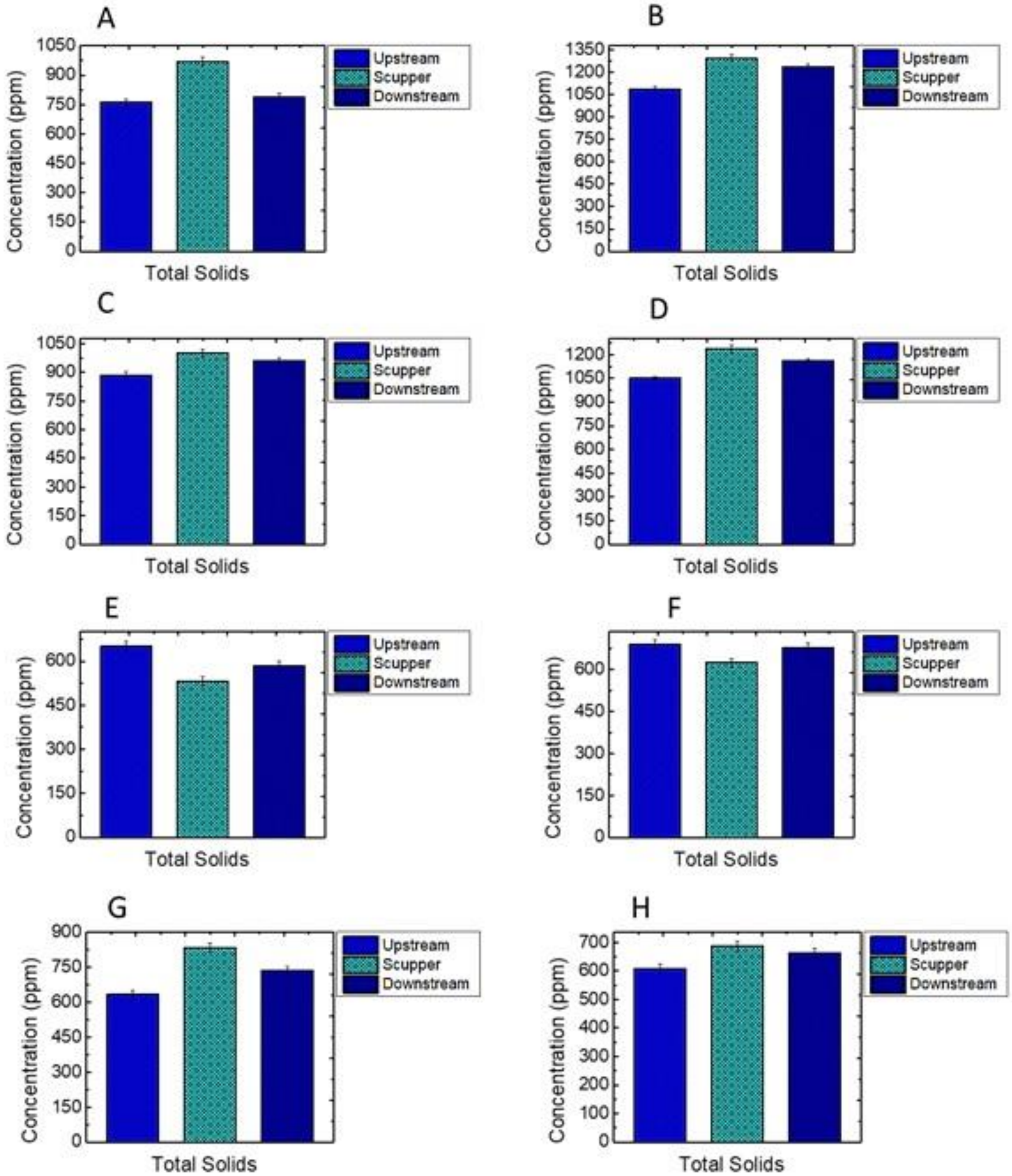

Figure 52. Bar graphs. Total solids concentrations (total and dissolved) from upstream, scupper drains, and downstream for: (A) Site SR 24 on 04/05/2019, (B) Site SR 24 on 11/20/2019, (C) Site RF on 10/17/2019, (D) Site US 80 on 10/17/2019, (E) Site SR 297 on 10/22/2019, (F) Site SR 297 on 10/29/2019, (G) Site US 255 on 12/10/2020, and (H) Site SR 197 on 12/10/2020.

