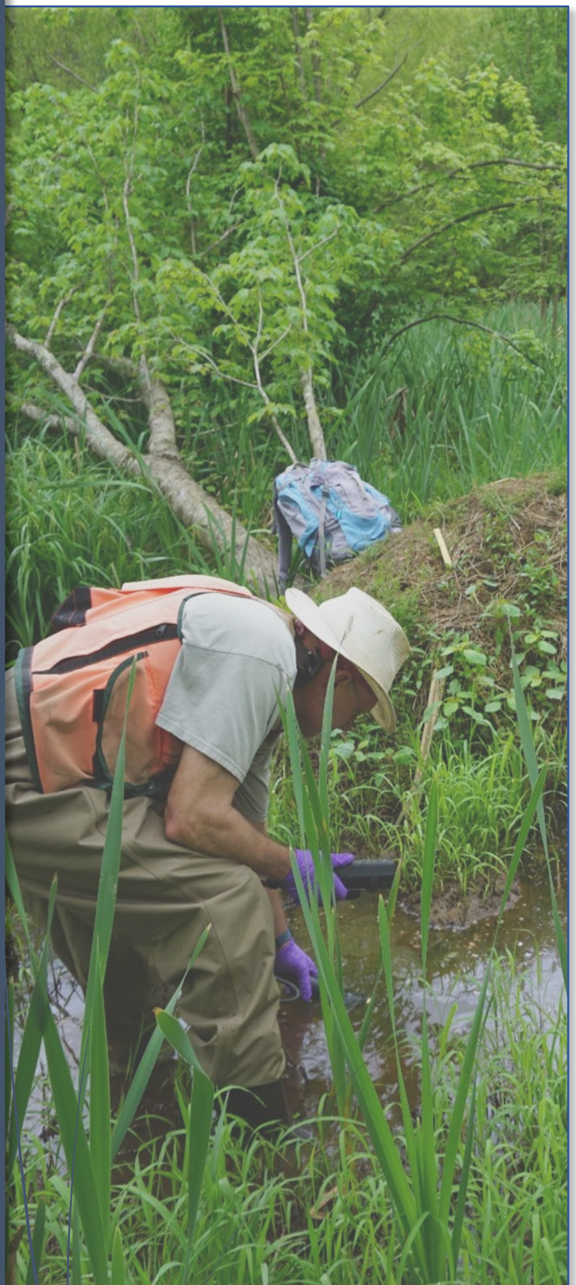

Quality of Surface Water Entering and Leaving Natural Wetlands in an Urban Setting

Final Project Report

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The NC Department of Environmental Quality

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Water Sciences Section

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List of Abbreviations

DO	Dissolved Oxygen
NC DEQ	North Carolina Department of Environmental Quality
NC DWR	North Carolina Division of Water Resources
NH_3	Ammonia
$NO_3^- + NO_2^-$	Nitrate + Nitrite
TKN	Total Kjeldahl Nitrogen
TP	Total Phosphorus
TSS	Total Suspended Solids
USACE	US Army Corps of Engineers
US EPA	United States Environmental Protection Agency
USGS	United States Geological Survey
WSS	Water Sciences Section (of NC DWR)
Wetland Site Names	
APH	Apollo Heights
HR	Hammond Rd
WP	Women’s Prison
ZH	Zhang Property

Distribution List

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Executive Summary

Impacts to wetlands and streams are disproportionately occurring in urban areas, both from newly permitted projects and from lack of stormwater treatment from historical development. Data have been generally lacking, including in North Carolina, on water quality functions of natural wetlands receiving stormwater runoff in urban settings.

This project documented water quality in natural wetland systems in an urbanized setting. Concentrations of pollutants were compared in natural urban wetlands at permanently flowing inlet and outlet points to understand the impacts, if any, of natural urban wetlands on the quality of water entering Walnut Creek from the wetlands. Water quality samples were analyzed for oil and grease, metals (copper, lead, and zinc), nutrients (nitrogen and phosphorus), and suspended solids. Along with field data collection in natural wetlands, literature on water quality benefits of constructed stormwater wetlands was compiled and summarized for comparison to natural wetland systems in urban settings.

Oil and grease were nearly always undetectable in water samples. Nutrient concentrations were generally low enough to be considered at background levels, in inlets and outlets. Nitrate + nitrite was the only parameter to show statistically significantly different (lower) concentrations at outlets compared to inlets under all conditions (baseflow/storm; growing season/non-growing season). During baseflow conditions at three of the four intensive study sites, copper, lead, and zinc concentrations were higher in wetland centers than at inlets, suggesting sequestering of metals by the large amounts of algal biomass commonly present in the wetland centers and/or binding to the sediments in the wetland interiors. This difference was not present during storm flow, probably because of dilution. Lead levels in the wetland soils were high in some wetlands, indicating that legacy contamination from leaded fuel had been sequestered in wetland sediments. During the study, lead generally stayed in the wetland soils and was not proportionately re-suspended into the output water. Total suspended solids were significantly higher in the water at outlets than inlets at baseflow times, but this was found to be attributable to three individual samples, including one from an outlet that had recently been dammed by beavers. Furthermore, a detailed sediment movement assessment at one study wetland suggested that overall sediment coming into the wetlands during storms

was a much greater volume than that exiting the outlet through bank erosion. This study recorded a significant accumulation of sediment in the wetland from inlets, only 7% of which was eroding at the outlets to the mainstem Walnut Creek. The urban wetlands in this study were functioning to capture and hold back sediment which otherwise would have entered Walnut Creek and subsequently the Neuse River. Large quantities of discarded trash, mostly plastic, have also accumulated in these urban wetlands, providing a capture mechanism that would be absent without them. Concentrations of measured contaminants were usually quite low for urban surface waters; these low concentrations underscore the function and importance of intact vegetated buffers on streams and waterbodies in urban areas.

A literature review of constructed stormwater wetland performance was conducted as part of this project (Appendix H). Constructed stormwater wetlands are generally sized and designed for the specific inputs they receive, which makes their contaminant removal effectiveness high in the first years after construction. However, unexpected contaminant inputs may not be able to be removed or reduced when the original design tends to be limited to specific contaminants. Secondly, because constructed stormwater wetlands tend to fill up with sediment over time and they are seldom maintained in the long term, their effectiveness often dramatically decreases over time. Natural urban wetlands tend to be larger in size and have the vegetation and structural capacity to capture a range of contaminants, particularly nutrients and sediment, without the need for intensive and/or continual maintenance.

1.0 Background Information and Purpose

North Carolina Department of Environmental Quality (NC DEQ) 401 wetland and buffer permitting staff have expressed the need for scientific data to provide guidance to state regulators as they review permit applications. Of primary concern was finding ways to improve water quality in urban and suburban areas when sufficient land is not available for the stream or wetland mitigation that may be required by impacts. Along with the state of North Carolina, the US Environmental Protection Agency (US EPA) has also been concerned about water quality in urban settings, where the majority of stream and wetland impacts occur but the minority of mitigation occurs (Hill et al. 2012). Currently NC DEQ staff have no mechanism for dealing with impact mitigation in forms other than stream or wetland restoration, preservation, or enhancement, but they have been asked by the regulated community to consider retrofitted stormwater structures or new stormwater wetlands as potential mechanisms for uplifting water quality in urban settings, to offset permitted losses.

While some documentation exists in the scientific literature on pre- and post-installation performance of stormwater management structures, few studies exist on natural wetlands and their roles within a flowing surface system from a water quality perspective. Many of the studies on water quality in natural wetlands have not examined wetlands in urban settings (Lee et al. 1975; Brown 1985; Savage et al. 2015). A study by the NC Division of Water Resources (NC DWR; formerly NC Division of Water Quality) investigated the impact of stormwater influent on the overall health of several rural, semi-rural, and urban NC wetlands but not the possible effluent connections or contributions to local watersheds (Schwartzman et al. 2004). Stormwater investigations have dealt with a wide range of watershed characteristics including nutrient and metals loading in stream beds, nutrient uptake in the hyporheic zone, overland stormwater flow patterns, dissolved sediment loads in streams, bridge crossing runoff into riparian systems, and first flush phenomenon in sewers and stream mainstems (Shamseldin and Fassman 2011; McMillan et al. 2013; Hunt et al. 2015). Other studies have investigated the performance of constructed systems such as bioretention cells (Trowsdale and Simcock 2011) or how to successfully install such systems within budget (Claytor 1996).

Stormwater (and other influent) pollutant types and concentrations reaching wetlands depend on land use, soil properties, overlying vegetation, and geology in a watershed of concern, in addition to global factors such as rainfall chemistry. It is theoretically possible that any material produced in a watershed can be found in the down-gradient surface waters at some point. Surface water concentration ranges reported in the literature generally exist on a spectrum from undeveloped and parkland areas (least impacted) to low-density residential and/or agricultural lands to high-density residential, commercial, industrial, and agricultural areas (most impacted) (Kadlec and Wallace 2009).

Stormwater constituents may enter a natural floodplain wetland through several hydraulic paths including precipitation, overland flow, tributary flow, through-flow, groundwater discharge and/or mixing, and overbanking. Stormwater input from short, intense storms in urban areas (with high amounts of impervious surface and highly compacted soils) would not be expected to have a significant below-ground component during the events; stormwater input into urban wetlands is largely in the form of surface water from inlets and overbanking of adjacent streams.

1.1. Study Design and Goals

This study centered on natural wetlands along a 17.9-mile creek within an almost entirely urbanized watershed, in Raleigh, North Carolina. From 2010 to 2020, Raleigh, NC was the third fastest growing US metro area, and nearby Apex, NC was the fastest growing city in the US from 2018 to 2020 (Herdean 2020). Wake County—home to Raleigh, Apex, and several other towns—experienced a 25.4% population growth from 2010 to 2020 and currently has over 1.1 million residents (Wake County Census and Demographics 2020). Of all NC counties, Wake County has had the highest number of approved 401 permit applications for both stream impacts (1,302 permits; 2007 to 2017) and wetland impacts (910 permits; 2006 to 2016), resulting in 159,266 linear feet of stream impacts and 147.59 acres of wetland impacts over 10 years (NC DEQ 2017). These impacts potentially threaten water quality within Wake County and/or other communities downstream. Urban development increases the severity of downstream erosion and flooding from storm event discharges. In addition, poorly managed stormwater discharges from historical development can potentially harm down-gradient water resource quality. These factors can put the water quality of a metro area such as Raleigh at higher risk of degradation compared to less-developed areas.

The primary goal of this project was to provide baseline data on water quality for water coming into natural urban wetlands and water leaving the wetlands into an urban creek. A secondary goal was to compare the data collected with results reported in the literature for constructed stormwater wetlands. During this project, water quality data were gathered from inlets, centers, and outlets of four natural wetlands ("intensive study wetlands") along Walnut Creek in Raleigh, NC. This was done over a 24-month period during baseflow and after storm events, to understand pollutant concentrations in baseflow and stormwater. Additional information on habitat quality (rapid assessments), sediment movement, hydrodynamics, stratigraphy, and rainfall were also recorded.

2.0 Methods

2.1. Project Area

This project was focused on wetlands along the highly urbanized Walnut Creek, a major creek in Wake County, central North Carolina (Raleigh metro area). Walnut Creek is a tributary to the Neuse River, which itself flows along the eastern edge of the Raleigh metro area and ultimately out to the Pamlico Sound (Figure 1). Several natural wetlands occur along Walnut Creek, all of which discharge surface water into the Creek at least intermittently through small tributary channels or rills that flow through the majority of the wetland areas.

Walnut Creek headwaters are located on the western end of the creek in the town of Cary, NC. Two amenity lakes are located upstream of the chosen study wetlands: the 150-acre Lake Johnson and the 75-acre Lake Raleigh. Water levels in both lakes are controlled by dams and outfall structures. Because emphasis in this urban area has historically been on moving stormwater to streams rapidly (typical of urban areas), the hydrologic cycle in the Walnut Creek watershed has been heavily altered (Walnut Creek Watershed Action Plan 2021). Throughout its length, the creek itself is often quite incised due to flashy heavy flow, with bed elevations at 8 to 10 feet below bank. However, because water level fluctuations in Walnut Creek after substantial storms can be in excess of 15 feet, water regularly overbanks into adjacent low-lying areas, including wetlands (Figure 2 and Figure 3). During large storms, wetlands along the creek receive stormwater from their inlets as well as through overbank flow from Walnut Creek.

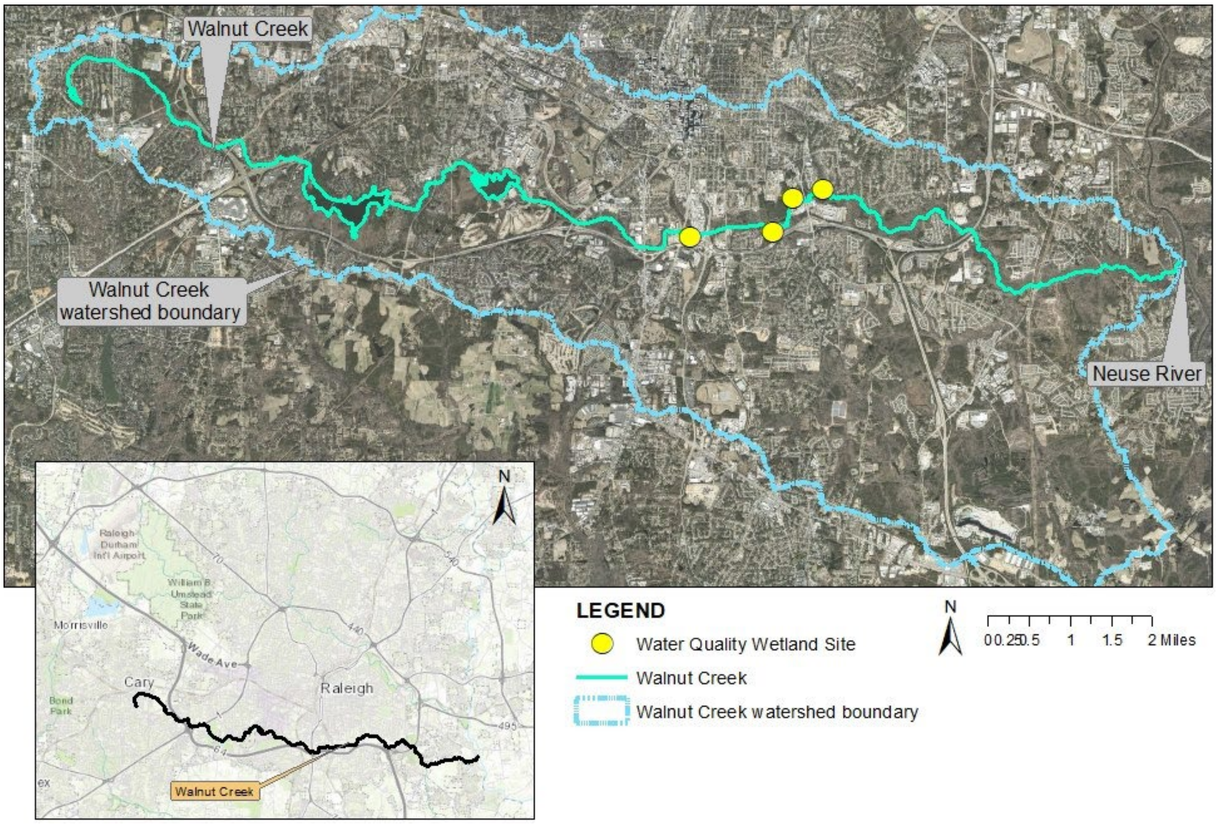


Figure 1. Walnut Creek watershed in Raleigh, NC with water quality wetland study site locations shown.

Figure 2. Walnut Creek near South State Street at baseflow (top) and storm conditions, after 0.52 inch of rain on 12-16-20 (bottom).

Figure 3. Walnut Creek at Rock Quarry Road at baseflow (top) and flood stage (overflowing banks after 3.32 inches of rain) (Feb 7-8, 2020) (bottom).

2.2. Site Selection and Locations

2.2.1. Selection Criteria

Four natural wetland sites with perennially flowing surface water were selected within the City of Raleigh limits. These riparian wetlands each had a single discernable stream or streamlet inlet (permanently flowing) that was the only visible source of surface water into the main portion of each wetland. In each wetland, a single permanently flowing wetland outlet debouched directly into Walnut Creek. Wetland sites were chosen to have a discernable underlying aquitard layer starting at four to five feet below ground surface consisting of a dry, tight gley clay of presumed autochthonous origin. The aquitard was desired to eliminate or at least minimize mixing of true groundwater with surface waters. The original target sample size was 11 wetlands; however, the variable nature of the soil properties across these urban wetlands made the site selection process extremely painstaking and many of the wetlands had multiple inlets and/or outlets and so were unsuitable for this study.

2.2.2. Detailed Site Descriptions

The four natural wetlands selected for intensive study, located in the floodplain of Walnut Creek, were given the names of Apollo Heights (APH), Hammond Road (HR), Women's Prison (WP), and Zhang Property (ZH) (Figure 4). The wetland boundary of each wetland was delineated by a trained wetland delineator using the USACE 1987 Manual: Eastern Mountain and Piedmont Supplement methodology (USACE 1987). Catchment watersheds for the wetland sites were determined using topographic mapping, aerial photo analysis, and Google Earth elevation model data (Table 1). Analysis of historical aerial photographs indicated that these wetland sites were part of a larger network of riparian forested wetlands that were left at the edges of former farm fields along Walnut Creek due to their unsuitability for cropping. These relict wetlands have gone through major transformations including the die-off of preexisting tree stands with replacement by ponds, marsh, or hydric trees such as black willow (*Salix nigra*) and/or with hydric-tolerant invasives such as Chinese privet (*Ligustrum sinense*) as the watersheds experienced increased runoff following the replacement of farms with urban development.

USGS stream level and precipitation gages were located in the project area at the South State Street bridge adjacent to the west edge of ZH and at the South Wilmington bridge over Walnut Creek immediately to the west of HR (Figure 4). Data from these gages were available online in real-time during the project period.

Table 1. Wetland site and catchment characteristics.