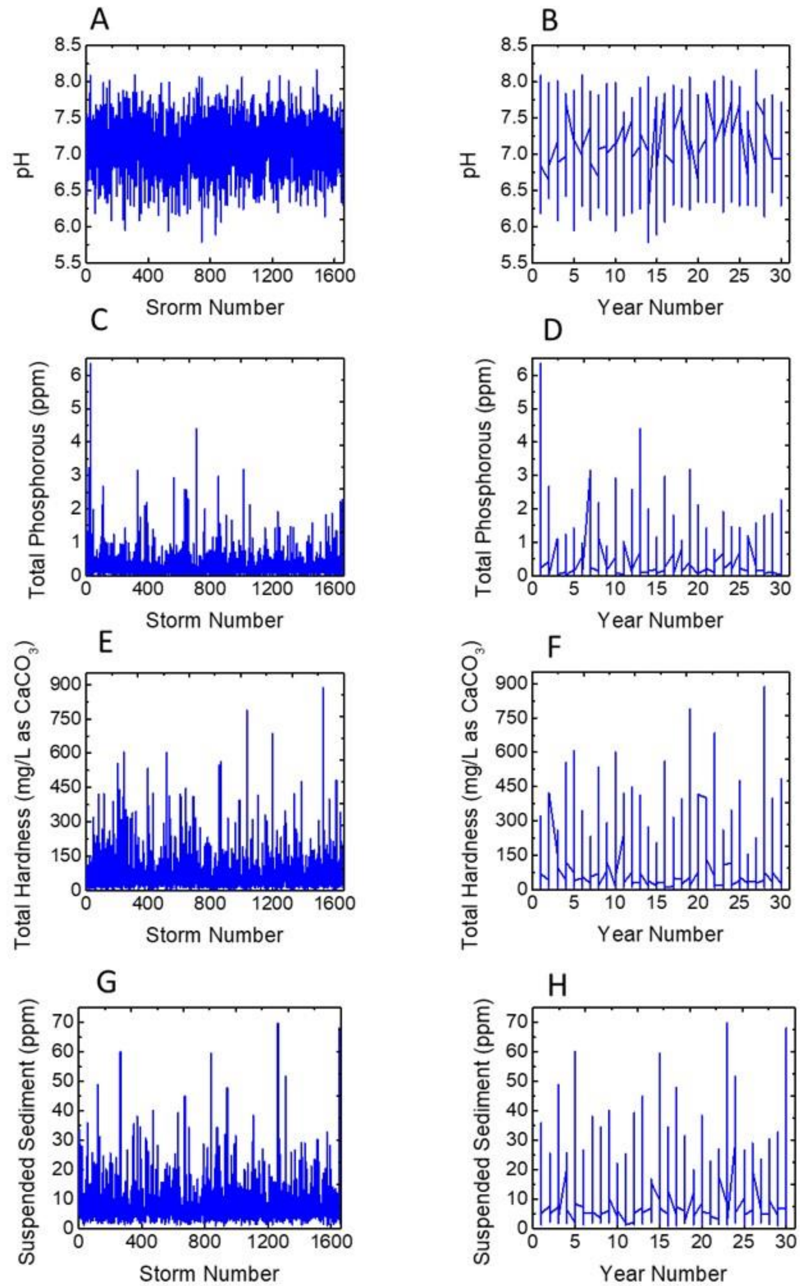

Figure 53. Graphs. SELDM downstream concentrations for Site SR 24: (A) pH versus storm number, (B) pH versus year number, (C) TP versus storm number, (D) TP versus year number, (E) TH versus storm number, (F) TH versus year number, (G) SS versus storm number, and (H) SS versus year number.

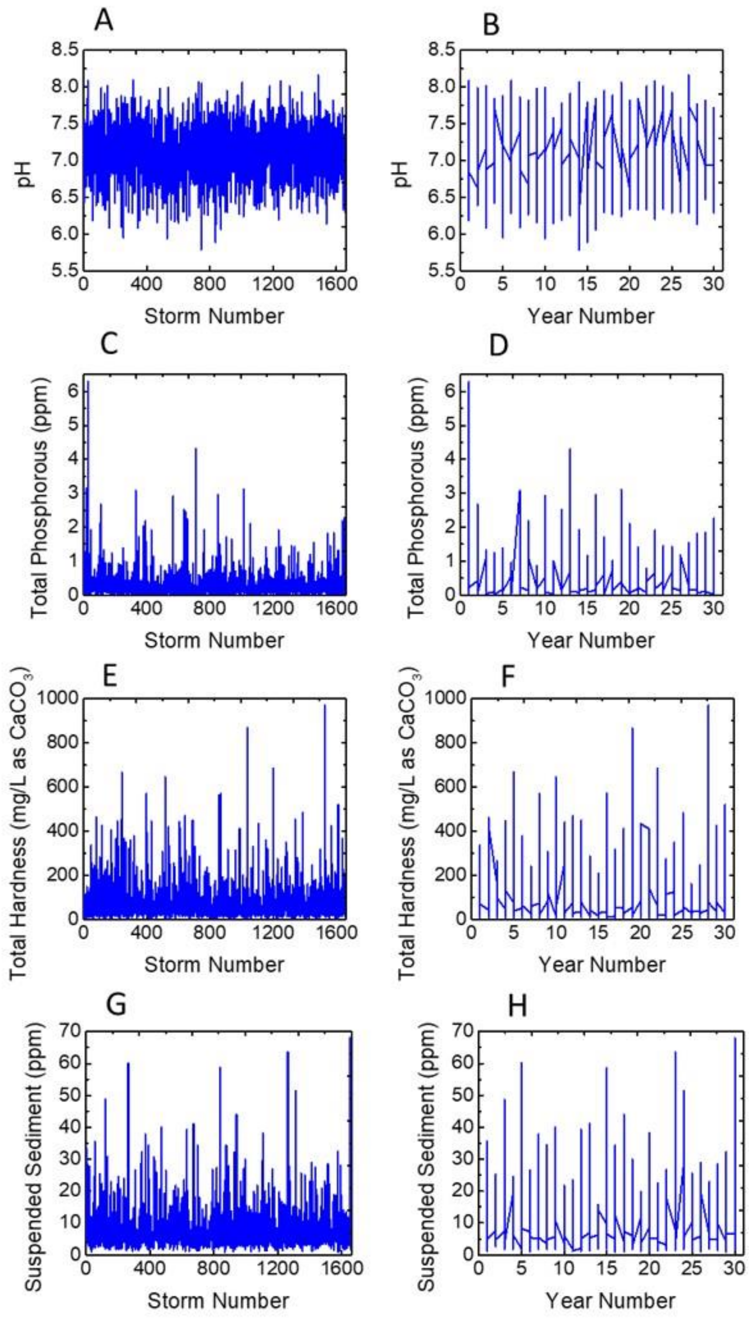

Figure 54. Graphs. SELDM downstream concentrations for Site RF: (A) pH versus storm number, (B) pH versus year number, (C) TP versus storm number, (D) TP versus year number, (E) TH versus storm number, (F) TH versus year number, (G) SS versus storm number, and (H) SS versus year number.

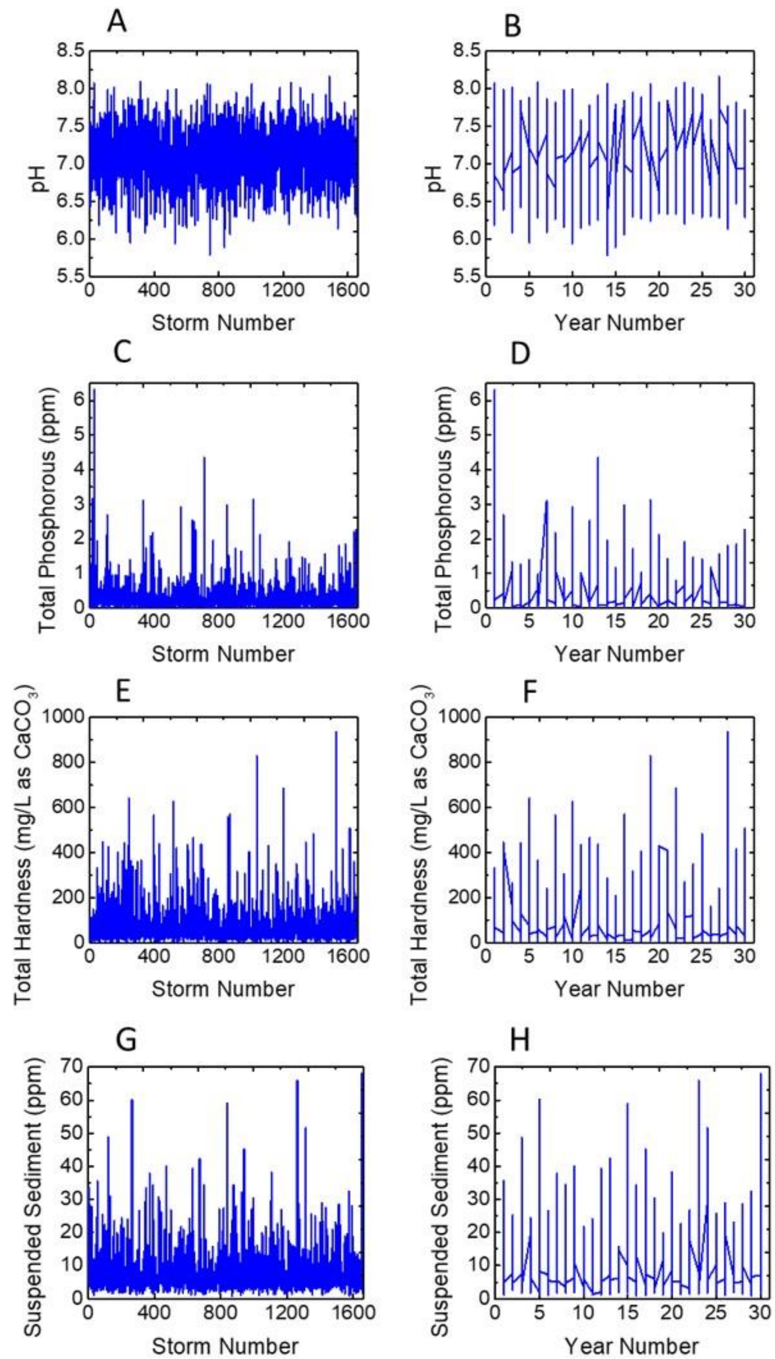

Figure 55. Graphs. SELDM downstream concentrations for Site US 80: (A) pH versus storm number, (B) pH versus year number, (C) TP versus storm number, (D) TP versus year number, (E) TH versus storm number, (F) TH versus year number, (G) SS versus storm number, and (H) SS versus year number.