Site Name	APH	HR	WP	ZH
Wetland type	Riparian hardwood swamp forest	Willow swamp/ freshwater marsh	Freshwater marsh	Freshwater marsh
Wetland area (acres / hectares)	3.44 / 1.39	5.79 / 2.34	16.01 / 6.48	19.03 / 7.70
Catchment size (acres / sq. miles)	384 / 0.60	44.8 / 0.07	19.2 / 0.03	153.6 / 0.24
Impervious surface %; developed % (from StreamStats [USGS 2016] / National Land Cover Database 2011 [Homer et al. 2015])	22%; 98%	56%; 98%	12%; 45%	17%; 90%

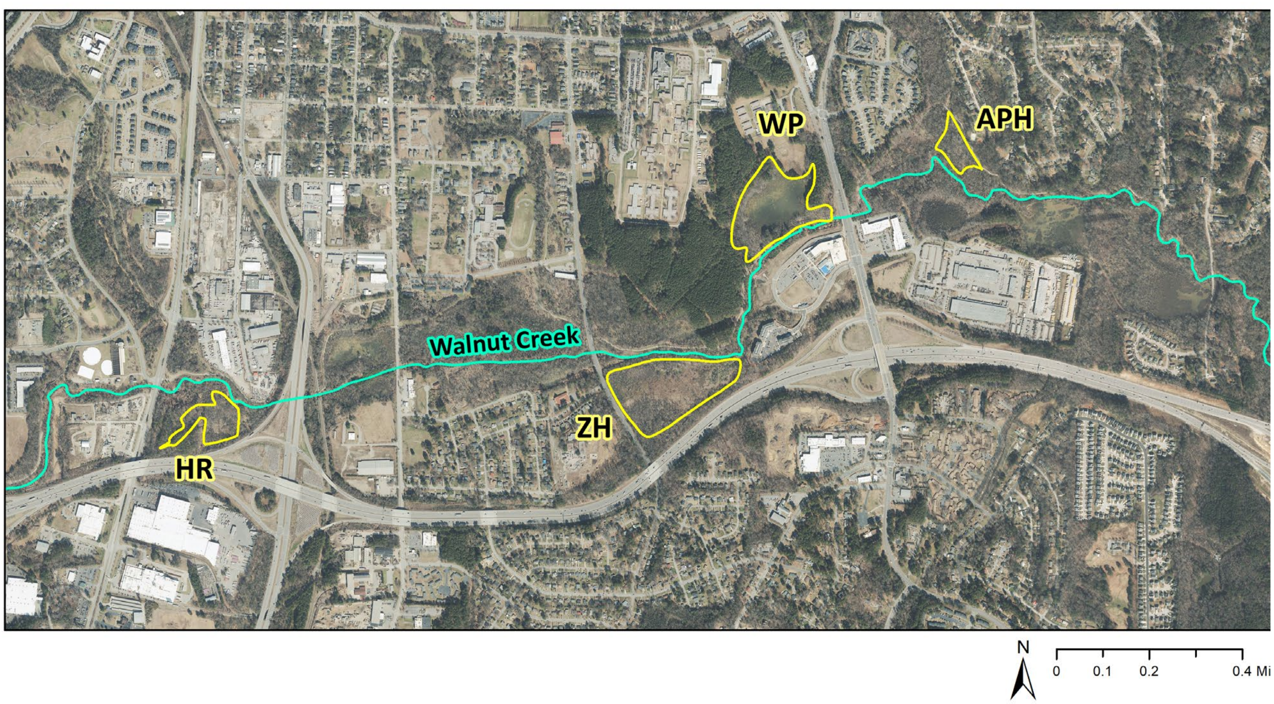


Figure 4. Locations and field-delineated boundaries of intensively sampled urban wetland sites along Walnut Creek in southern Raleigh, North Carolina.

2.2.2.1. Apollo Heights site (APH)

The Apollo Heights (APH) wetland site was the easternmost intensive study site. It had the largest catchment of any of these study sites, at 384 acres or 0.6 sq. miles, and that catchment was dominated by older residential neighborhoods (Figure 5). The wetland itself was approximately 3.44 acres in size (Figure 6). Most of this wetland was forested, with a beaver dam impounding water in the center and resulting in a fair amount of tree mortality.

The APH wetland was dominated by Chinese privet, boxelder (*Acer negundo*), red maple (*Acer rubrum*), and green ash (*Fraxinus pennsylvanica*). The understory vegetation, where present, was dominated by Japanese stiltgrass (*Microstegium vimineum*), green ash saplings, young Chinese privet, and a variety of herbaceous species such as Canadian clearweed (*Pilea pumila*), wart-removing-herb (*Murdannia keisak*), false nettle (*Boehmeria cylindrica*), eastern poison ivy (*Toxicodendron radicans*), arrowleaf tearthumb (*Persicaria sagittata*), and dayflower (*Commelina virginica*). It had a culverted inlet tributary that flowed beneath the Walnut Creek greenway pavement surface (Figure 7). The wetland center contained a patchy distribution of living and dead trees with herbaceous species beneath (Figure 8). The outlet was a slightly incised channel armored with herbs, shrubs, and trees (Figure 9). Channel material consisted mostly of silt and sand.

Figure 5. Catchment map - APH site. Hillshade layer source: Wake County GIS.

Figure 6. Site detail showing sampling points and wetland boundary - APH site.

Figure 7. APH site inlet, baseflow (top) and storm flow after 0.35 inch of rain on 12-16-20 (bottom).

Figure 8. APH site center.

Figure 9. APH site outlet, baseflow (top) and storm flow after 0.42 inch of rain on 12-16-20 (bottom).

2.2.2.2. Hammond Road site (HR)

The Hammond Road (HR) site was located just west of the on-ramp to US Highway 40 from Hammond Road. The catchment for this wetland was small (44.8 acres or 0.070 sq. miles) and dominated by impervious surface at a shipping company hub to the south and by a large gravel parking area west of the wetland (Figure 10). The wetland was 5.79 acres in size and lay to the south of Walnut Creek (Figure 11). The inlet into this site constituted a wide, one-meter-deep channel leading east from two culverts: one under Wilmington Street and one under US Highway 40 from nearby development (shipping company hub) to the south, adjacent to US Highway 40 (Figure 12). Along the inlet channel, there was only slight evidence of overbanking after storms, probably because of the small catchment size.

Scattered wetland trees were mainly comprised of black willow and red maple, with a marsh component dominated by broadleaf cattail (*Typha latifolia*), rice cutgrass (*Leersia oryzoides*), jewelweed (*Impatiens capensis*), and Japanese stiltgrass. Other species present included box elder, black elderberry (*Sambucus nigra*), Chinese privet, American sycamore (*Platanus occidentalis*), and trumpet creeper (*Campsis radicans*). The center of the wetland was characterized by fewer trees (Figure 13).

At its eastern end, the channel became shallow and flooded into the wetland, flowed through it, and eventually turned into a small stream as an outlet (Figure 14). Outlet channel material was mostly silt.

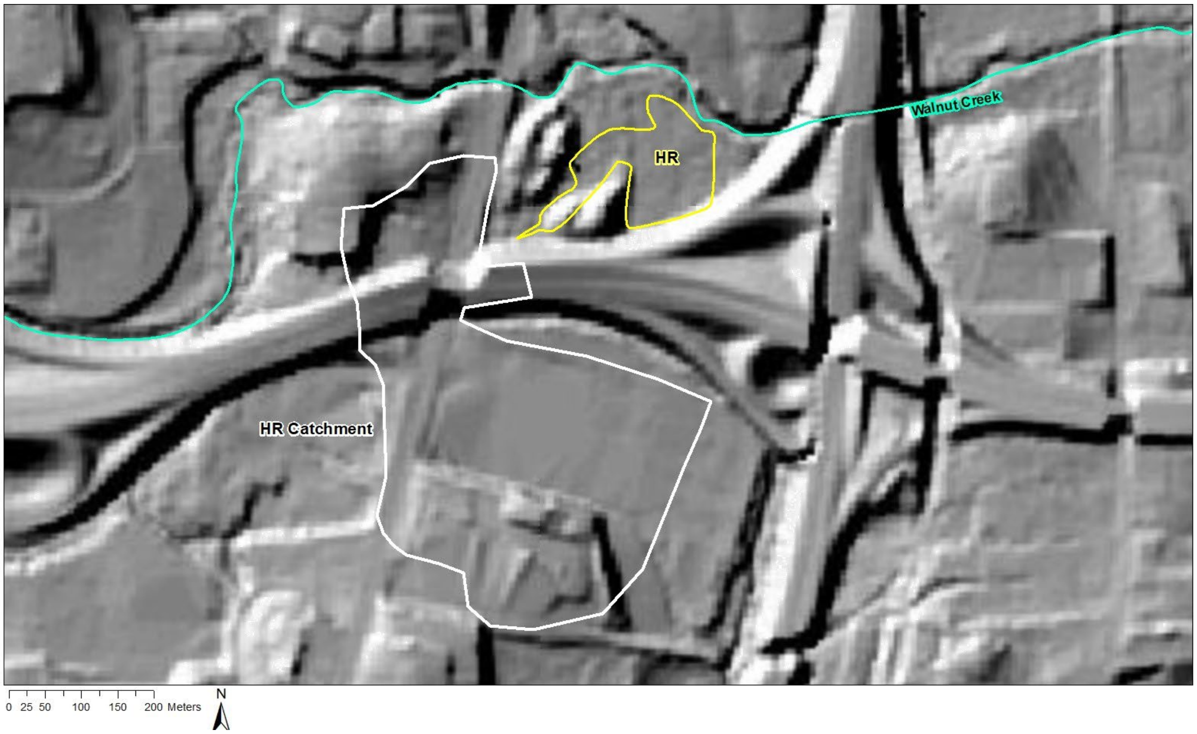


Figure 10. Catchment map - HR site. Hillshade layer source: Wake County GIS.

Figure 11. Site detail showing sampling points and wetland boundary - HR site.

Figure 12. HR wetland site inlet, facing downstream to wetland, baseflow (top) and storm flow, still within channel, after 0.65 inch of rain on 12-16-20 (bottom).

Figure 13. HR wetland site center.

Figure 14. HR wetland site outlet, facing downstream, base flow (top) and storm flow, still within channel after 0.65 inches of rain, 12-16-20 (bottom).

2.2.2.3. Women's Prison site (WP)

The Women's Prison (WP) wetland site was roughly circular in shape, with an herbaceous marsh center and forested hardwood wetland fringe. The catchment for this wetland (19.2 acres or 0.03 sq. miles) was not much larger than the wetland itself, which was 16.01 acres (Figure 15). The catchment area was mostly forested with a small amount of developed area. The ponded water in the marsh was created by well-maintained beaver damming of a city ditch that drained into Walnut Creek under the greenway. The inlet was located on the north side of the wetland and was a shallow, sandy runoff product from the North Carolina Correctional Institution for Women campus (Figure 16 and Figure 17).

The herbaceous marsh center of the wetland was dominated by green arrow arum (*Peltandra virginica*), common buttonbush (*Cephalanthus occidentalis*), lizard's tail (*Saururus cernuus*), and Japanese stiltgrass. The forested fringe was dominated by red maple, black willow, green ash, water tupelo (*Nyssa aquatica*), and Chinese privet, with occasional loblolly pine (*Pinus taeda*) and eastern cottonwood (*Populus deltoides*) (Figure 18). There was evidence that some water tupelo and common buttonbush trees were planted many years earlier (i.e., remnant protective plastic tubing around trunks).

The outlet used for half of the study duration consisted of a small rivulet that debouched directly into Walnut Creek through the Creek's northern levee; when this outlet was destroyed by City greenway maintenance activities in late 2019, a nearby beaver-dammed outlet (about 50 feet away) began flowing and sampling was performed at that location during the last five sampling events (Figure 19). The outlet bed and bank material consisted of clay and silt.

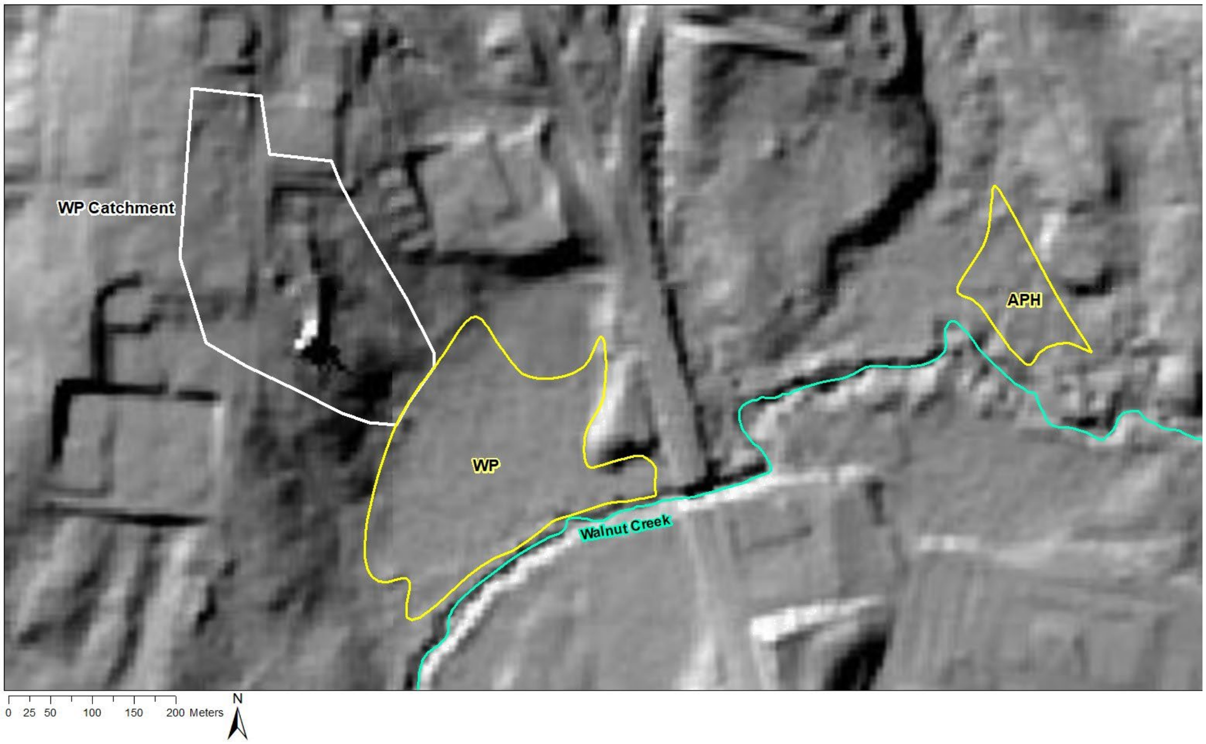


Figure 15. Catchment map - WP site. Hillshade layer source: Wake County GIS.

Figure 16. Site detail showing sampling points and wetland boundary - WP site.

Figure 17. WP site inlet, at baseflow facing downstream to wetland (top) and at storm flow facing upstream after 0.16 inch of rain on 10-16-2019 (bottom).

Figure 18. WP wetland site center (top) and forested fringe (bottom).

Figure 19. WP wetland site outlet, baseflow facing downstream (top) and storm flow facing upstream after 0.47 inches of rain on 12-16-20 (bottom).

2.2.2.4. Zhang Property site (ZH)

The Zhang Property (ZH) wetland site was a relatively large wetland with a moderate catchment area of 153.6 acres or 0.24 sq. miles (Figure 20). The catchment area was primarily residential.

The ZH wetland site was an 18.8 acre/7.6 ha wetland, mostly herbaceous/shrub marsh with a forested hardwood fringe (Figure 21). The inlet into this wetland came from the south in the form of a flowing stream culverted under US Highway 40, joined by intermittently flowing input from a culvert under South State Street north of US Highway 40 (Figure 22). The inlet channel consisted of a mix of silt and coarse sand and was located under the hardwood fringe.

The center sampling area was within the herbaceous marsh component, which was dominated by broadleaf cattail, rice cutgrass, Japanese stiltgrass, arrowleaf tearthumb, Japanese hops (*Humulus japonicus*), common buttonbush, and swamp rose mallow (*Hibiscus moscheutos*) (Figure 23). The forested area was dominated by red maple, American elm (*Ulmus americana*), sweetgum (*Liquidambar styraciflua*), Chinese privet, and the occasional northern spicebush (*Lindera benzoin*).

Outlet water flowed through a streamlet which was highly incised through clayey marsh sediments that arose at the creek-ward edge of the central wetland area (Figure 24).

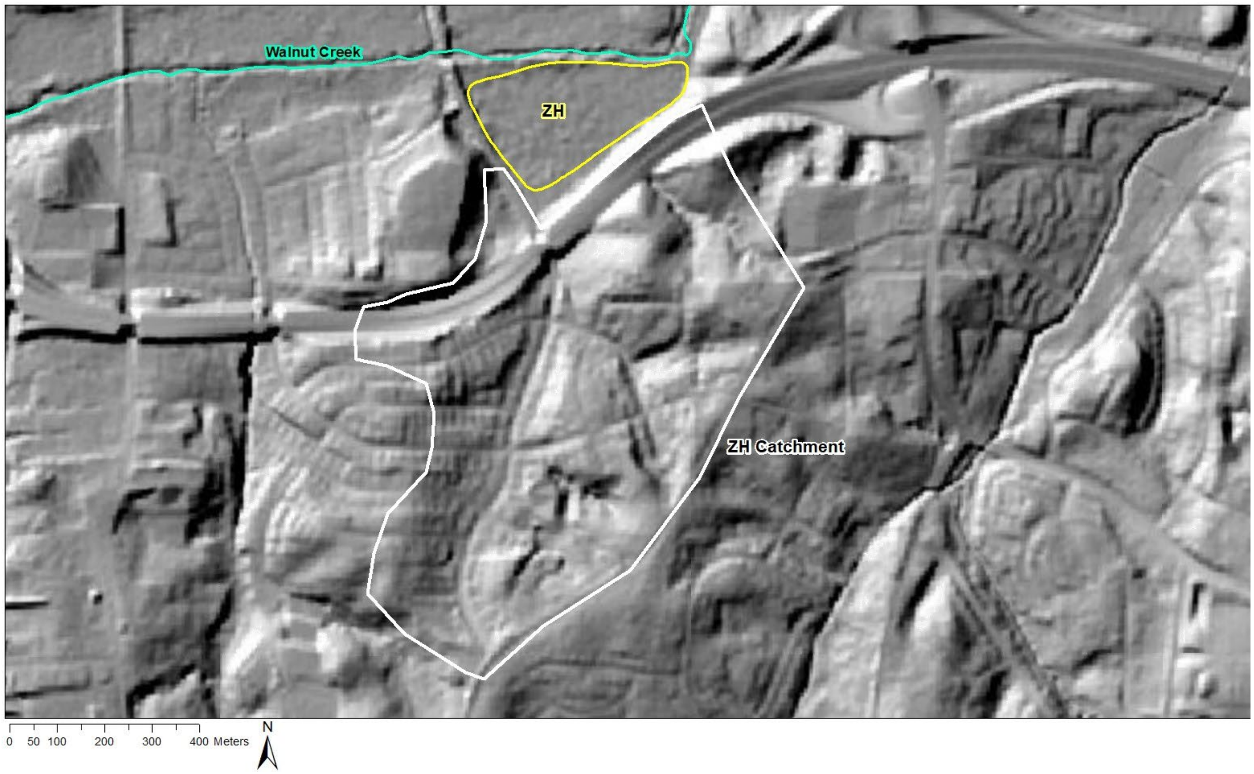


Figure 20. Catchment map - ZH site. Hillshade layer source: Wake County GIS.