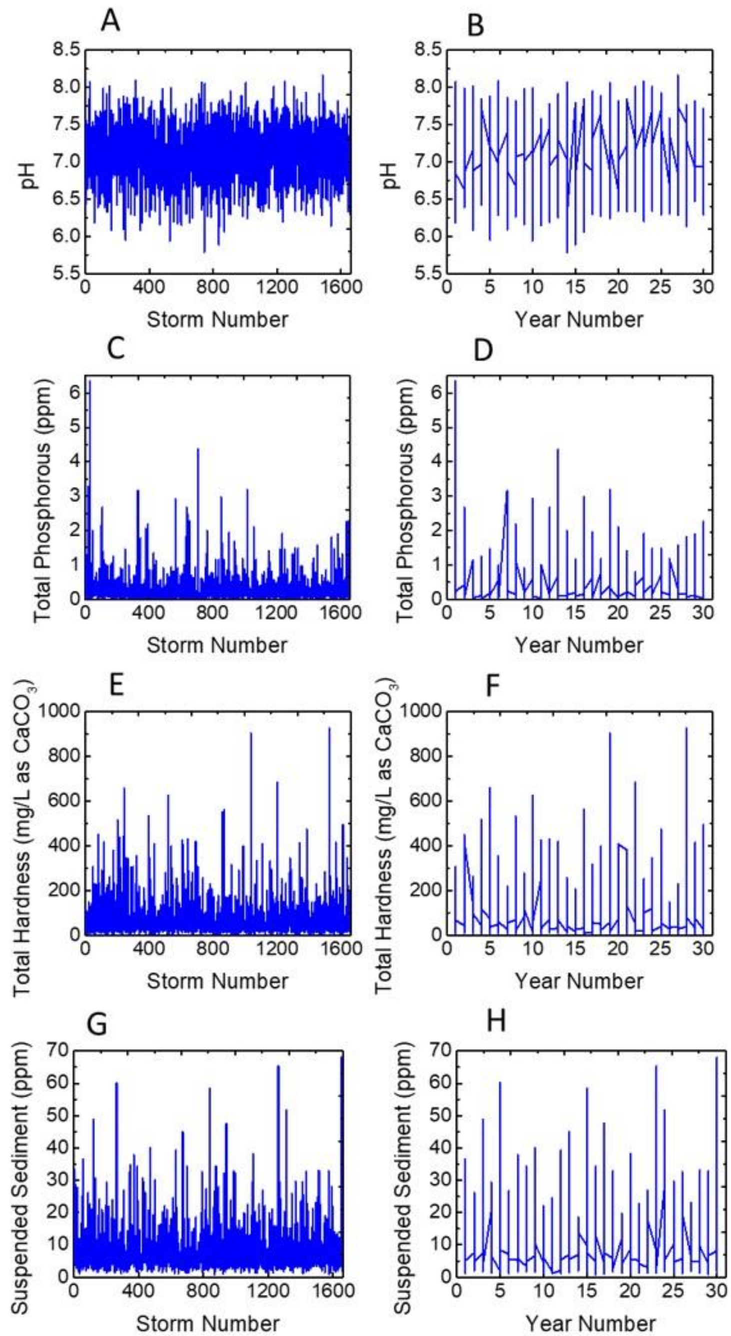

Figure 56. Graphs. SELDM downstream concentrations for Site SR 297: (A) pH versus storm number, (B) pH versus year number, (C) TP versus storm number, (D) TP versus year number, (E) TH versus storm number, (F) TH versus year number, (G) SS versus storm number, and (H) SS versus year number.

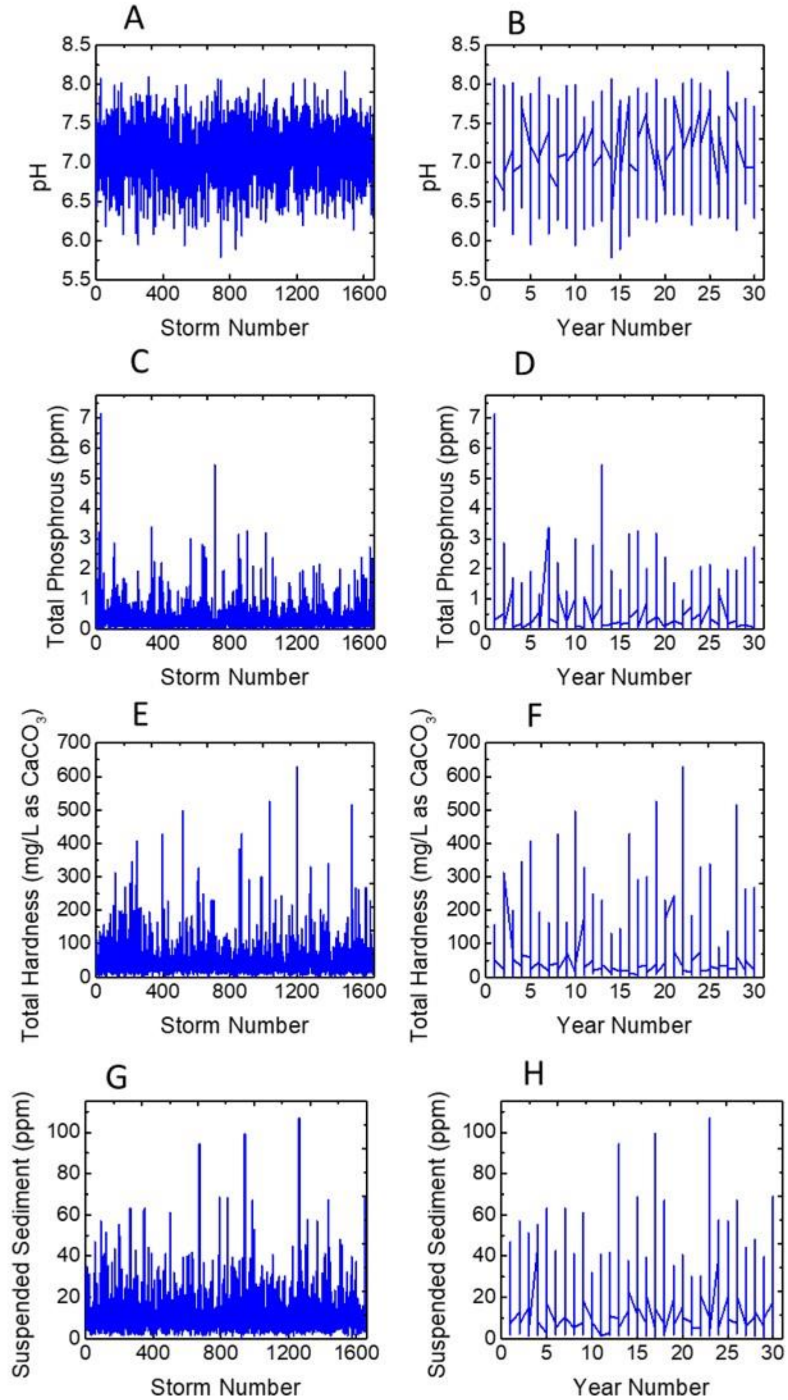

Figure 57. Graphs. SELDM downstream concentrations for Site US 255: (A) pH versus storm number, (B) pH versus year number, (C) TP versus storm number, (D) TP versus year number, (E) TH versus storm number, (F) TH versus year number, (G) SS versus storm number, and (H) SS versus year number.

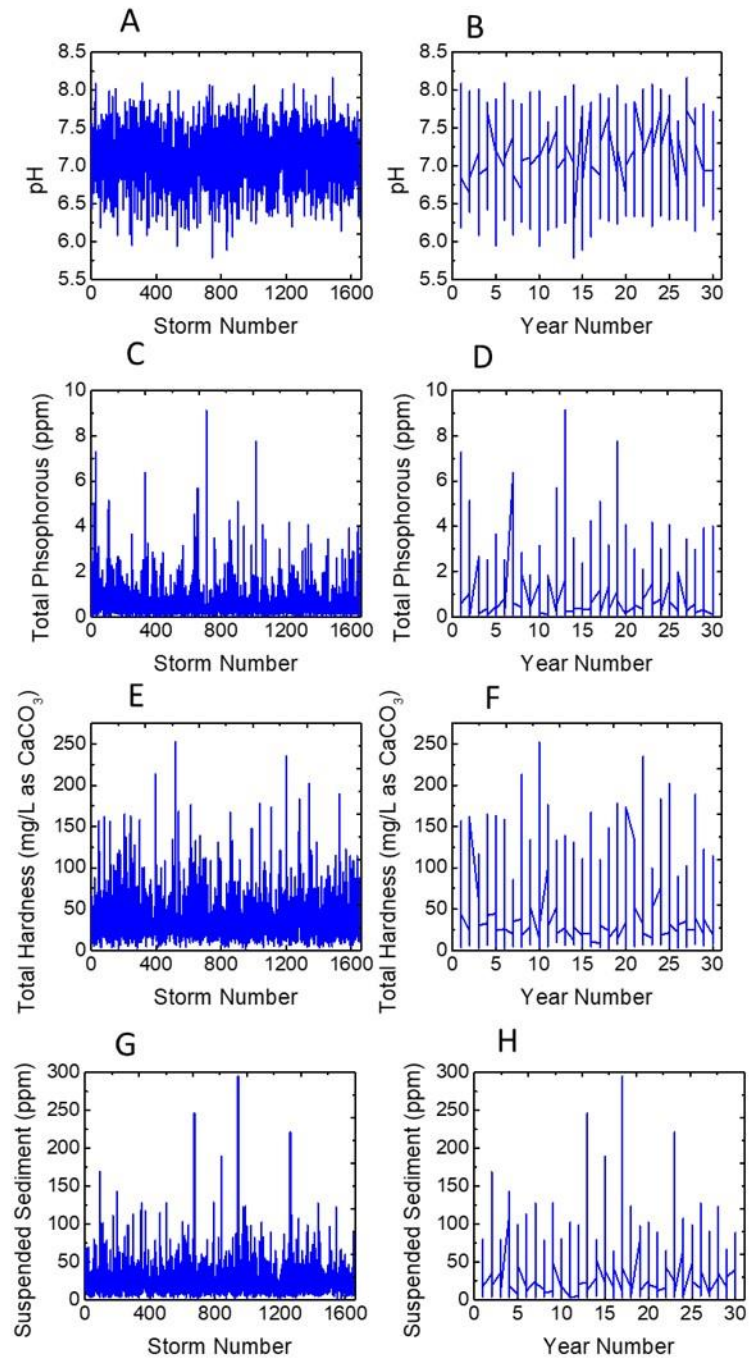

Figure 58. Graphs. SELDM downstream concentrations for Site SR 197: (A) pH versus storm number, (B) pH versus year number, (C) TP versus storm number, (D) TP versus year number, (E) TH versus storm number, (F) TH versus year number, (G) SS versus storm number, and (H) SS versus year number.

ACKNOWLEDGMENTS

The authors acknowledge the support of the Georgia Department of Transportation (GDOT) and the Department of Civil Engineering and Construction (CEC) at Georgia Southern University (GSU).

The GSU research team offers special thanks to Mr. David Hedeen, Ecology Section Manager; Mr. Donn Digamon, State Bridge Engineer; Dr. Anne Sexton-Paperno, Ecologist; Mr. Gary Pierce, P.E.; and Mr. Brennan Roney, Research Engineer, from GDOT for their full support in this research project, including providing technical advice, site selection guidance, and review work. The GSU research team also sends special thanks to the GSU Facilities Department for their help with installations of on-site sampling apparatus; GSU Senior Projects Teams (Fall 2018 and Springs 2019 and 2020) for their participation in this research project; and Mr. Shawn Jackson, Laboratory Coordinator of Civil Engineering in the Department of CEC, for his valuable help with site installations and support on instruments used in this research project. Finally, the GSU research team thanks Mr. Gregory Granato, the Developer of SELDM and Hydrologist, New England Science Center, U.S. Geological Survey, for his insightful advice on how to apply SELDM in this project.

REFERENCES

1. Bakr, A.R., Fu, G.Y., and Hedeen, D. (2020). "Water Quality Impacts of Bridge Stormwater Runoff from Scupper Drains on Receiving Waters: A Review." *Science of The Total Environment*, 726, p. 138068.

2. Alisa, C. (2007). "From Roadkill to Road Ecology: A Review of the Ecological Effects of Roads." *Journal of Transport Geography*, *15*(5), pp. 396–406.

3. Blomberg, J., Jonnson, T. (2016.) *Runoff Management on Bridges: A Decision Model for Management Selection*. Department of Civil and Environmental Engineering. Chalmers University Of Technology, Gothenburg, Sweden.

4. National Academies of Sciences, Engineering, and Medicine. (2014.) *Bridge Stormwater Runoff Analysis and Treatment Options*. The National Academies Press, Washington, DC. https://doi.org/10.17226/22395.

5. URS Corporation. (2012.) *Stormwater Runoff from Bridges*. Final Report to Joint Legislation Transportation Oversight Committee, North Carolina Department of Transporation, Raleigh, NC.

6. Bartelt-Hunt, S.L., Swadener, L., and Jones. E.G. (2012.) *Evaluation of the Environmental Impacts of Bridge Deck Runoff*. Nebraska Department of Transportation, Lincoln, NE.

7. Bartelt-Hunt, S.L., Jones, E.G., and Barrios, R. (2015.) *Characterization of Bridge Deck Runoff*. Nebraska Department of Roads, Lincoln, NE.

8. Malina, J.F., Jr., Barrett, M.E., Jackson, A., and Kramer, T. (2005.) *Characterization of Stormwater Runoff from a Bridge Deck and Approach Highway: Effects on Receiving Water Quality*. Report 0-4543-1, Texas Department of Transportation, Austin, TX.