Figure 21. Site detail showing sampling points and wetland boundary - ZH site.

Figure 22. ZH inlet, facing downstream to wetland, baseflow (top) and storm flow after 0.6 inch of rain on 12-16-20 (bottom). Pipe housed the water level sensor.

Figure 23. ZH center.

Figure 24. ZH outlet, facing downstream, base flow (top) and storm flow, overflowing channel after 0.65 inch of rain on 12-16-20 (bottom).

2.3. Water Quality Sampling

It was determined after site selection and extensive consultations with local practitioners that the project wetlands were not suitable for investigation using autosampler stations (e.g., Teledyne ISCO sequential samplers) as originally desired. Automatic water samplers are usually placed at culverts and bridges, where the channel is defined and unchangeable. The natural inlets and outlets of the intensive study wetlands did not conform to this requirement, and building a flume in the streams was out of the question, given their small sizes, their changeable and erodible channels, flooding depth up to 6 feet above bank, and the quantity of sediment sometimes deposited by flooding events. Instead, at each of the intensive study wetlands, water quality grab samples were collected from the inlet and outlet positions of each tributary stream flowing through each wetland site and in a central position within the delineated border of the wetland. The central position was approximately midpoint of the flow pattern through the wetland and was sampled to complete the characterization of flow-through water in total. Baseflow water quality grab samples were obtained from the intensive study wetlands on nine different dates from February 2019 through August 2022 (Table 2). Lack of water in a given inlet or outlet occasionally prevented sampling at particular locations. Baseflow samples included the four intensive sites, plus eight supplemental sites (sampled two times each; see Section 2.3.1).

Storm events were targeted for sampling when they were preceded by droughts of at least seven days in an attempt to capture measurable accumulations of targeted constituents in the subwatersheds. Storm samples were collected at the intensive wetland sites on five different dates over an 18-month sampling period (June 2019 through December 2020) (Table 2). During two of these five dates, storm samples were collected from the wetland centers, when it was safe to do so.

For comparison to Walnut Creek water quality, grab samples were obtained from Walnut Creek at the ZH site for baseflow (four times) and storm flows (two times) (Table 2; Figure 2; Figure 25).

Grab samples were analyzed by NC DWR Water Sciences Section (WSS) Central Laboratory for several constituents of concern including oil and grease, total suspended solids (TSS), nutrients (nitrate + nitrite, total Kjeldahl nitrogen [TKN], and total phosphorus [TP]), and some heavy metals (copper, lead, zinc). Analysis of initial samples for dissolved metals yielded approximately the same concentrations as total metals; comparison of initial dissolved and total metals sample results showed no statistical difference, so only total metals were analyzed in subsequent samples.

Bottles and preservatives were handled using nitrile gloves. Sample bottles were prepared prior to field administration according to Section 6.0 of the Quality Assurance Manual for the NC DWR WSS Chemistry Laboratories (NC DEQ DWR 2015). Water samples were chilled to 6° Celsius on site and transported to the NC DWR WSS Central Laboratory.

Concurrent with grab sample collection, a Xylem Pro Plus water quality meter was used to measure field parameters, specifically pH, specific conductivity, water temperature, and dissolved oxygen. Later in the project timeline, an Aqua TROLL 400 multi-parameter water quality sonde with handheld tablet was used to measure the same parameters. Regular calibration of these meters followed manufacturer instructions and NC DWR Standard Operating Procedures. Field parameters were measured with the meters at the time of sample collection at the inlet, center, and outlet of each wetland.

Table 2. Water quality sampling dates for baseflow and storm events; all sites were sampled during each sampling date or date range. GS = growing season; NGS = non-growing season.
*Water samples were taken on this date from wetland inlets only, in an effort to capture first flush.
+Dates of grab samples from Walnut Creek.

Baseflow Sampling	Storm Event Sampling
2/1/19 (NGS)+	6/10/19 (GS)+
4/24/19 (GS)+	8/14/19 (GS)
7/10/19 (GS)	10/16/19 (NGS)*
12/4/19 (NGS)	9/17/20 (GS)
2/4/20 (NGS)	12/16/20 (NGS)+
3/11/20 (NGS)+	
6/25/20 (GS)+	
1/12/22 – 2/2/22 (NGS)	
8/16/22 – 8/30/22 (GS)	

2.3.1. Supplemental Water Quality Sampling

To put results from the four intensive study sites into a broader context, eight supplemental wetland sites were identified along Walnut Creek from which surface water could be collected as grab samples from an inlet, center, and outlet (Figure 25). These sites were not conducive to characterization as intensive study wetlands, usually because they had multiple inlets or outlets, but they were all situated along the same stretch of Walnut Creek and shared the same urbanized watershed context. Data from the supplemental sites were shown on graphs for context, but not included with data from intensive sites for analysis.

Baseflow grab water samples were obtained from these eight supplemental wetlands, at an inlet, center location, and outlet once during January/February 2022 (non-growing season) and once during August 2022 (growing season). Field parameters (i.e., pH, specific conductivity, water temperature, and dissolved oxygen) were also measured on these dates.

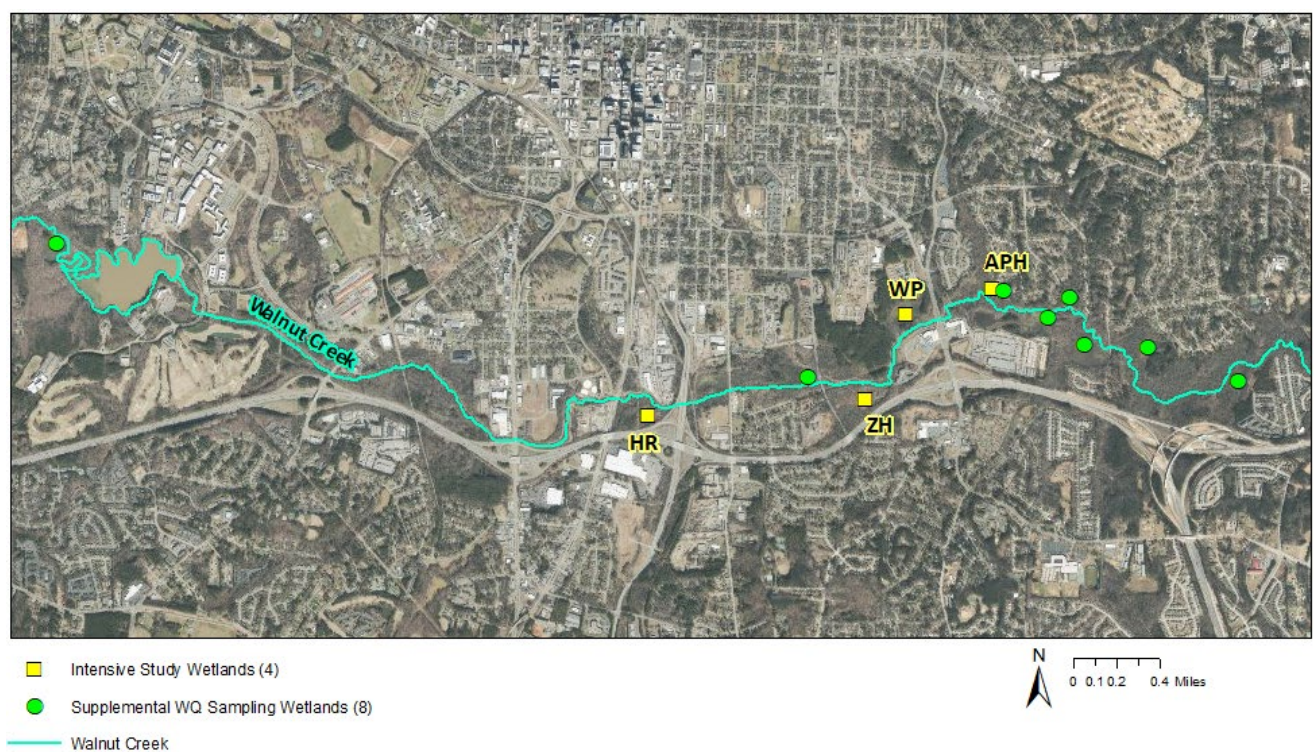


Figure 25. Locations of supplemental water sampling wetland sites in relation to the intensively sampled wetlands along Walnut Creek in Raleigh, North Carolina.

2.4. Water Level Monitoring

Water levels were measured at water sampling locations and the interiors of the wetted wetland areas using non-vented, pressure activated, automated water level dataloggers (Onset HOBO U20L Series). Monitoring water levels provided data on wetland hydrodynamics and the timing of water sampling visits in relation to the anticipated first flushing of contaminants before overbanking of the tributaries had occurred.

HOBO units were installed in wells in the center of the wetlands to track changes in surface water levels and/or subsurface water retention in the wetlands themselves. Groundwater levels were monitored with deep (5-8 feet) wells in two of the sites (HR and ZH) to track changes in groundwater levels below these wetlands as Walnut Creek received and discharged stormwater. Surface water levels in two beaver-impounded sites (WP and APH) were monitored with stilling wells emplaced within the impoundment footprint and outfitted with HOBO units.

HOBO units installed in wide tributary streams needed to be stable and level to gather correct data and were at risk of being washed away with storms, so a protective apparatus was developed to house the units. The apparatus constituted a large PVC pipe with a protective box housing the HOBO unit anchored inside (Figure 26). A 0.25-inch metal screen was placed at the upstream end of the pipe to keep out large debris. HOBO units in narrower streams were installed using metal stakes and cross braces which connected to a standing PVC pipe with the HOBO unit in it (Figure 27). One additional HOBO unit was mounted high in a tree near the center of the full project reach to calculate the atmospheric barometric pressure for calibration purposes. Readings were taken every 30 minutes for all HOBO units. Continuous water level data were obtained from July 25, 2019 through July 7, 2020 for each wetland. Readings were checked in the field against hand measurements of water depth and defective equipment was replaced as needed. Resulting hydrographs are included in Appendix D. Gaps in the data for WP center from December 2019 to January 2020 and at the end of the timeframe for center and inlet at HR were caused by equipment malfunction.

Figure 26. Apparatus for housing HOBO water level datalogger in flowing streams to prevent HOBO units from getting washed away; protective box was placed inside large pipe, which was staked in and had a screen at the upstream end to keep out debris.

Figure 27. Apparatus for housing HOBO water level datalogger in small streams; metal stakes and cross braces connected to a standing PVC pipe with the HOBO unit suspended inside.

2.5. Precipitation and Stage Characterizations

Precipitation events (amounts and intensity) and USGS stream gage hydrographs for Walnut Creek were monitored using data from the USGS stream gages located at the South State Street and Wilmington Street crossings of Walnut Creek. These data were used to plan water sampling attempts in the study wetlands and understand the ongoing hydrodynamic relationships between the sites and Walnut Creek mainstem.

2.6. Sediment Sampling

Sediments were collected and analyzed to understand whether stream and wetland soils were contributing contaminants to the water column and/or whether flowing surface waters were depositing long-term accumulations of contaminants at the water sample stations. After all water sampling was completed, surficial sediments/soils were collected and analyzed from each of the water quality sample collection points, including those from the supplemental sites. Sediments were sampled using a Hori Hori weeding knife into the top six inches of material. Each sample was placed into a plastic bag and stapled according to EPA's 2016 National Wetland Condition Assessment soil collection protocol (US EPA 2016a). The sampling utensils were carefully decontaminated by rinsing thoroughly with tap water between each sampling location. The samples were chilled in the NC DEQ WSS Central Laboratory walk-in cooler upon return from the field.

Sediments were analyzed by Brookside Laboratories, Inc. (Bremen, OH) for most of the same parameters as the water analysis: metals (copper, lead, zinc), nutrients (available nitrogen, total phosphorus), organic matter, and pH, as well as bulk density and several additional parameters (e.g., magnesium, iron, calcium). Analysis for petroleum products was not obtained because oil and grease were very rarely detected in the surface water samples.

2.7. Sediment Movement Assessment

A supplementary investigation of sediment movement was undertaken to gain insight into how the study wetlands could be dealing with incoming sediment, including allowing sediment to pass into Walnut Creek mainstem. This insight was deemed valuable because, though not necessarily a regulated contaminant per se, excessive sediment influx in receiving streams can cause major degradation during and after storm events (Mallin et al. 2009). Estimates of tributary erosion, catchment sediment contributions (tributary sedimentation input), overbank sediment deposition, and wetland surface accretion were undertaken at the intensive study site best suited for this characterization: the ZH site.

2.7.1. Assessment of Tributary Erosion

To estimate tributary erosion, six cross sections were measured in the perennial ZH stream tributary to Walnut Creek, from the upstream edge of the interior marsh to a downstream section where a Raleigh City stormwater pipe crossed the tributary. These were set up and evaluated on February 19, 2019. The cross sections were remeasured approximately one year later on February 26, 2020 and an annual estimate of material eroded from ZH tributary banks was calculated.

2.7.2. Assessment of Tributary Sedimentation Input

To estimate sedimentation input from the inlet tributary at ZH, a spatial grid of nine stakes was established on February 18, 2019 approximately 200 feet (~61 meters) downstream of the inlet area of the perennial ZH tributary and the grid covered approximately 952 m^2 (1,139 sq yards) of the forested southwestern corner of the wetland. The stakes were marked with a notch at the ground surface at that time. The accretion above notches was monitored on a random basis, with gaps of two to six months, to measure vertical sediment buildup.

2.7.3. Assessment of Overbank Sediment Deposition into Marsh Area

To estimate overbank sediment deposition into the marsh area at ZH, five PVC pipes were established in a transect from a low bank area on the western border of the ZH wetland adjacent to Walnut Creek south toward the central marsh area. The five 2-inch pipes were approximately four feet high, emplaced vertically in the wetland surface, and marked with a reference line near the top to establish a datum from which the wetland surface accretion from overbank depositing could be measured (Figure 28). The distance from the bank to the most interior marker was 300.8 feet (91.7 meters). Wetland surface levels at the pipes were measured randomly several times over the 20 months from July 22, 2020 through March 15, 2022.

Figure 28. Example of PVC pipe used to measure overbank sedimentation at ZH wetland.

2.8. Data Analysis

There are several ways to characterize a wetland's ability to remove contaminants including inlet to outlet concentration drop, inlet to outlet loadings drop, outlet concentration versus areal input loading, rate constant versus areal input loading, and various mass balance models that determine amounts of material over time or over area over time (e.g., Brinson et al. 1981; Vepraskas et al. 2016). Kadlec and Wallace (2009) considered only the first and third conceptualizations "useful". In the literature, the most commonly reported performance characterization is concentration change from inlet to outlet as a percent. Results for this study are reported in terms of concentration differences from inlet to outlet, which were calculated using the following formula:

$$\left(\frac{\textit{outlet concentration - inlet concentration}}{\textit{inlet concentration}}\right) \times \textit{100\%}$$

Data were analyzed using the statistical software packages JMP 12 (SAS® 2015) and PAST version 4.03 (Hammer et al. 2001). Results were analyzed by baseflow vs. storm flow and by growing season (April through September) vs. non-growing season (October through March). Paired comparisons (e.g., inlet vs. outlet, inlet vs. center) were made using the nonparametric Wilcoxon signed-rank test. Results were considered significant at p values less than 0.05. Non-detects were included in the databases at the minimum detection limit of the lab (Table 3).

Table 3. Water quality lab and soils lab minimum detection limits by parameter.

Parameter	Water Quality Lab Minimum Detection Limit	Soils Lab Minimum Detection Limit
Oil and Grease (Hexane Extractable Material; HEM)	10 mg/L	Not analyzed
Total Suspended Solids (TSS)	6.2 mg/L	N/A
Nutrients		
Ammonia (NH_3)	0.02 mg/L	0.5 ppm
Nitrate + Nitrite ($NO_3^- + NO_2^-$)	0.02 mg/L	Not analyzed
Total Kjeldahl Nitrogen (TKN)	0.2 mg/L	Percent Nitrogen: 0.05%
Total Phosphorus (TP)	0.02 mg/L	1 mg/kg
Metals		
Copper (Cu)	2 µg/L	0.2 mg/kg
Lead (Pb)	2 µg/L	5 mg/kg
Zinc (Zn)	10 µg/L	0.4 mg/kg

3.0 Results

3.1. Water Quality Sampling

3.1.1. Storm Characteristics

Water samples were obtained from the four intensive study wetlands at the beginning of five different storms from June 2019 through December 2020, which ranged in total precipitation from 0.46 to 1.95 inches, based on the USGS Walnut Creek precipitation gage at South State Street (Figure 29 through Figure 33). All samples were taken before precipitation ended for the given storm. A concerted effort was made to capture first flush from the storm in October 2019, a storm which had a long preceding drought period of 39 days (the other storm events had preceding droughts of seven to nine days). Analysis showed concentrations of measured parameters were not statistically different between that October 2019 storm sampling and other storm samplings ($p>0.05$; Wilcoxon signed-rank test). An exception was specific conductivity, which was significantly higher in the inlets during the October 2019 storm sampling (mean 241.4 µS/cm ± 25.1 SD) than during other storm samplings (mean 140.1 ± 8.4 µS/cm) ($p<0.05$; Wilcoxon signed-rank test). However, that higher specific conductivity did not coincide with higher concentrations of other measured parameters.