9. Smith, K.P., Sorenson, J.R., and Granato, G.E. (2018.) *Characterization of Stormwater Runoff from Bridge Decks in Eastern Massachusetts, 2014–16*. U.S. Department of the Interior, U.S. Geological Survey, Reston, VA.

10. Swadener, L.M., Javeherian, M.H., Jones, E.G., and Bartelt-Hunt, S.L. (2014.) "Chemical Characterization and Toxicity of Bridge Deck Runoff and Impacts to Receiving Water Quality." *Journal of Environmental Engineering*, *140*(4), p. 04014010.

11. Nwaneshiudu, O. *Assessing Effects Of Highway Bridge Deck Runoff On Nearby Receiving Waters In Coastal Margins Using Remote Monitoring Techniques*. Master's thesis, Texas A&M University, College Station, TX.

12. Perkins, R.A. and Hazirbaba, Y.D. (2010.) *Bridge Deck Runoff: Water Quality and Best Management Practices Effectiveness*. Final Report RR08.13, Alaska University Transportation Center, Alaska Department of Transportation and Public Facilities, Juneau, AK.

13. Tan, S-G., Liu, X-X., Zou, G-P., Xiong, X-Z., and Tao, S-C. (2018.) "Discussion on Runoff Purification Technology of Highway Bridge Deck Based on Water Quality Safety." *E3S Web of Conferences*, *38*, p. 03043.

14. Dupuis, T.V. (2002). *Assessing the Impacts of Bridge Deck Runoff Contaminants in Receiving Waters.* NCHRP Report, Project B25-13, Transportation Research Board, Washington, DC.

15. Winston, R.J., Lauffer, M.S., Narayanaswamy, K., McDaniel, A.H., Lipscomb, B.S., Nice, A.J., Hunt, W.F. (2015.) "Comparing Bridge Deck Runoff and Stormwater Control Measure Quality in North Carolina." *Journal of Environmental Engineering*, *141*(1), p. 04014045.

16. Goonetilleke, A., Liu, A., Managi, S., Wilson, C., Gardner, T., Bandala, E.R., Walker, L., Holden, J., Wibowo, M.A., Suripin, S., Joshi, H., Bonotto, D.M., Rajapaksa, D. (2017.) "Stormwater Reuse, A Viable Option: Fact or fiction?" *Economic Analysis and Policy*, *56*, pp. 14–17.

17. Barbosa, A.E., Fernandes, J.N., and David, L.M. (2012.) "Key Issues for Sustainable Urban Stormwater Management." *Water Resources*, *46*(20), pp. 6787–6798.

18. Qiao, X-J., Kristoffersson, A., Randrup, T.B. (2018.) "Challenges to Implementing Urban Sustainable Stormwater Management from a Governance Perspective: A Literature Review." *Journal of Cleaner Production*, *196*, pp. 943–952.

19. U.S. Environmental Protection Agency (EPA). (1996.) *National Water Quality Inventory Report to Congress*. U.S. Environmental Protection Agency, Washington, DC.

20. U.S. Environmental Protection Agency (EPA). (2005.) *National Management Measures to Control Nonpoint Source Pollution from Urban Areas*. EPA-841-B-05-004, U.S. Environmental Protection Agency, Office of Water, Washington, DC.

21. Grant, S.B., Rekhi, N.V., Pise, N.R., Reeves, R.L., et al. (2003.) *A Review of The Contaminants and Toxicity Associated with Particles in Stormwater Runoff*. CTSW-RT-03-059.73.15, California Department of Transporation, Sacramento, CA.

22. Trombulak, S.C. and Frissell, C.A. (2002.) "Review of Ecological Effects of Roads on Terrestrial and Aquatic Communities." *Conservation Biology*, *14*(1), pp. 18–30.

23. Field, R. and Sullivan, D. (eds.). (2002.) *Wet-Weather Flow in the Urban Watershed: Technology and Management*. 1st edition. CRC Press, Boca Raton, FL.

24. Jongedyk, H. and Bank, F. (1999.) *FHWA Environmental Technology Brief: Is Highway Runoff a Serious Problem?* Federal Highway Administration, Washington DC.

25. Kayhanian, M., Singh, A., Suverkropp, C., and Borroum, S. (2003.) "Impact of Annual Average Daily Traffic on Highway Runoff Pollutant Concentrations." *Journal of Environmental Engineering*, *129*(11), pp. 975–990.

26. Buckler, D.R. and Granato, G. (1999.) *Assessing Biological Effects from Highway-Runoff Constituents*. Report 99-240, U.S. Geological Survey, Northborough, MA.

27. Lau, S-L., Han, Y., Kang, J-H., Kayhanian, M., and Strenstrom, M.K. (2009.) "Characteristics of Highway Stormwater Runoff in Los Angeles - Part 2: Metals and Polycyclic Aromatic Hydrocarbons." *Water Environment Research*, *81*(3), pp. 308–318.

28. Brown, J.N. and Peake, B.M. (2006.) "Sources of Heavy Metals and Polycyclic Aromatic Hydrocarbons in Urban Stormwater Runoff." *Science of the Total Environment*, *359*(1–3), pp. 145–155.

29. Lundy, L., Ellis, J.B., Revitt, D.M. (2012.) "Risk Prioritisation of Stormwater Pollutant Sources." *Water Resources*, *46*(20), pp. 6589–6600.

30. Bound, J.P. and Voulvoulis, N. (2004.) "Pharmaceuticals in the Aquatic Environment—A Comparison of Risk Assessment Strategies. *Chemosphere, 56*(11), pp. 1143–1155.

31. Staples, C.A., Parkerton, T.F., and Peterson, D.R. (2000). "A Risk Assessment of Selected Phthalate Esters in North American and Western European Surface Waters." *Chemosphere*, *40*(8), pp. 885–891.

32. Obropta Christopher, K.J., *Review of Urban Stormwater Quality Models: Deterministic, Stochastic, and Hybrid Approaches1.* JAWRA Journal of the American Water Resources Association, 2007. 43(6): p. 1508-1523.

33. Zhang Z., L.W.C., Panu U.S., *Effect of percent nondetects on estimation bias in censored distributions.* Journal of Hydrology, 2004. 297(1-4): p. 74-94.

34. German J., V.M., Svensson G., Gustafsson L.-G. Integrated stormwater strategies to reduce impact on receiving waters. in 10th Annual Conference on Urban Drainage. 2005. Copenhagen/Denmark.

35. Elliott A.H., T.S.A., *A review of models for low impact urban stormwater drainage.* Environmental Modelling & Software, 2007. 22(3): p. 394-405.

36. Buddhi Wijesiri, P.E., James McGree, Ashantha Goonetilleke, Understanding the uncertainty associated with particle-bound pollutant build-up and wash-off: A critical review. Water Res, 2016. 101: p. 582-596.

37. Sawyer Clair, M.P., Parkin Gene, , *Chemistry for Environmental Enggineering and Science*. 5th ed. 2003.

38. Newman James, S.K., Novakova E., *Air Pollution Effects on Terrestrial and Aquatic Animals*, in *Air Pollution Effects on Biodiversity*, J.R. Barker and D.T. Tingey, Editors. 1992, Springer US: Boston, MA. p. 177-233.

39. Carpenter, S.R., Caraco, N. F., Correll, D. L., Howarth, R. W., Sharpley, A. N., Smith, V. H., *Nonpoint pollution of surface waters with phosphorus and nitrogen.* Ecological Applications, 1998. 8(3): p. 559-568.

40. *Georgia stormwater management manual*. First ed. Vol. 1. 2001: Atlanta Regional Commission.

41. Kirk Smith, G.G., Quality of Stormwater Runoff Discharged from Massachusetts Highways, 2005–07. 2009.

42. Wayne Huber, J.H., Eric Strecker, Neil Weinstein, Evaluation Of Best Management Practices For Highway Runoff Control: Low Impact Development Design Manual for Highway Runoff Control. 2006, Oregon State University.

43. Amy Rowe, T.O.C., Assessment of Water Quality of Runoff from Sealed Asphalt Surfaces. 2011.

44. Wright Ian, D.P., Findlay Sophia, Jonasson Olof, *A new type of water pollution: concrete drainage infrastructure and geochemical contamination of urban waters.* Marine and Freshwater Research, 2011. 62(12): p. 1355-1361.

45. William, M., Variations in Highway Stormwater Runoff Quality and Stormwater Treatment Performance in Relation to the Age of Porous Friction Courses, J. P. Moores, P. E. Pattinson, C. R. Hyde, 85, 772–781 (2013); corrected at 86, 285 (2014). Water Environment Research, 2014. 86(3): p. 286-287.

46. Dong Trang, L.B.-K., Characteristics, toxicity, and source apportionment of polycyclic aromatic hydrocarbons (PAHs) in road dust of Ulsan, Korea. Chemosphere, 2009. 74(9): p. 1245-1253.

47. Hvitved–Jacobson T., Y.Y.A., *Chapter 5 Highway Runoff Quality, Environmental Impacts and Control*, in *Studies in Environmental Science*, R.S. Hamilton and R.M. Harrison, Editors. 1991, Elsevier. p. 165-208.

48. Huber Maximilian, W.A., Helmreich Brigitte, Critical review of heavy metal pollution of traffic area runoff: Occurrence, influencing factors, and partitioning. Science of The Total Environment, 2016. 541: p. 895-919.

49. Fallah Shorshani Masoud, B.C., Petrucci Guido, André Michel, Seigneur Christian, *Road traffic impact on urban water quality: a step towards integrated traffic, air and stormwater modelling.* Environmental Science and Pollution Research, 2014. 21(8): p. 5297-5310.

50. Davis Brett, B.G., Comparison of heavy metal loads in stormwater runoff from major and minor urban roads using pollutant yield rating curves. Environmental Pollution, 2010. 158(8): p. 2541-2545.

51. Yukun Ma, A.L., Prasanna Egodawatta, James McGree, Ashantha Goonetilleke, Assessment and management of human health risk from toxic metals and polycyclic aromatic

hydrocarbons in urban stormwater arising from anthropogenic activities and traffic congestion. Sci Total Environ, 2017. 579: p. 202-211.

52. Driscoll E.D., S.P.E., Strecker E.W., *Pollutant Loadings and Impacts from Highway Stormwater Runoff: Analytical Investigation and Research Report.* 1990, Federal Highway Administration: Federal Highway Administration.

53. An Liu, Y.M., Kaveh Deilami, Prasanna Egodawatta, Ashantha Goonetilleke, *Ranking the factors influencing polycyclic aromatic hydrocarbons (PAHs) build-up on urban roads.* Ecotoxicol Environ Saf, 2017. 139: p. 416-422.

54. Karlsson Kristin, V.M., Scholes Lian, Revitt Mike, *Heavy metal concentrations and toxicity in water and sediment from stormwater ponds and sedimentation tanks.* Journal of Hazardous Materials, 2010. 178(1): p. 612-618.

55. Lygren E., G.E., Berglind, L., *Pollution transport from a highway.* Science of The Total Environment, 1984. 33(1): p. 147-159.

56. Dupuis Thomas, K.N., Effects of Highway Runoff on Receiving Waters: Procedural Guidelines for Environmental Assessments. 1985: Fedral Highway Administration.

57. Kayhanian Masoud, S.A., Suverkropp Claus, Borroum Steve, *Impact of Annual Average Daily Traffic on Highway Runoff Pollutant Concentrations.* Journal of Environmental Engineering, 2003. 129(11): p. 975-990.

58. Tom Murdoch, M.C. *Stream Flow*. Water: Monitoring & Assessment Available from: https://archive.epa.gov/water/archive/web/html/vms51.html.