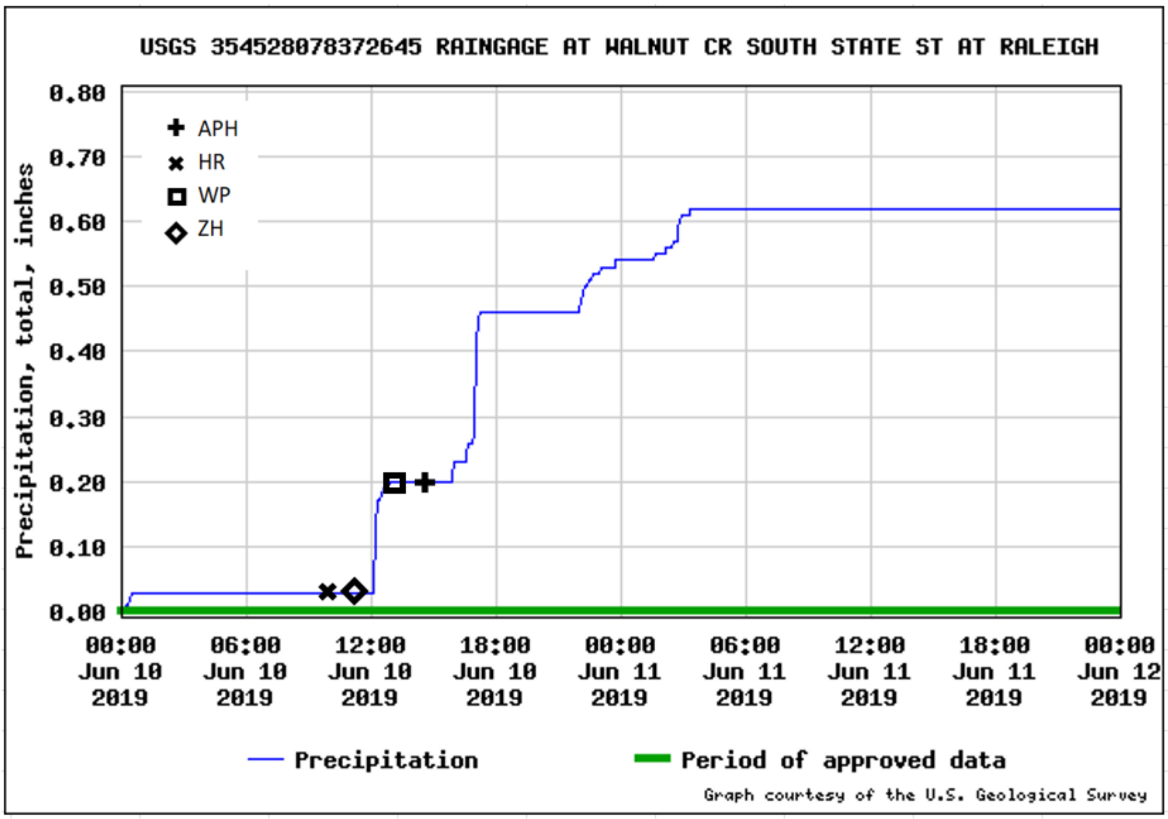


Figure 29. Cumulative precipitation for June 10, 2019 storm; symbols indicate when inlet water sampling occurred at each site. Center and outlet samples were taken at each site after inlet collection and before moving on to the next site.

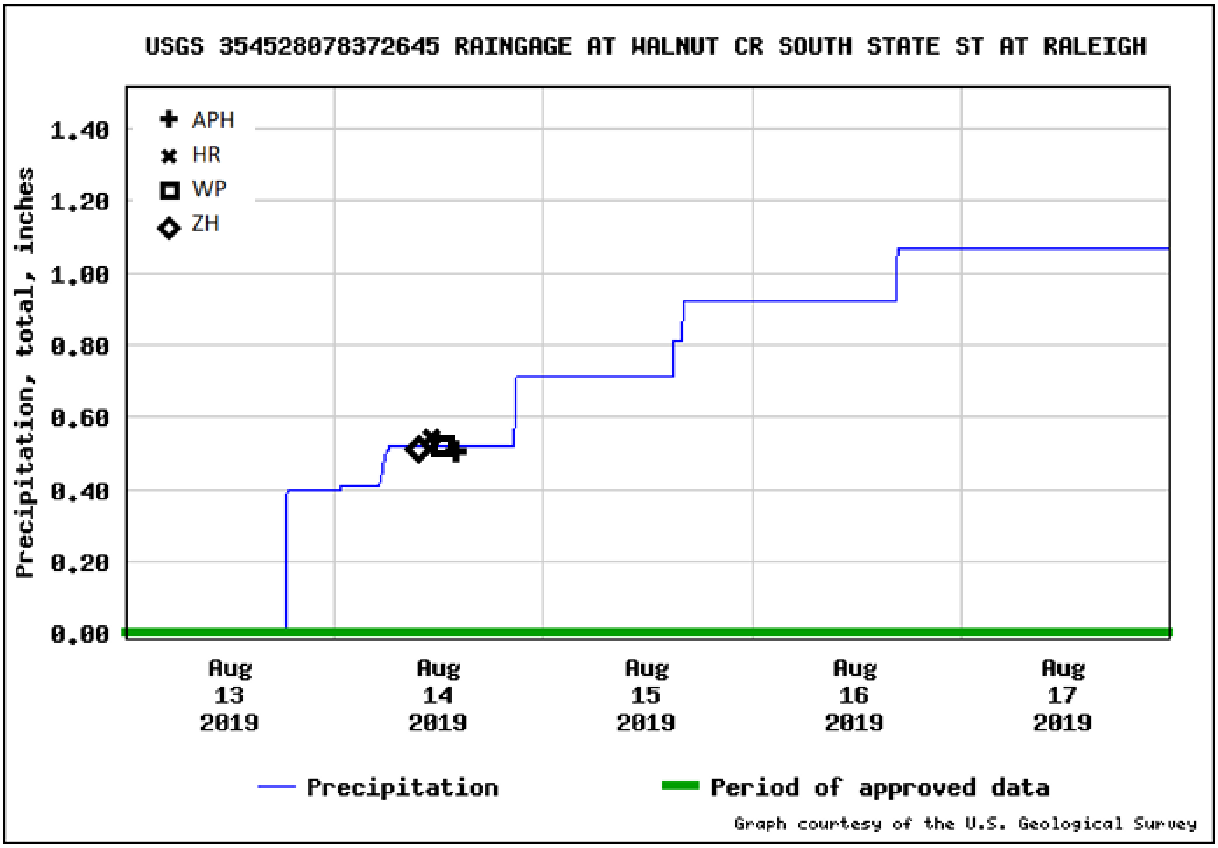


Figure 30. Cumulative precipitation for August 14, 2019 storm; symbols indicate when inlet water sampling occurred at each site. Center and outlet samples were taken at each site after inlet collection and before moving on to the next site.

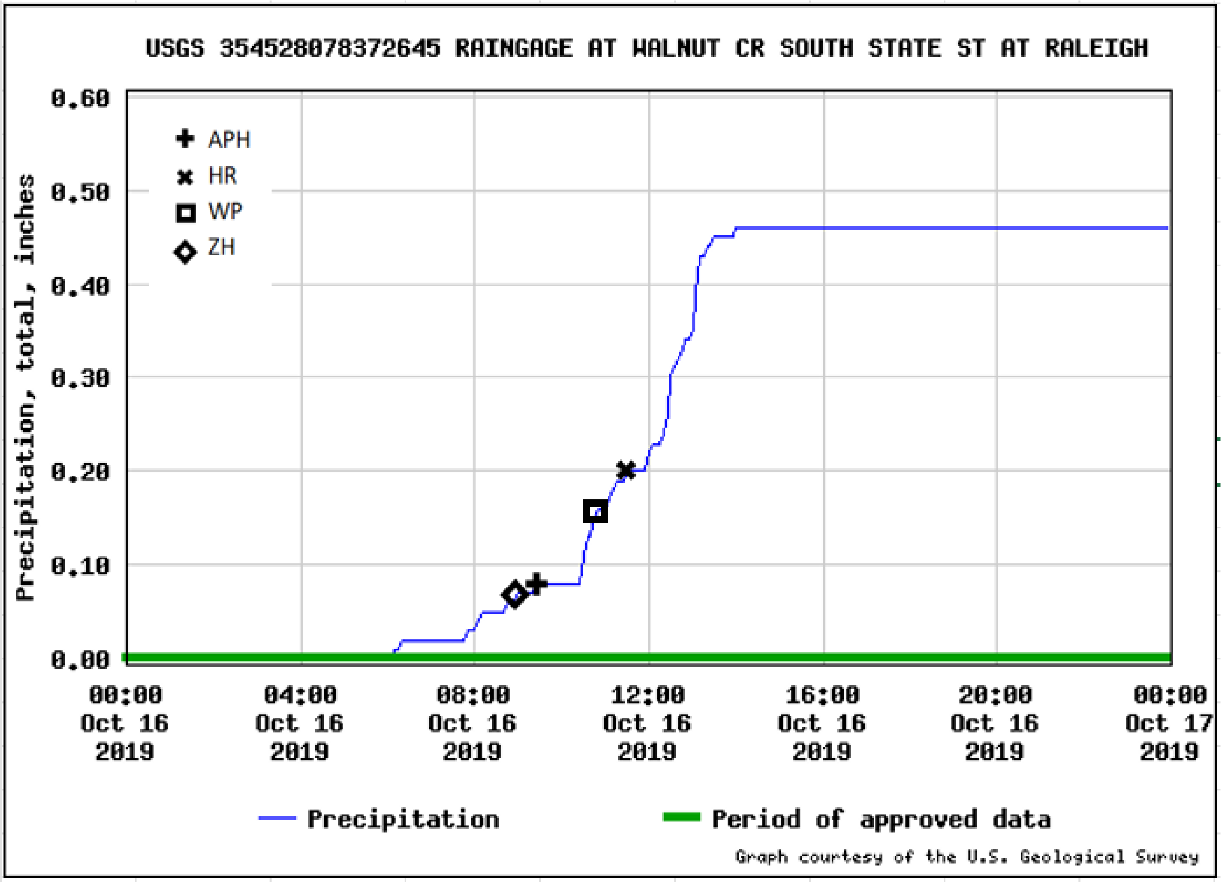


Figure 31. Cumulative precipitation for October 16, 2019 storm; symbols indicate when inlet water sampling occurred at each site. Outlets and centers of wetlands were not sampled in an attempt to capture first flush at all sites on this date.

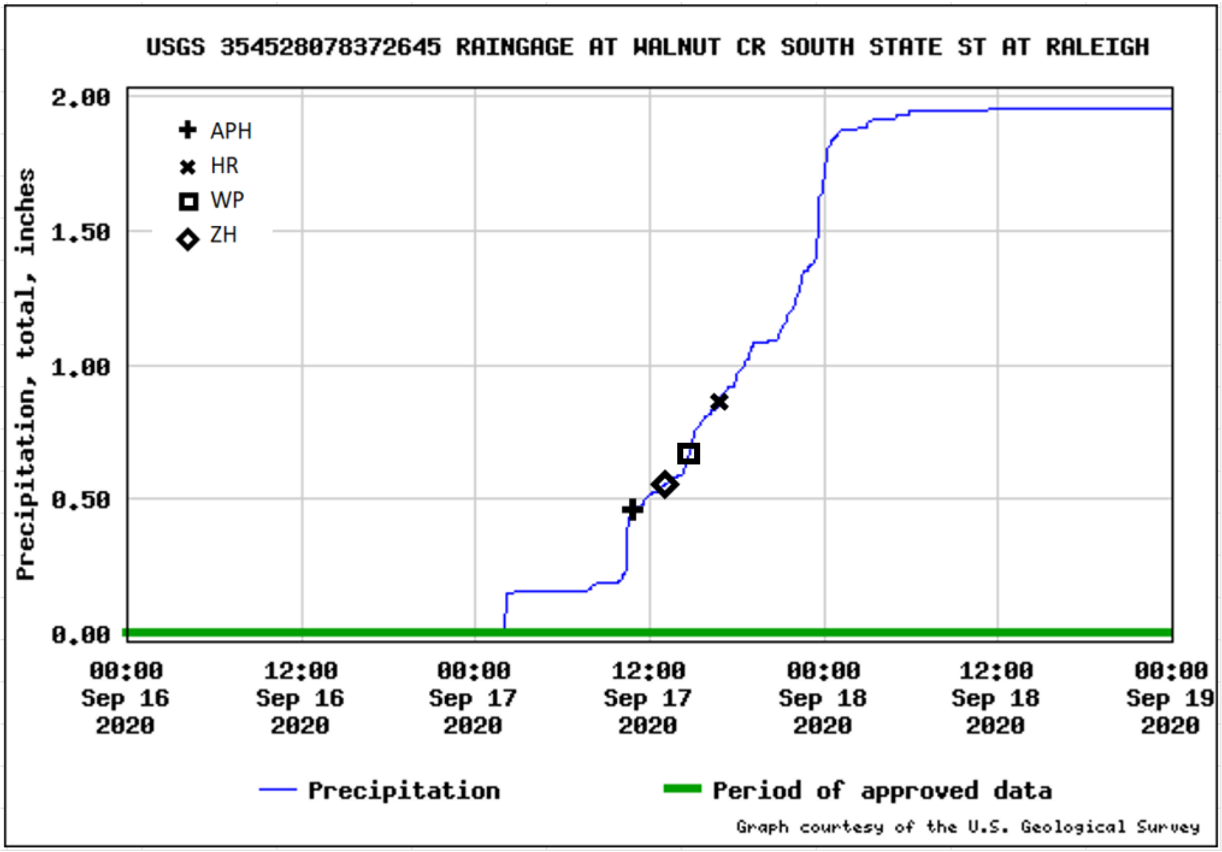


Figure 32. Cumulative precipitation for September 17, 2020 storm; symbols indicate when inlet water sampling occurred at each site. Center and outlet samples were taken at each site after inlet collection and before moving on to the next site.

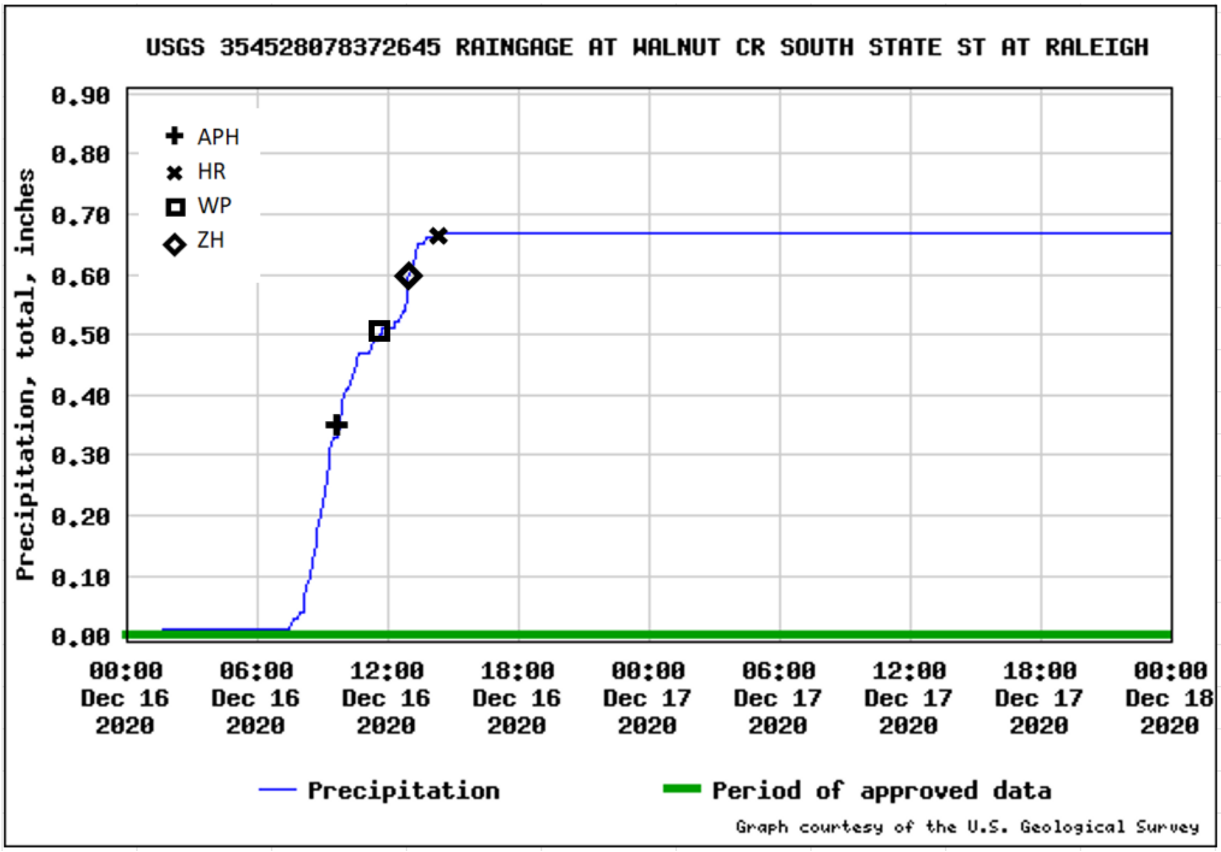


Figure 33. Cumulative precipitation for December 16, 2020 storm; symbols indicate when inlet water sampling occurred at each site. Center and outlet samples were taken at each site after inlet collection and before moving on to the next site.

3.1.2. Summary Statistics for Water Quality Parameters Overall

Total suspended solids concentrations were generally low but sometimes varied widely. Phosphorus and nitrogen species concentrations overall were generally low (<1.0 mg/L) and sometimes not able to be detected (Table 4). Grease and oil were generally undetectable (below the detection limit of 10 mg/L). Copper, lead, and zinc concentrations across all samples tended to be lower than those found in large-scale analyses of urban stormwater across the US (Pitt et al. 2018, Pamuru et al. 2022), with occasional values that were much higher than median (75% quartile values: 6.4 µg/L for copper, 6.9 µg/L for lead, and 28 µg/L for zinc). Concentrations of all three metals covaried with each other (copper and lead: $r^2 = 0.94$; copper and zinc: $r^2 = 0.99$; zinc and lead: $r^2 = 0.93$; all $p<0.001$).

Median pH of water samples overall was slightly acidic (6.8) and ranged from 5.8 to 8.5. Median dissolved oxygen was around 50% but ranged from over 106% to nearly zero. Median specific conductivity was 203.2 µS/cm, with ranges from 35.2 to 881.5 µS/cm. Median water temperature followed seasonal temperatures and varied from 2.9 to 28.3 degrees C.

A single sample, taken from the center of ZH in June 2020, was responsible for the highest values measured for most parameters across all sampling: all three metals (copper: 158x median concentration, lead: 262x median concentration, zinc: 170x median concentration), ammonia (10x median concentration), TKN (126x median concentration), total phosphorus (462x median concentration), grease and oil (4x median concentration), and total suspended solids (10x next highest concentration).

Figure 34 shows the percent reduction or increase in concentrations found for each parameter from the inlet to the outlet across all sampling. Some parameters generally showed percent increases from inlet to outlet (TSS, lead) and some generally showed reductions from inlet to outlet (ammonia, nitrate + nitrite, dissolved oxygen).

Table 4. Medians, means, and ranges for parameters overall and at inlets, centers, and outlets of intensive study sites. ND = non-detect.

	Total Suspended Solids (TSS) (mg/L)	Total Phosphorus (TP) (mg/L)	Ammonia (NH_3) (mg/L)	Nitrate + Nitrite (NO_3^- + NO_2^-) (mg/L)	Total Kjeldahl Nitrogen (TKN) (mg/L)	Copper (Cu) (µg/L)	Lead (Pb) (µg/L)	Zinc (Zn) (µg/L)
OVERALL (N = 145)								
Median	26.0	0.16	0.10	0.10	0.7	3.4	2.7	17
Mean	305.1	0.92	0.15	0.25	1.7	9.7	13.7	52
Max	24,600.0	74.00	1.20	2.20	91.0	540.0	710.0	2,900
Min	6.2/ND	<0.02/ND	<0.02/ND	<0.02/ND	<0.2/ND	<2/ND	<2/ND	<10/ND
INLETS (N = 52)								
Median	15.0	0.14	0.11	0.46	0.6	3.1	2	16
Mean	45.6	0.22	0.14	0.52	0.7	4.2	3.5	24
Max	396.0	1.40	0.76	2.20	3.4	24.0	19.2	120
Min	4.7	0.04	<0.02/ND	<0.02/ND	0.2	<2/ND	<2/ND	<10/ND
CENTERS (N = 42)								
Median	53.0	0.26	0.07	0.02	1.1	3.4	4.8	19
Mean	879.0	2.73	0.16	0.08	4.0	19.2	27.3	111
Max	24,600.0	74.00	1.20	0.69	91.0	540.0	710.0	2,900
Min	7.2	0.03	<0.02/ND	<0.02/ND	0.3	<2/ND	<2/ND	<10/ND
OUTLETS (N = 48)								
Median	23.5	0.12	0.10	0.03	0.7	4.0	3.2	18
Mean	100.4	0.19	0.15	0.09	0.8	7.6	13.5	33
Max	2,050.0	0.83	0.64	0.44	1.8	91.0	300.0	460
Min	4.3	<0.02/ND	<0.02/ND	<0.02/ND	<0.2/ND	<2/ND	<2/ND	<10/ND

	Oil and Grease (mg/L)	pH	Dissolved Oxygen (DO) (%)	Dissolved Oxygen (DO) (mg/L)	Specific Conductivity (µS/cm)	Water Temperature (°C)
OVERALL (N = 145)						
Median	<10/ND	6.8	49.2	4.68	203.2	17.2
Mean	11	6.8	51.3	5.27	225.2	16.2
Max	42	8.5	106.5	11.97	881.5	28.3
Min	<10/ND	5.8	0.1	0.01	35.2	2.01
INLETS (N = 52)						
Median	<10/ND	7.0	71.3	6.95	213.0	17.1
Mean	10	7.0	67.0	6.75	234.9	16.4
Max	15	8.5	106.5	11.97	716.1	26.4
Min	<10/ND	6.0	5.8	0.70	35.2	4.9
CENTERS (N = 42)						
Median	<10/ND	6.7	28.8	3.32	205.0	17.6
Mean	11	6.6	35.3	3.57	235.8	16.2
Max	42	7.4	97.7	10.08	668.5	28.3
Min	<10/ND	5.9	0.1	0.01	80.2	2.9
OUTLETS (N = 48)						
Median	<10/ND	6.7	45.7	4.34	186.6	18.2
Mean	10	6.6	47.4	5.07	204.9	15.8
Max	12	7.3	102.6	11.03	881.5	27.2
Min	<10/ND	5.8	0.6	0.25	73.3	2.01

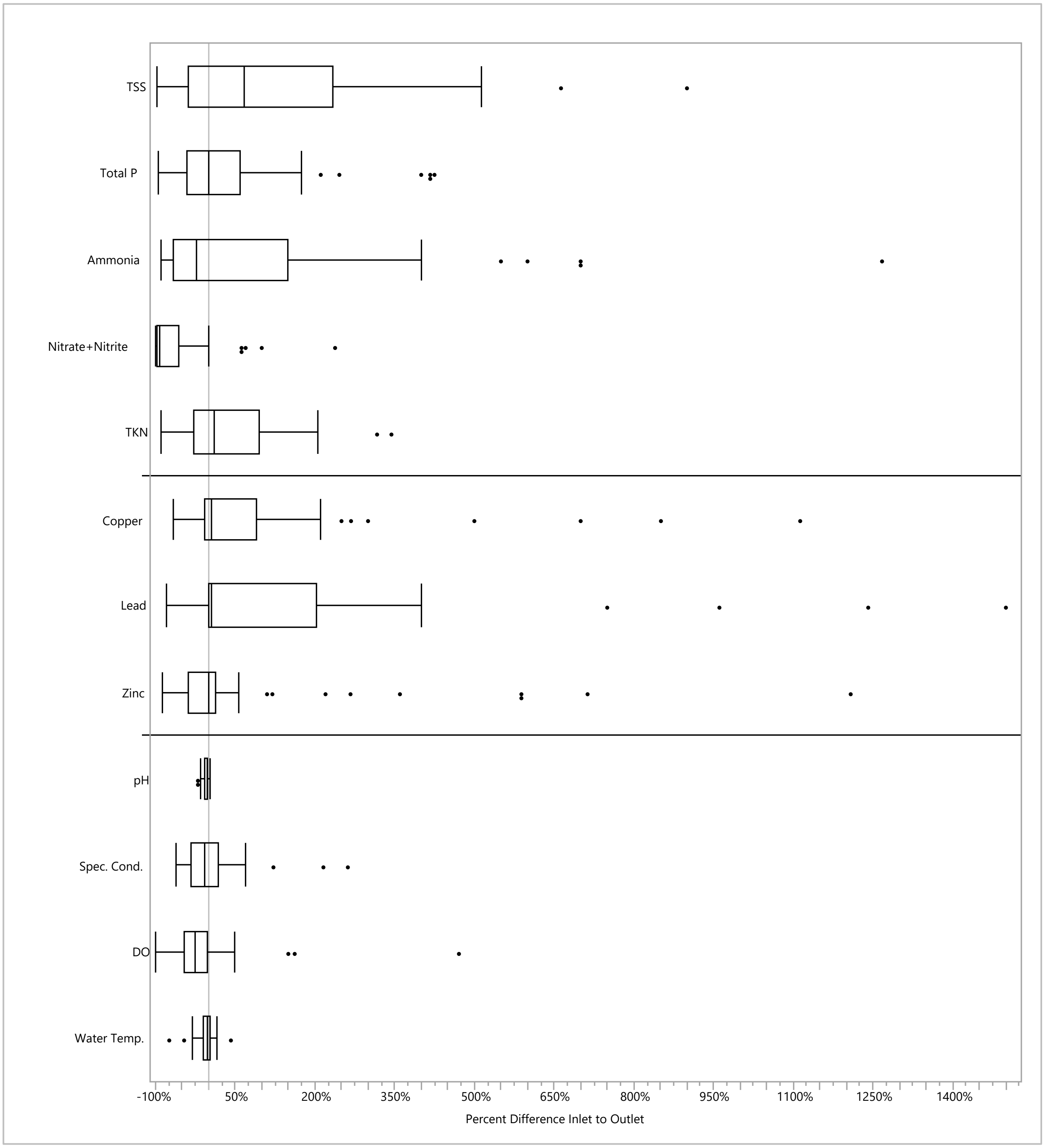


Figure 34. Percent difference between inlet and outlet of intensive wetland study sites by parameter, all samples combined. Three outliers were excluded for graph readability – (lead: 4,588% increase; TSS: 2,179% and 2,104% increase).
A negative percent difference means the parameter concentration was lower at the outlet than the inlet; a positive percent difference means the parameter was higher at the outlet than the inlet.

3.1.3. Water Quality Parameters at Wetland Inlets, Outlets, and Centers During Baseflow and Storms

Samples were collected during five storm events and nine baseflow times. Concentration levels were compared between inlets, centers, and outlets during baseflow and storm flow using the Wilcoxon signed-rank test (Table 5). Data obtained from eight supplemental wetland sites during baseflow were graphed alongside intensive site data and indicated similar ranges for all parameters (Figure 35 through Figure 37); these supplemental data were not included in statistical tests.

Concentrations of total suspended solids were significantly higher in wetland centers than inlets and outlets during baseflow times ($p<0.05$) but not during storms, and were usually higher at outlets than inlets in both scenarios but the differences were not statistically significant. Nitrate + nitrite concentrations were significantly higher in wetland inlets than in wetland centers and outlets during both baseflow ($p<0.05$) and storm events ($p<0.05$) (Figure 35). Nitrate + nitrite concentration was reduced by a median of 93% between inlet and outlet during baseflow conditions and reduced by a median of 76% during storm conditions (Figure 38). Of the other nutrients (ammonia, TKN, and total phosphorus), no significant differences in concentrations were detected between inlets and outlets during baseflow or storms, although TKN and total phosphorus concentrations were significantly higher in wetland centers than at inlets and outlets during baseflow conditions ($p<0.05$).

During baseflow times, but not during storms, copper and lead concentrations were significantly higher in wetland outlets compared to inlets ($p<0.05$) (Figure 36). Zinc concentrations showed no difference between inlets and outlets at any time ($p>0.05$). However, during baseflow times, concentrations of all three metals (copper, lead, and zinc) were higher in wetland centers than inlets at three of the four intensive study sites (HR, WP, and ZH; $p<0.05$). During storms, metal concentrations were not significantly different between all station types (inlet vs. center vs. outlet). However, it is worth noting that copper and zinc concentrations were significantly higher coming into wetlands during storms than during baseflow conditions.

Oil and grease levels were nearly always at or below the level of detection by the water quality lab (10 mg/L) during baseflow and storms, except for one outlier at ZH center (42 mg/L) and detection during supplemental baseflow sampling in 2022 at several wetland inlets (12 to 15 mg/L).

Dissolved oxygen and pH were always significantly lower at outlets than inlets, during both baseflow ($p<0.05$) and storms ($p<0.05$) (Figure 37). Specific conductivity was significantly lower at outlets than inlets (median 15% reduction; $p<0.05$) during baseflow times, but higher at outlets than inlets during storms, though this difference was not statistically significant (Figure 38). During baseflow times, specific conductivity was significantly lower in wetland centers than at inlets ($p<0.05$). Wetland centers were warmer, more acidic, and less oxygenated than inlets during both baseflow times and storms ($p<0.05$).

Table 5. Median values for water quality parameters in four intensive study wetlands for baseflow and storm flow; Sample size (N) depended on presence of water during sampling. Asterisks denote significant differences between inlet and outlet concentrations (Wilcoxon signed-rank test; p<0.05). ND = non-detect.

Station	Sample Type (N)	Total Suspended Solids (TSS) (mg/L)	Total Phosphorus (TP) (mg/L)	Ammonia (NH_3) (mg/L)	Nitrate + Nitrite (NO_3^- + NO_2^-) (mg/L)	Total Kjeldahl Nitrogen (TKN) (mg/L)
INLET	Baseflow (35)	13.0	0.12	0.12	0.56*	0.54
CENTER	Baseflow (34)	58.5	0.23	0.08	0.02/ND	1.10
OUTLET	Baseflow (33)	21.0	0.12	0.12	0.02/ND*	0.63
INLET	Storm (20)	16.5	0.18	0.07	0.35*	0.67
CENTER	Storm (8)	37.5	0.32	0.05	0.02/ND	1.16
OUTLET	Storm (16)	25.5	0.13	0.05	0.08*	0.72

Station	Sample Type (N)	Oil and Grease (mg/L)	Copper (Cu) (µg/L)	Lead (Pb) (µg/L)	Zinc (Zn) (µg/L)
INLET	Baseflow (35)	10/ND	2.10*	2.00*	10.0
CENTER	Baseflow (34)	10/ND	3.05	5.55	19.0
OUTLET	Baseflow (33)	10/ND	3.50*	3.70*	18.0
INLET	Storm (20)	10/ND	4.45	2.60	18.5
CENTER	Storm (8)	10/ND	3.95	3.25	19.5
OUTLET	Storm (16)	10/ND	4.90	3.20	16.5

Station	Sample Type (N)	pH	Dissolved Oxygen (DO) (%)	Dissolved Oxygen (DO) (mg/L)	Specific Conductivity (µS/cm)	Water Temperature (°C)
INLET	Baseflow (35)	7.01*	69.3*	7.12*	266.8*	14.2
CENTER	Baseflow (34)	6.71	32.0	3.78	226.9	14.3
OUTLET	Baseflow (33)	6.70*	40.4*	4.28*	213.0*	13.7
INLET	Storm (20)	7.04*	75.6*	6.80	155.8	22.8
CENTER	Storm (8)	6.56	24.0	1.95	138.5	24.7
OUTLET	Storm (16)	6.67*	50.7*	4.24	150.4	22.5

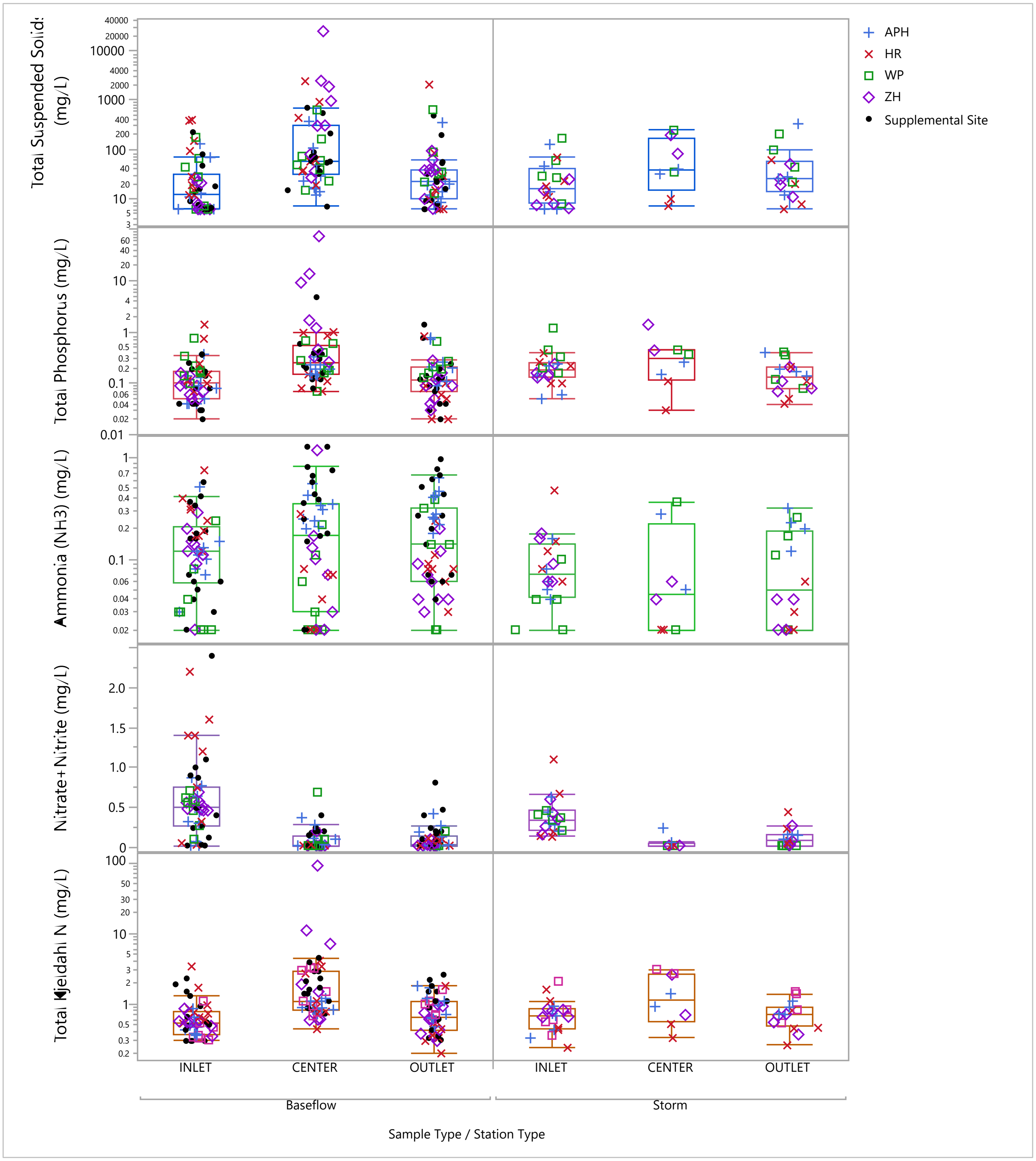


Figure 35. Nutrient concentrations during baseflow and storms for all intensive and supplemental sites, log base 10 y-axes.
N=8 to 50, depending on presence of water and ability to sample storms.

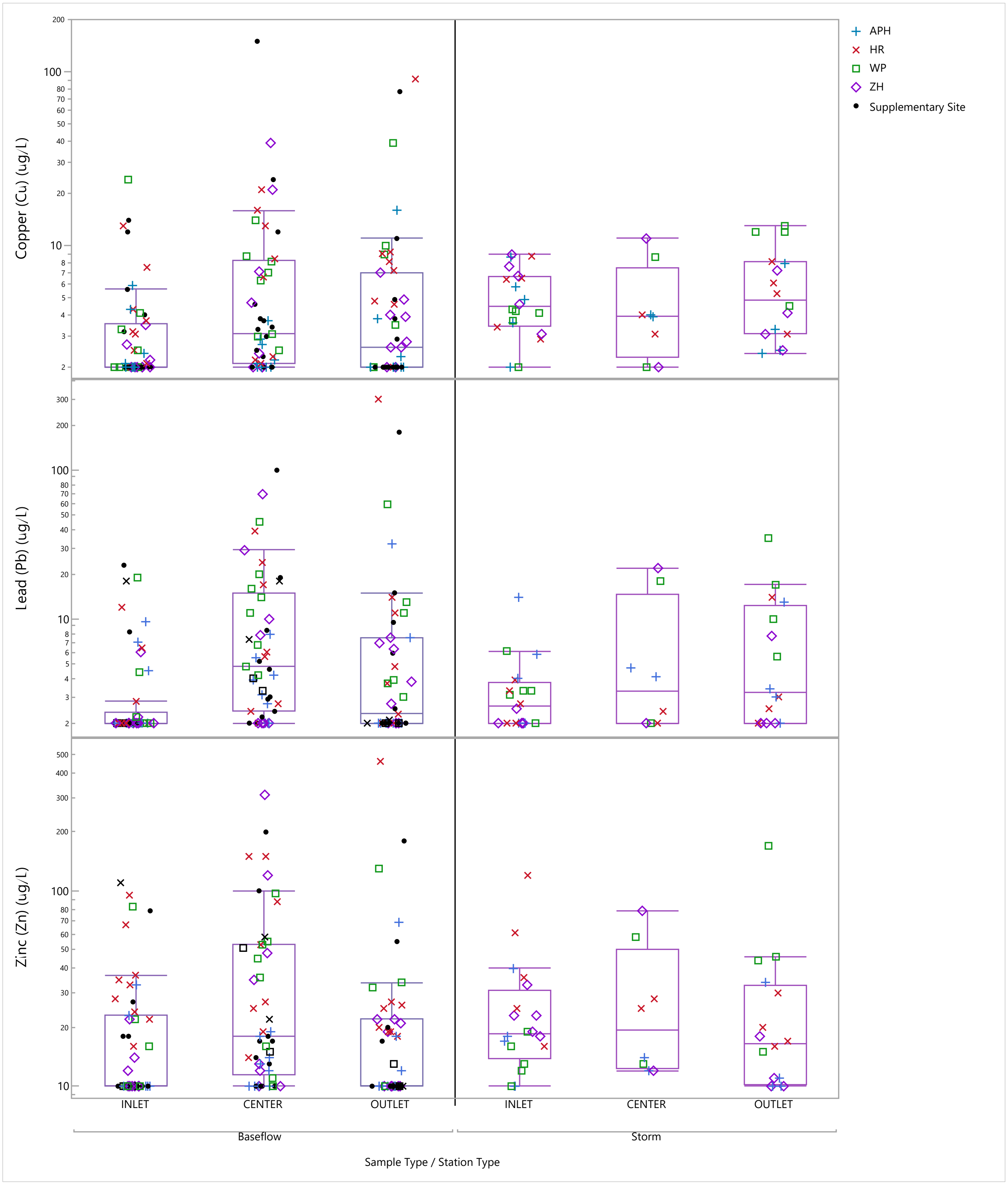


Figure 36. Metal concentrations during baseflow and storms for all intensive and supplemental sites, log base 10 y-axes. One outlier sample from ZH center excluded from all graphs for readability (copper: 540 µg/L; lead: 710 µg/L; zinc: 2900 µg/L).
N=16 to 35, depending on presence of water and ability to sample storms.