59. Meybeck M., F.G., Thomas R., Chapman D., Water Quality Assessments - A Guide to Use of Biota, Sediments and Water in Environmental Monitoring. 1992, 1996, UNESCO/WHO/UNEP.

60. Zgheib S., M.R., Chebbo G., Priority pollutants in urban stormwater: part 1 - case of separate storm sewers. Water Res, 2012. 46(20): p. 6683-92.

61. Bressy A., G.M.C., Lorgeoux C., Saad M., Leroy F., Chebbo G., Towards the determination of an optimal scale for stormwater quality management: Micropollutants in a small residential catchment. Water Research, 2012. 46(20): p. 6799-6810.

62. Kim Lee-Hyung, K.S.-O., Jeong Sangman, Yoon Jaeyoung, *Characteristics of Washed off Pollutants and Dynamic EMCs in Parking Lots and Bridges During a Storm.* Science of the Total Environment, 2007. 376.1-3: p. 178-184.

63. Santosh, N., Characterization of Bridge Discharge to Receiving Streams. 2004.

64. Kim Lee-Hyung, K.S.-O., Jeong Sangman, Yoon Jaeyoung, *Characteristics of washed-off pollutants and dynamic EMCs in parking lots and bridges during a storm.* Science of The Total Environment, 2007. 376(1): p. 178-184.

65. Masoud Kayhanian, B.F., John Gulliver, Comasia Montanaro, Ezio Ranieri, Stefan Wuertz, *Review of highway runoff characteristics: comparative analysis and universal implications.* Water Res, 2012. 46(20): p. 6609-24.

66. Feng Xiao, M.S., John Gulliver, *Perfluoroalkyl acids in urban stormwater runoff: influence of land use.* Water Res, 2012. 46(20): p. 6601-8.

67. Yasuyuki Zushi, S.M., First-flush loads of perfluorinated compounds in stormwater runoff from Hayabuchi River basin, Japan served by separated sewerage system. Chemosphere, 2009. 76(6): p. 833-40.

68. Ariamalar Selvakumar, M.B., Variation of microorganism concentrations in urban stormwater runoff with land use and seasons. Journal of Water and Health, 2006. 04.1.

69. Morace, J., Reconnaissance of Contaminants in Selected Wastewater-Treatment-Plant Effluent and Stormwater Runoff Entering the Columbia River, Columbia River Basin, Washington and Oregon, 2008–10. 2012.

70. Fletcher T. D., A.H., Hamel P., Understanding, management and modelling of urban hydrology and its consequences for receiving waters: A state of the art. Advances in Water Resources, 2013. 51: p. 261-279.

71. Lainé S., P.T., Dufay S., Baron J., Robert P., Treatment of stormwater to bathing water quality by dissolved air flotation, filtration and ultraviolet disinfection. Water Science and Technology, 1998. 38(10): p. 99-105.

72. Pavia Edgar, P.C., *Stormwater Disinfection at New Orleans.* Journal (Water Pollution Control Federation), 1969. 41(4): p. 591-606.

73. Shirley Clark, R.P., Targeting treatment technologies to address specific stormwater pollutants and numeric discharge limits. Water Res, 2012. 46(20): p. 6715-30.

74. Granato, G. *SELDM: Stochastic Empirical Loading and Dilution Model - Project Page*. 2013; Available from: https://www.usgs.gov/centers/new-england-water/science/seldm-stochastic-empirical-loading-and-dilution-model-project-page?qt-science_center_objects=0#qt-science_center_objects.

75. Granato, G. *SELDM: Stochastic Empirical Loading and Dilution Model - Software page*. 2013; Available from: https://www.usgs.gov/software/seldm-stochastic-empirical-loading-and-dilution-model-software-page.

76. Stonewall A.J., G.G.E., Haluska T.L, Assessing roadway contributions to stormwater flows, concentrations and loads by using the StreamStats application: in Compendium of Papers for the Transportation Research Board 97th Annual Meeting. 2018: Washington, D.C.

77. Granato, G. and S. Jones, Estimating the risks for adverse effects of total phosphorus in receiving streams with the Stochastic Empirical Loading and Dilution Model (SELDM). 2015.

78. Granato, G.E. and S.C. Jones, Simulating Runoff Quality with the Highway Runoff Database and the Stochastic Empirical Loading and Dilution Model. Transportation Research Record, 2019. 2673(1): p. 136-142.

79. *USGS StreamStats*. Available from: https://streamstats.usgs.gov/ss/.

80. *EPA takes final step in phaseout of leaded gasoline*. 1996; Available from: https://archive.epa.gov/epa/aboutepa/epa-takes-final-step-phaseout-leaded-gasoline.html.

81. *USGS 02202190 Ogeechee River At Ga 24, Near Oliver, Ga*. Available from: https://waterdata.usgs.gov/nwis/dv?cb_00060=on&format=html&site_no=02202190&referred_module=sw&period=&begin_date=2019-11-20&end_date=2019-11-20.

82. *USGS 02202040 Ogeechee River At Rocky Ford Rd, Nr Rocky Ford, Ga*. Available from: https://waterdata.usgs.gov/nwis/uv?site_no=02202040.

83. *USGS 02202500 Ogeechee River Near Eden, Ga*. Available from: https://waterdata.usgs.gov/nwis/inventory/?site_no=02202500&agency_cd=USGS.

84. *USGS 02225270 Ohoopee River At Ga 297, Near Swainsboro, Ga*. Available from: https://waterdata.usgs.gov/nwis/inventory/?site_no=02225270&agency_cd=USGS.

85. *USGS 02331000 Chattahoochee River Near Leaf, Ga*. Available from: https://waterdata.usgs.gov/ga/nwis/dv?cb_00060=on&format=html&site_no=02331000&referred_module=sw&period=&begin_date=2020-01-24&end_date=2020-01-24.

86. *Water: Monitoring & Assessment*. Available from: https://archive.epa.gov/water/archive/web/html/vms59.html.