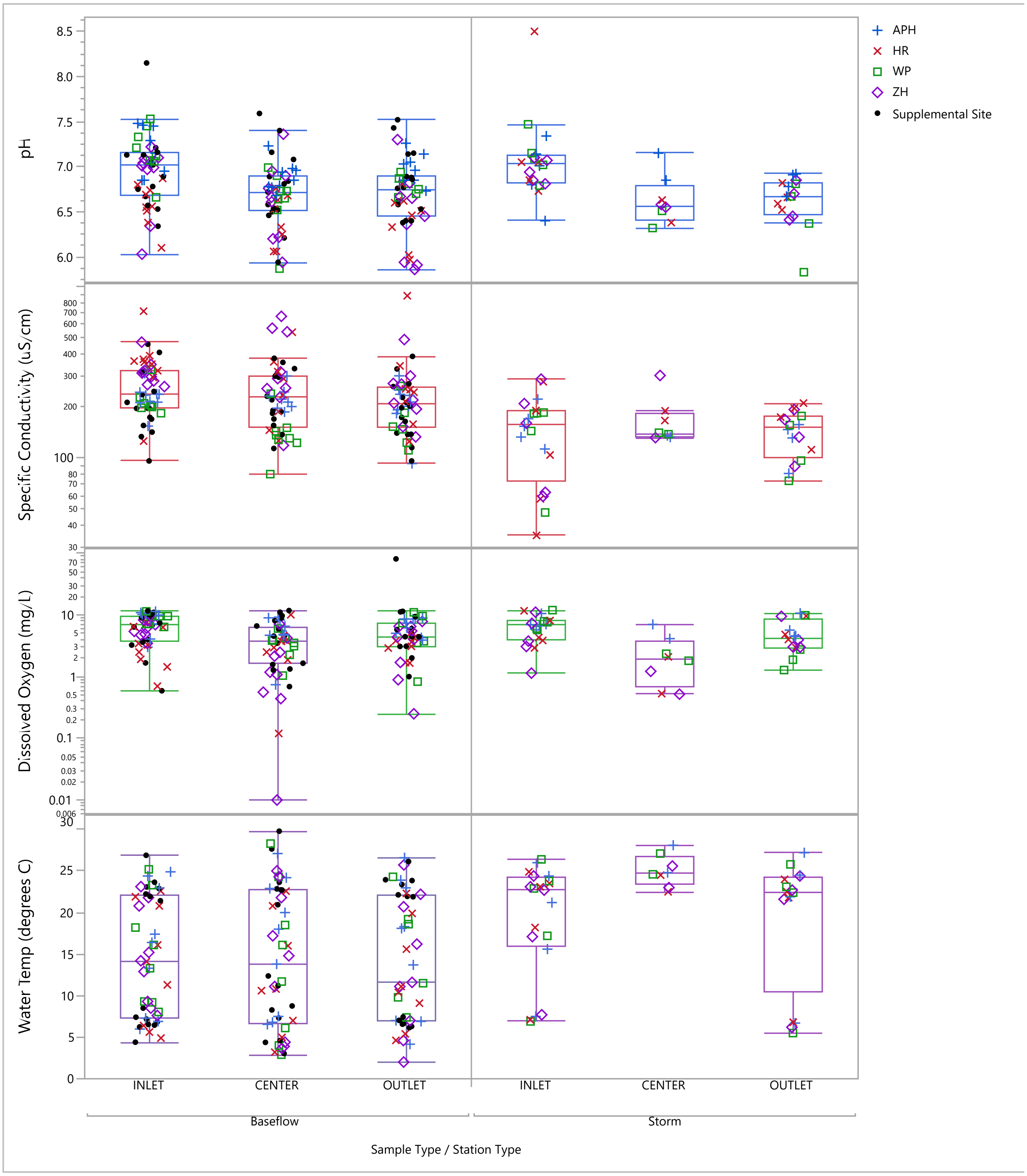


Figure 37. Field water quality parameters during baseflow and storms for all intensive and supplemental sites. Specific conductivity and dissolved oxygen y-axes on log base 10 scale. N=16 to 35, depending on presence of water and ability to sample storms.

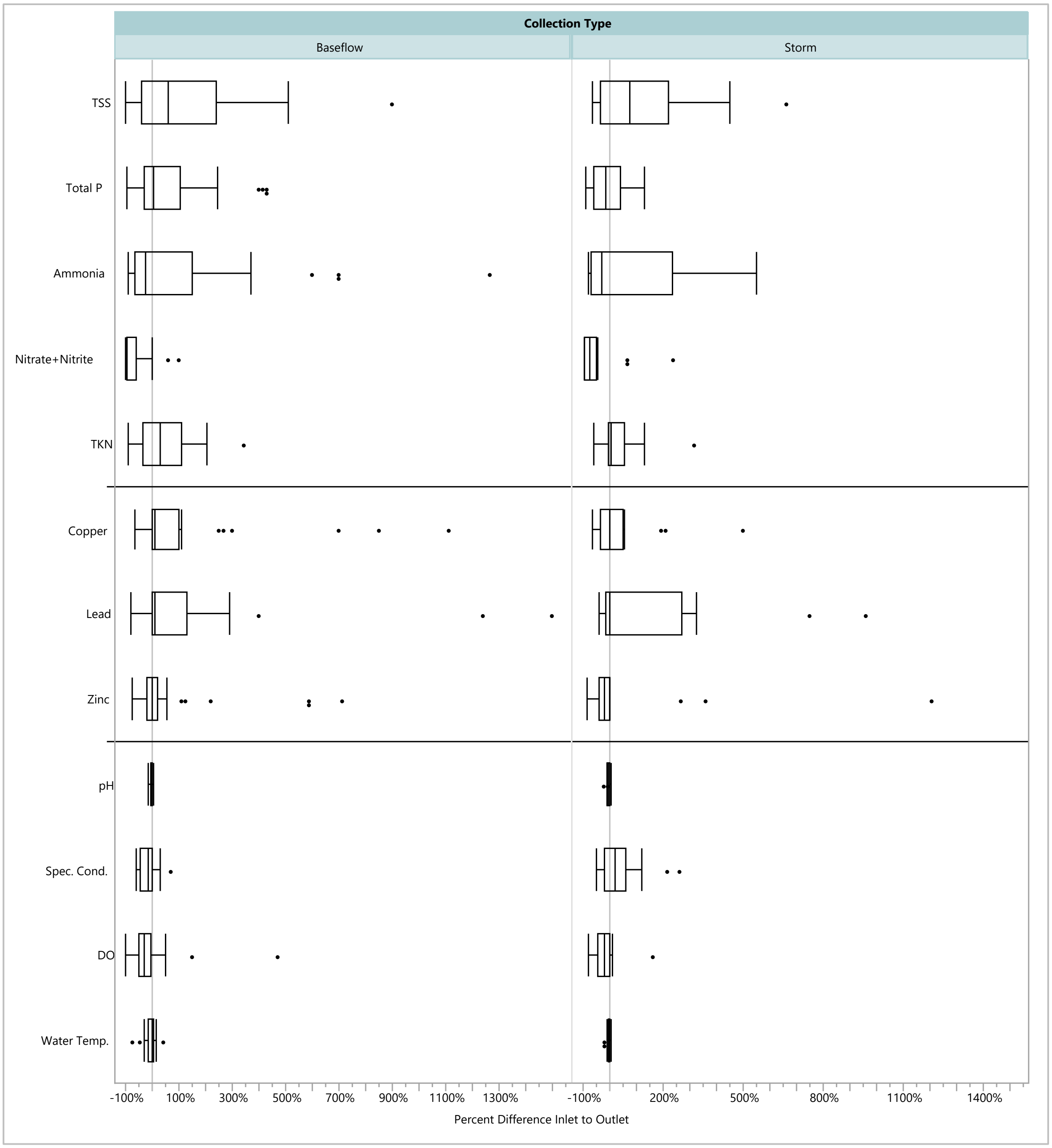


Figure 38. Percent difference between inlet and outlet of intensive wetland study sites by parameter during baseflow and storms. Three baseflow outliers were excluded for graph readability (lead: 4,588% increase; TSS: 2,179% and 2,104% increase).
A negative percent difference means the parameter concentration was lower at the outlet than the inlet; a positive percent difference means the parameter was higher at the outlet than the inlet.

3.1.4. Water Quality Parameters at Wetland Inlets, Outlets, and Centers During Growing Seasons and Non-growing Seasons

Samples were collected during the growing season (April through September) seven times and during the non-growing season (October through March) seven times, during both baseflow and storm events. Concentration levels between inlets and outlets were analyzed by growing season to determine whether actively growing vegetation was a factor in differences between inlet and outlet water quality (Table 6). Inlet/outlet comparisons were made within parameters for the intensive sites in growing season and non-growing season using Wilcoxon signed-rank tests. Data from the eight supplemental sites were graphed alongside the intensive sites data and indicated similar ranges for all parameters during both seasons (Figure 39 through Figure 41); these supplemental data were not included in statistical tests.

Nitrate + nitrite was significantly lower in wetland outlets compared to inlets, and significantly lower in wetland centers than inlets, regardless of the time of year ($p<0.05$; Figure 39). TKN concentrations were significantly lower at inlets than at outlets during growing seasons ($p<0.05$; median increase of 36% from inlet to outlet) but did not change significantly from inlet to outlet during non-growing seasons ($p>0.05$; median decrease of 4%; Table 6, Figure 39). TKN was significantly higher in wetland centers than at inlets or outlets during both seasons ($p<0.05$).

For ammonia and total phosphorus, no differences were detected between inlets and outlets during either season. Concentrations of both total phosphorus and total suspended solids were significantly higher in wetland centers than at inlets during both seasons ($p<0.05$). Total suspended solids were significantly higher in wetland outlets than inlets during growing seasons only (median increase of 96%; $p<0.05$). However, the difference could be attributed to three samples; removing these three outliers eliminated the significant difference between TSS in the inlets versus the outlets. The outliers were from WP (Sept. 17, 2020 storm: 206 mg/L; June 25, 2020 baseflow: 638 mg/L) and HR (June 25, 2020 baseflow: 2050 mg/L) outlets. The WP site had a large, open water, pooled area in the center which was always wet, leading to the accumulation of plant matter and development of deep fine sediment, which was easily resuspended into the water column by waterfowl, beavers, or storms. On the occasion of the high TSS at the HR site outlet, a newly constructed beaver dam had backed up water in the outlet, suspending sediment into the water column and slowing flow (velocity was almost undetectable at 0.09 ft/sec when the sample was taken).

Lead and copper concentrations were significantly higher in wetland outlets than inlets, but only during growing seasons ($p<0.05$; Figure 40). The significant difference found for copper could be attributed to samples taken on June 25, 2020 from HR (91 µg/L) and WP (39 µg/L) outlets. Removal of these two samples, which were also two of the three aforementioned TSS outlier samples, also eliminated the significant difference originally found for copper between the inlets and outlets during the growing season. Removal of those outliers did not eliminate the significant difference between inlet and outlet lead concentrations, which had many more instances of higher concentrations in the outlet than the inlet.

Lead concentrations were significantly higher in wetland centers than in inlets during both the growing and non-growing seasons ($p<0.05$), but there was no difference in lead concentrations between wetland centers and outlets in either season ($p>0.05$). Lead levels were variable and exhibited a wide range in concentrations from below 2 µg/L (undetectable) to 710 µg/L. Copper and zinc showed no significant difference between wetland centers and inlets or outlets during either season. Zinc showed no significant differences in concentration between inlets and outlets, regardless of season.

Dissolved oxygen and pH were significantly lower in wetland outlets compared to inlets, regardless of season ($p<0.05$) and lower in wetland centers than inlets ($p<0.05$). During the non-growing season, specific conductivity was significantly lower at outlets than inlets ($p<0.05$). Water temperature showed no statistically significant change from inlet to outlet in either season.

Table 6. Median values for water quality parameters in four intensive study wetlands by growing season/non-growing season; sample size (N) depended on presence of water during sampling. Asterisks denote significant differences between inlet and outlet concentrations (Wilcoxon signed-rank test; p<0.05). Plus sign indicates significant differences in TSS and copper that disappeared after three outliers were removed. ND = non-detect.

Station	Sample Type (N)	Total Suspended Solids (TSS) (mg/L)	Total Phosphorus (TP) (mg/L)	Ammonia (NH_3) (mg/L)	Nitrate + Nitrite (NO_3^- + NO_2^-) (mg/L)	Total Kjeldahl Nitrogen (TKN) (mg/L)
INLET	Growing Season (27)	12.0^{+}	0.16	0.10	0.43*	0.65*
CENTER	Growing Season (22)	37.5	0.30	0.07	0.02/ND	1.10
OUTLET	Growing Season (26)	26.0^{+}	0.18	0.16	0.03*	0.81*
INLET	Non-growing Season (28)	18.5	0.11	0.12	0.46*	0.55
CENTER	Non-growing Season (20)	66.5	0.20	0.07	0.02/ND	0.89
OUTLET	Non-growing Season (23)	20.0	0.09	0.07	0.03*	0.54

Station	Sample Type (N)	Oil and Grease (mg/L)	Copper (Cu) (µg/L)	Lead (Pb) (µg/L)	Zinc (Zn) (µg/L)
INLET	Growing Season (27)	10/ND	3.40^{+}	2.00*	16.0
CENTER	Growing Season (22)	10/ND	3.80	4.45	17.5
OUTLET	Growing Season (26)	10/ND	4.75^{+}	3.80*	17.5
INLET	Non-growing Season (28)	10/ND	2.50	2.00	16.0
CENTER	Non-growing Season (20)	10/ND	2.90	5.75	22.0
OUTLET	Non-growing Season (23)	10/ND	2.60	3.00	18.0

Station	Sample Type (N)	pH	Dissolved Oxygen (DO) (%)	Dissolved Oxygen (DO) (mg/L)	Specific Conductivity (µS/cm)	Water Temperature (°C)
INLET	Growing Season (27)	7.02*	63.5*	5.72*	200.70	23.1
CENTER	Growing Season (22)	6.64	23.9	2.18	186.85	24.3
OUTLET	Growing Season (26)	6.63*	36.9*	3.43*	192.60	22.5
INLET	Non-growing Season (28)	7.02*	82.1*	9.55*	251.15*	9.3
CENTER	Non-growing Season (20)	6.75	35.8	4.08	226.90	6.9
OUTLET	Non-growing Season (23)	6.81*	50.7*	6.52*	156.80*	7.0

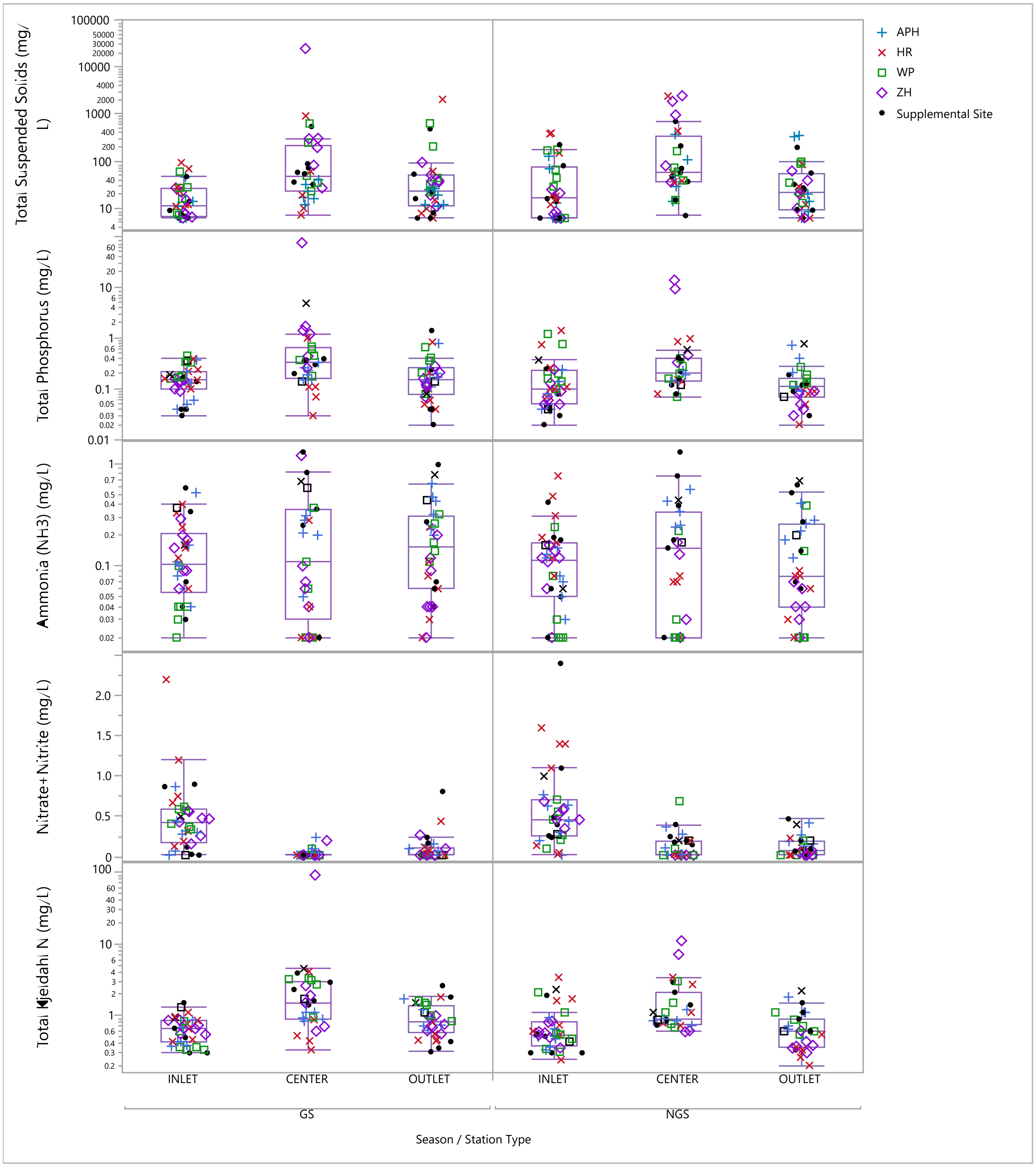


Figure 39. Nutrients during growing season (GS) and non-growing season (NGS), baseflow and storm flow combined for all intensive and supplemental sites. Y-axes on log base 10 scale, except nitrate + nitrite. N=23 to 28, depending on presence of water.

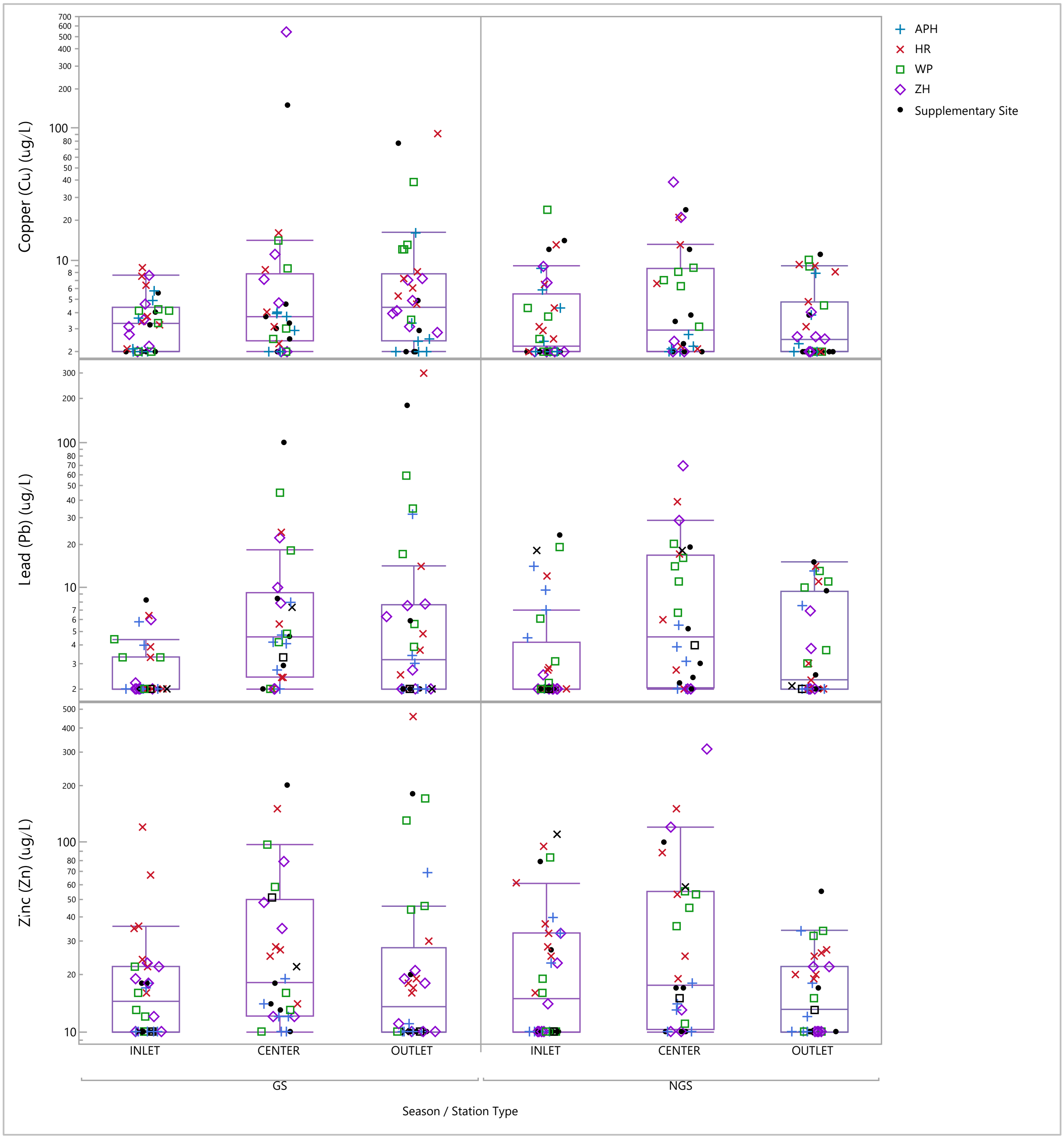


Figure 40. Metal concentrations during growing season (GS) and non-growing season (NGS), baseflow and storm flow combined for all intensive and supplemental sites. Y-axes on log base 10 scale. N=23 to 28, depending on presence of water.

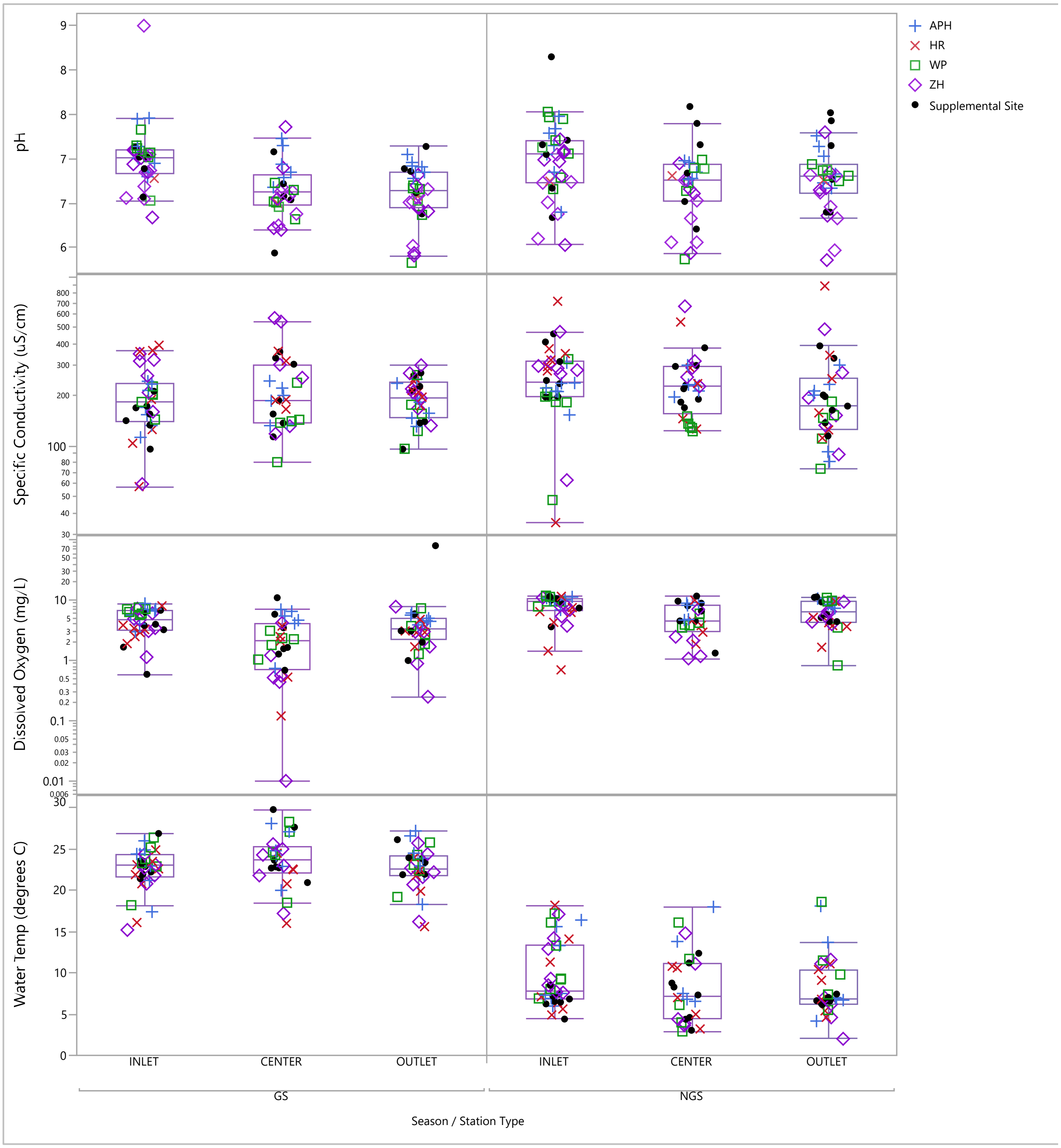


Figure 41. Field water quality parameters during growing season (GS) and non-growing season (NGS), baseflow and storm flow combined for all intensive and supplemental sites. Specific conductivity and dissolved oxygen y-axes on log base 10 scale.
N=23 to 28, depending on presence of water.

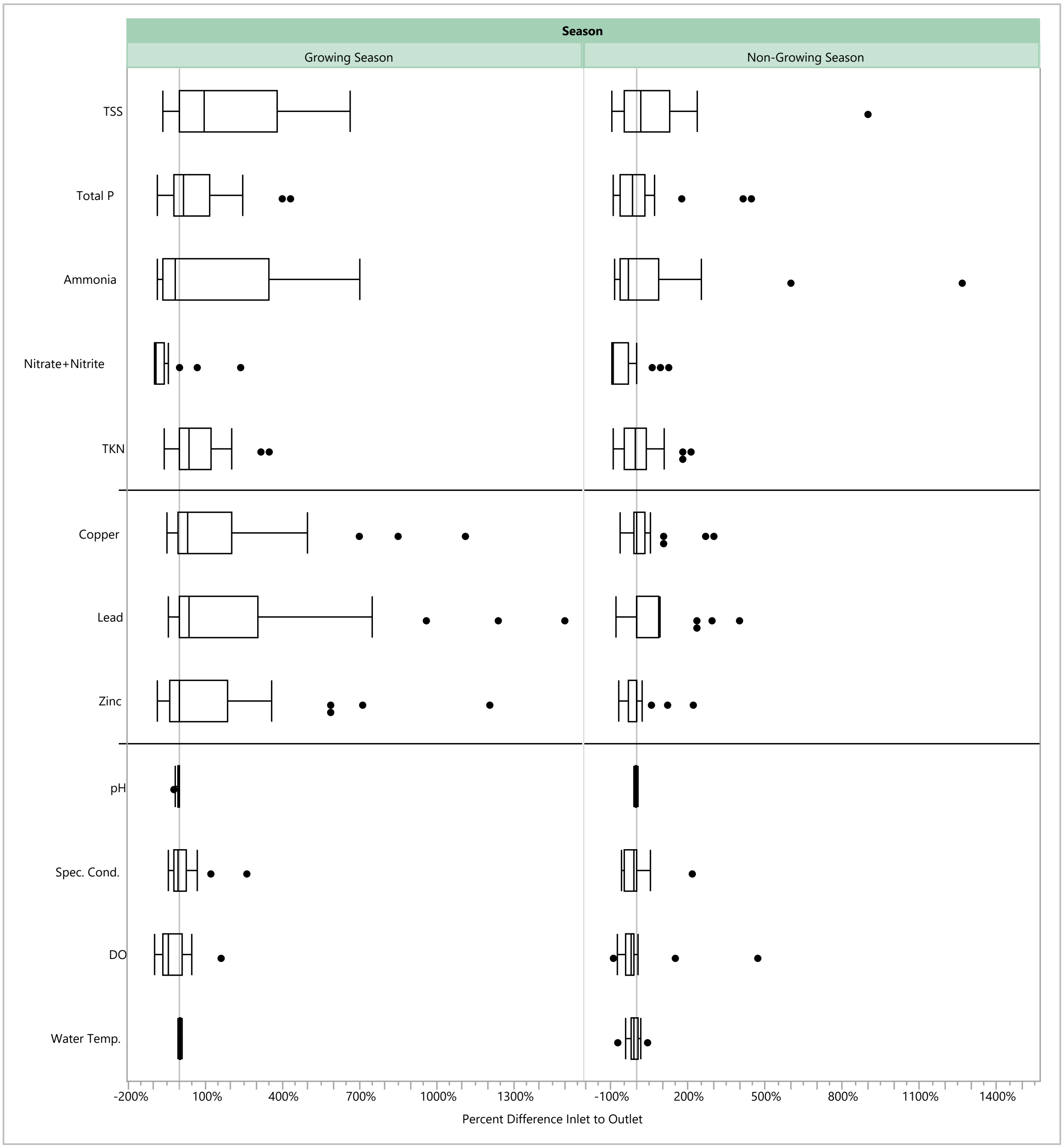
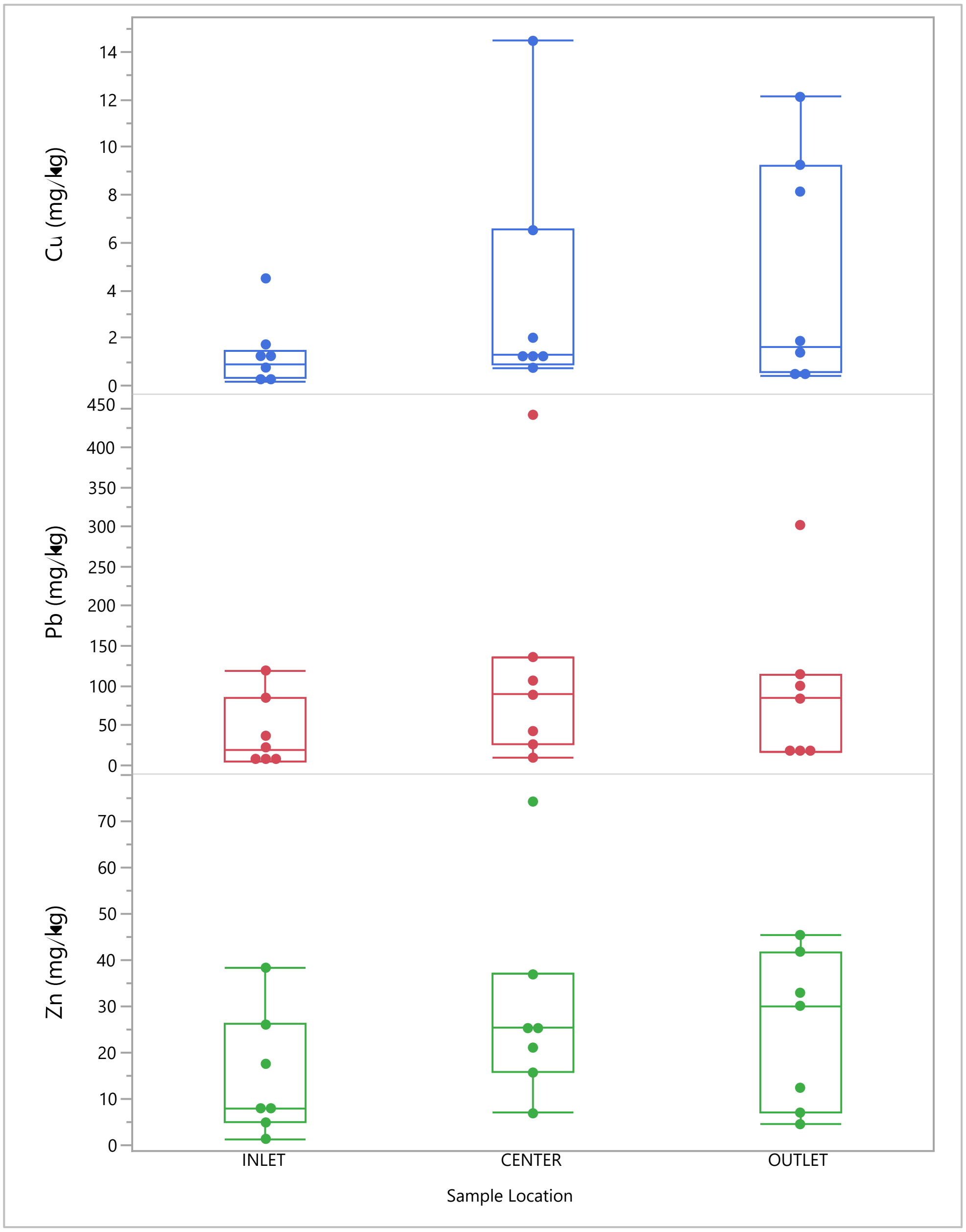


Figure 42. Percent difference between inlet and outlet of wetland study sites by parameter and by season. Three growing season outliers were excluded for graph readability – (lead: 4,588% percent increase; TSS: 2,179% and 2,104% increase).
A negative percent difference means the parameter concentration was lower at the outlet than the inlet; a positive percent difference means the parameter was higher at the outlet than the inlet.

3.2. Sediment Sampling Results

In September 2022, the upper 15 cm of sediment was collected at established water sampling stations from two intensive and five supplemental wetland sites to compare this surface material with grab water samples already taken from the above surface water zone during previous sampling efforts, particularly for copper, lead, and zinc. Two intensive sites could not be sampled for sediment/soils; the APH site was inaccessible due to City greenway maintenance activities at sampling time and the WP site had lost its inlet and long-term outlet stations by the time of soil sampling. The raw sediment analysis data are included in Appendix F.

Results suggested that immediately subaqueous soils were either fixing heavy metals (copper, lead, zinc) commonly found in stormwater in the central parts of the wetland sites or building them up between inlets and outlets, particularly lead and zinc (Figure 43). Soils from the intensive and supplemental wetland sites had a mean copper concentration of 3.5 mg/kg, mean lead concentration of 84.9 mg/kg, and mean zinc concentration of 23 mg/kg.



Sample Location	Mean Copper (Cu) (mg/kg)	Mean Lead (Pb) (mg/kg)	Mean Zinc (Zn) (mg/kg)
Inlet	1.320	40.07	14.802
Center	3.854	121.40	29.281
Outlet	4.790	93.31	24.801

Figure 43. Sediment metals data by sample location from seven wetland sites.

3.3. Sediment Movement Assessment: Sedimentation Flux Assessment

3.3.1. Assessment of Tributary Erosion

Over the course of one year from February 19, 2019 to February 26, 2020, 19.23 cubic meters (25.2 cubic yards) or 15.2% of bank volume was eroded and deposited downstream, including into Walnut Creek main stem. This volume translated to between 22,307 and 24,230 kg of soil eroded from ZH wetland during that time period.

3.3.2. Assessment of Tributary Sedimentation Input

Despite intermediate erosional episodes at certain measurement stakes, the overall accretive build-up in ZH wetland averaged 4.4 cm depth over the course of an approximately three-year observation period (February 18, 2019 to March 15, 2022). Given the area covered, this extrapolated out to a minimum estimate of 41.88 cubic meters (54.78 cubic yards) of inflowing sand and silt at the upper reach of the wetland. This was not a comprehensive estimate; other areas of input in that corner of ZH wetland were untraceable but present. Tracking data and calculations are included in Appendix E.

3.3.3. Assessment of Overbank Sediment Deposition into Marsh Area

Monitoring of overbank deposits at ZH wetland indicated an average overbank deposit depth of 8.74 cm over the course of the observation period (July 22, 2020 to March 15, 2022). Most of the sediment consisted of fine sand but also graded into silt toward the marsh edges. A rough estimate of areal coverage translated into an estimate of 612 cubic meters (800.47 cubic yards) of overbank sediment from Walnut Creek entering ZH wetland during the study time period (Appendix E).

3.3.4. Sediment Movement Assessment Summary

Over 2.5 years, an estimated minimum of 670 cubic meters (876.32 cubic yards) of sediment influxed into ZH wetland from Walnut Creek overbanking. Approximately 7% of this total amount was eroded back into the Creek from the outlet of the marsh area. Most of the material measured in overbanking was fine sand with some silt. Most of the material eroded from ZH outlet was comprised of silt- to clay-sized particles.

4.0 Discussion

This project was designed to provide baseline data to NC DEQ 401 wetland and buffer permitting staff regarding potential mechanisms of water quality uplift in urban settings. Four natural wetlands located adjacent to Walnut Creek in urban Raleigh, North Carolina were studied intensively to investigate how or if natural wetlands affect contamination levels in surface water during baseflow and storm events, as well as during growing and non-growing seasons.

4.1. Antecedent Dry Periods in Relation to Stormwater Constituent Concentrations

Storm events were targeted for sampling when they were preceded by droughts of at least seven days in an attempt to capture measurable accumulations of targeted constituents in the subwatersheds. However, it is possible that this length of time was not long enough for many of the measured contaminants to accumulate enough to create pulses above baseflow levels. During the study period, rain occurred frequently at intervals of fewer than seven days, making it a challenge to sample during a storm with seven or more dry preceding days. During the one storm with a longer preceding drought (39 days; Oct. 16, 2019), some ions appear to have accumulated in the wetland catchments, as indicated by higher specific conductivity at wetland inlets during that storm than during other storms. Other measured parameter concentrations did not appear to be as affected by the length of the drought period, as they were not higher during the October 2019 storm than in other storms. This is consistent with findings by investigators including Gaut et al. (2019), Gorgoglione et al. (2019), Nayeb Yazdi et al. (2021) and Sun et al. (2015) who have called into question the idea that stormwater concentrations are related to antecedent dry period length. Precipitation intensity and surface slope appear to affect stormwater quality more than antecedent dry period (Shaw et al. 2010; Alias et al. 2014; Muthusamy et al. 2018).

4.2. Stormwater Constituent Concentrations

4.2.1. Suspended Solids and Sediment

Processes affecting TSS levels within wetlands include sedimentation (particularly at the surface inlet area[s]), periphyton litterfall into the water body, chemical precipitation of solid compounds created from influent and/or existing interior water, plankton litterfall, macrophyte litterfall, resuspension of interior sediments/soils, and macrophyte surface interception of flowing particulates (Kadlec and Wallace 2009). Wetlands can generate more TSS than is loaded in influent because of the interaction of plant life cycles with nutrient-containing sediments and dissolved constituents, as well as the non-biotic geochemical processes that may be occurring at the same time. A large fraction of the effluent TSS at any time therefore can be wetland-generated. Kadlec and Wallace (2009) concluded that “the solids leaving the wetland will very often not be related to the solids entering but rather to the detrital fragments originating internal to the system" (page 217). Interactions between influent and internal storages can also be highly variable by season and diurnal conditions, so modeling removal rates for TSS is highly complex.

Mean TSS concentration was not different from inlet to outlet, regardless of flow level (baseflow vs. storms) in the intensive study wetlands. TSS concentrations were, however, higher in wetland centers than inlets and outlets, probably due to much slower water velocities and the accumulation of plant and algal matter. Significantly higher TSS in outlets than inlets during the growing season could be attributed to three samples with very high concentrations from WP and HR; removing those three outliers eliminated the significant difference. The WP site had a large (~12 acre) and deep (> 3.3 feet or 1 meter) open water area in the center, leading to the accumulation of plant matter and development of deep fine sediment that was easily resuspended into the water column by waterfowl, beavers, storms, or wind action. Wildlife activity, particularly among beavers, was probably greater during the growing season. On the occasion of the high TSS at the HR site outlet, a newly constructed beaver dam had backed up water in the outlet, suspended sediment into the water column, and slowed flushing (flow was 0.09 ft/sec; almost undetectable). Significantly higher copper levels in outlets during the growing season could also be attributed to these same water samples from WP and HR in June 2020.

That said, estimates of sediment input into one study wetland showed significant trapping of sediment; thousands of cubic meters of sand and silt from overbanking were estimated to be trapped by ZH wetland with a small percentage eroding back into Walnut Creek from the wetland stream outlets.

4.2.2. Phosphorus

Lawns and fallen leaves are major contributors of phosphorus to stormwater (Waschbusch et al. 1999; Hobbie et al. 2017). Two studies in Wisconsin showed that phosphorus peaks in urban runoff could be largely attributed to contributions from fallen leaves in autumn (Selbig 2016; Wang et al. 2022) but in this study, seasonality did not appear to affect phosphorus inlet concentrations. No significant differences were detected in total phosphorus concentrations between inlets and outlets, regardless of flow conditions or season, despite the fact that total phosphorus was significantly higher in wetland centers during baseflow all year.

Mean total phosphorus concentration (0.22 mg/L) was somewhat lower than that expected for stormwater in urbanized settings based on US national stormwater data from metropolitan areas across the nation (predicted total phosphorus mean 0.31 mg/L; May and Sivakumar 2009). In Florida, stormwater runoff total phosphorus mean concentration across 29 storm events was 0.28 mg/L (Yang and Toor 2018).

4.2.3. Nitrogen

Concentrations of nitrate + nitrite were significantly reduced by the study wetlands, regardless of baseflow/storm situation or season. Nitrate + nitrite is indicative of general urban impacts, while the presence of ammonia can be indicative of impacts from sewers or septic systems (via anaerobic conditions) (Gibb 2000). Nitrate + nitrite and ammonia concentrations found at wetland inlets were not out of the ordinary for urban stormwater runoff in the US, perhaps even low (Moore et al. 2011; Li and Davis 2014; Jani et al. 2020). Mean ammonia (0.14 mg/L) and TKN (0.7 mg/L) at inlets were lower than that predicted for stormwater in urbanized settings based on US national stormwater data (predicted ammonia mean 0.26 mg/L; predicted TKN mean 1.93 mg/L; May and Sivakumar 2009). Ammonia and TKN concentrations showed no significant change from inlet to outlet in relation to storm/baseflow conditions, despite the fact that TKN was significantly higher in wetland centers under baseflow conditions all year.

However, seasonality appeared to affect changes in TKN concentration, which showed an increase from inlet to outlet during the growing season. Mean concentrations of TKN in the study wetland inlets (0.71 mg/L) and outlets (0.77 mg/L) were higher than those measured in streams across North Carolina (0.52 mg/L) and in Raleigh specifically (0.34 and 0.60 mg/L) (NC DEQ DWR 2015; National Water Monitoring Council 2019; USGS 2019; EPA 2020; Appendix H). TKN, as the sum of organic nitrogen and ammonia, can be an indicator of animal waste or fertilizer (Groffman et al. 2004; Kaushal et al. 2006; Lusk et al. 2020). Growing season had no effect on the relationship of the inlet to the outlet for ammonia, but organic nitrogen was significantly higher leaving through the outlet than in the inlets during the growing season ($p<0.05$). The presence of higher organic nitrogen levels at the outlets during the growing season is likely related to the photosynthetic activity and animal activity happening seasonally within the wetlands (Mu and Chen 2021).

4.2.4. Oil and Grease

Unexpectedly, oil and grease levels were nearly always at or below the level of detection (10 mg/L), except one outlier at ZH center (42 mg/L) and at the inlets of several supplemental wetlands during baseflow sampling in 2022 (concentrations just above detection limit). This was despite the Interstate 40 lying adjacent to two of the intensive study sites. These low oil and grease concentrations could be due to the general absence of large impervious parking areas in the wetland watersheds (Table 1) and stormwater management measures for the newly expanded interstate. However, recent construction of a large school and associated driving and parking areas adjacent to ZH wetland, with a culvert directing stormwater into its inlet, may result in changes in inlet oil and grease concentrations into the future.

4.2.5. Heavy Metals: Copper, Lead, and Zinc

The two major modes by which wetlands remove and retain metals from surface water are uptake by plants and sorption by organic molecules (e.g., ligands) (Vymazal and Richardson 1995; Kadlec and Wallace 2009). The uptake of metals by vascular plants in wetlands mostly occurs in the rootzone and not in the upper plant body (Nolte and Associates 1998; Nabulo et al. 2008). However, algae inhabiting the study wetland marshes (at the wetland centers) are known sequesters of heavy metals (Jahan et al. 2004; Ellison et al. 2014; Matei et al. 2015; Williams, A. pers. comm. 2021). Algae can uptake concentrations of metals in order[s] of magnitude greater than vascular plants; lead can be taken up by algae at levels 200 times higher than macrophytes (Kadlec and Wallace 2009). Algae have also been shown to effectively remove and retain zinc from contaminated waters (Fielding et al. 2022). Living algal tissues have been found to have copper concentrations of 100-1000 µg/g dry mass, while macrophytes generally have average copper concentrations of only 1-20 µg/g dry mass (Kadlec and Wallace 2009). Highly organic (peat) soils in wetlands can sorb copper to a very high concentration: 100 mg/g (Brown et al. 2001). Additionally, metals can settle to the bottom of wetland standing water through transformations and precipitations with other substances, but these change easily with changing water and substrate conditions (e.g., redox).

Atmospheric deposition most likely is a contributor to heavy metals in wetlands, as precipitation collects in low-lying areas such as these. Atmospheric deposition (both wet and dry) has been shown to be a source of copper, cadmium, lead, nickel, and zinc (Claytor and Schueler 1996). Additionally, industrial roof material can be a source of copper and zinc in stormwater (Winters et al. 2015; Galster and Helmreich 2022). Both copper and zinc were present in significantly higher concentrations in study wetland inlets during storms than during baseflow.

Higher concentrations of total metals were evident in wetland centers at baseflow times in particular; during storms, concentrations were diluted by higher water volumes. During the growing season in the intensive study wetlands, lead concentrations significantly increased from inlet to outlet. Soil testing showed higher copper and lead concentrations in wetland centers, so higher lead concentrations at the outlets during the growing seasons could also have been caused by animal and/or storm flow activity in the wetlands mobilizing soils with legacy contaminants in them. Because algae are known to bioconcentrate heavy metals, growing season proliferation and then mobilization of algae from wetland centers could drive elevated lead concentrations in outlets relative to inlets.

Metals must be in dissolved form (bioavailable) to be uptaken by plants or algae; metals bound to soil are not generally bioavailable. Water samples can be analyzed for either dissolved metals or total metals. During the first water sampling event in this study, the research team collected samples for dissolved metals analysis in addition to total metals and concluded from the data that results of the two methods were close enough to warrant collecting samples only for total metals. The decision was made in favor of using total metals in part because sampling for dissolved metals requires on-site filtering (at a vehicle) within 15 minutes of collection; doing so was impractical when collecting water from deep inside large wetlands with few research staff, particularly when trying to catch early stormwater during storms. Revisiting the dissolved vs. total metals data from the first sampling event revealed that sometimes dissolved metals were not detected or very low when there was a spike in total metals concentrations. The high total metals values (with low/undetected dissolved metals) were from WP center, in particular. Perhaps much of the high total metals concentrations observed in this study, especially in the wetland centers, were more from sediment or TSS than bioavailable dissolved metals. While total metals data are useful for an overall picture of contamination, in retrospect, it would have been preferable to also collect and filter samples for dissolved metals to measure how much was bioavailable, particularly in the wetland centers where total metals were frequently high. In 2015, the state of North Carolina changed most of its surface water quality standards from those based on total metals to those based on dissolved metals to better protect aquatic life.

4.2.5.1. Metals in Wetland Soils

Levels of soil copper and zinc in the intensive and supplemental wetland sites from this study (mean copper 3.5 mg/kg; mean zinc 23 mg/kg) could be considered high, compared to other North Carolina wetland soil analysis results from 216 rural natural wetland sites across the Piedmont and Coastal Plain of North Carolina (mean copper 0.99 mg/kg; mean zinc 2.63 mg/kg; NC DWR unpubl. data). Copper and zinc concentrations in water entering the study wetlands were higher during storms than during baseflow, implying that stormwater was delivering these metals to the wetlands. Commercial/industrial building roofing and zinc-based paints in urban areas, along with atmospheric deposition, are sources of copper and zinc; these metals could be accumulating in wetland soils.

Mean lead levels in the urban study wetland sediments were 66.6 mg/kg, or more than three times as high as mean background lead levels in soils in North Carolina (mean lead 21.3 mg/kg; Smith et al. 2013) and five times more than that reported from 50 rural natural forested wetlands across the Southeast (mean lead 12.6 mg/kg; Savage et al. 2015). High lead levels in urban wetland soils are most likely a universal legacy effect from historic leaded gasoline use leading to atmospheric deposition of lead, as documented for reservoirs in Kansas and other areas of urban, high-density vehicle use (Juracek and Ziegler 2006; Cabrera 2021). High lead concentrations have also been documented in streams and stream sediments in North Carolina (Caldwell 1992). This legacy lead effect was also reported in a study of soils in over 1,000 randomly selected wetlands across the conterminous US (Nahlik et al. 2019), where soil lead concentration increased with greater urbanization. However, lead can be bound strongly to wetland soils, effectively immobilizing it unless released from a wetland through erosion. Given the water sampling results in the study, there was little evidence of proportionately high amounts of lead leaching from wetland soils into the surface water reaching Walnut Creek.

The ZH site outlet in the study had evidence of a buildup in copper and lead, so it was highly likely that, given the erodible nature of the outlet tributary, ZH wetland could be a source of some amount of total heavy metals to Walnut Creek, after having received the metals from upstream, overland runoff, and/or atmospheric sources. The other site outlets were less erodible and hence less likely to contribute these metals to the Creek. It is unclear if the sites displaying metal accumulation in their centers could also experience future migration of this buildup toward the outlets debouching into Walnut Creek, but long-term observational experience indicated that this is unlikely. Additionally, in a review of the literature, Gambrell (1994) concluded that wetland soils retain metals more effectively than upland soils, even in the face of drainage or flooding, assuming soil pH does not become strongly acidic for some reason.

4.2.6. Dissolved Oxygen and pH

Dissolved oxygen decreased from inlet to outlet, regardless of flow conditions or season. Flowing water (i.e., stream inlets/outlets) generally has higher dissolved oxygen than non-flowing or very slow-moving water, which typically exists in wetlands (Chow et al. 2016). Vegetative structure in wetlands also has the effect of reducing dissolved oxygen as light is intercepted before reaching the water column, reducing photosynthetic oxygen production within the water column (Rose and Crumpton 1996) and vegetative structure has the effect of slowing flows which can potentially raise water temperatures enough to reduce dissolved oxygen, as colder water holds more dissolved oxygen.

The parameter of pH also decreased from inlet to outlet, regardless of flow conditions or season, similar to that reported by Savage et al. (2015) in rural natural riverine wetlands across the Southeast. It is not uncommon for wetlands to have water more acidic than adjacent rivers or streams, therefore it is not unexpected that the pH would drop as it flows across a wetland (Mitsch and Gosselink 2015).

4.2.7. Specific Conductivity

In an analysis of the cumulative effect of wetlands on stream water quality, Johnston et al. (1990) found that increased wetland extent within a watershed was related to decreases in specific conductivity in streams. The urban wetlands in this study had this desirable effect on specific conductivity, but only during the non-growing season (decreasing it from a mean of 264 ± 125 µS/cm at inlets to a mean of 215 ± 156 µS/cm at outlets). Specific conductivity was significantly higher coming into the wetlands through the inlets during the non-growing season when compared to the growing season ($p<0.05$; Wilcoxon test). Specific conductivity has been found to be correlated with dead leaf and woody matter in streams, which could account for higher levels in the non-growing season after leaf fall in autumn (Gessner and Schwoerbel 1989; Hutton et al. 2020). The ratio of groundwater to precipitation in the streams could change with season; groundwater could be contributing more ions in the non-growing season ("winter recharge") than during the spring/summer when it could be diluted more by precipitation (Baker et al. 2019; Bischof et al. 2019). Specific conductivity levels during the non-growing season in the watersheds also could have been impacted by the occasional spraying of deicing brine on nearby roadways during icy times; higher specific conductivity during winter in urban streams has been attributed to this in other places (Moore et al. 2019).

Typical specific conductivity found in natural reference quality (minimally impaired) streams in the Piedmont ecoregion (where Raleigh, NC sits) is usually between 43.8 µS/cm (25th percentile) and 103.0 µS/cm (75th percentile), indicating that the specific conductivity in the intensive study wetlands inlets and outlets was about three times higher (25th percentile: 141.6 µS/cm; 75th percentile: 263.4 µS/cm) than reference (Griffith 2014). Specific conductivity was not well-correlated with any of the water quality parameters measured in this study. However, specific conductivity has been found to be correlated with calcium, magnesium, sodium, potassium, chloride, sulfate, and bicarbonate (Griffith 2014).

4.3. Putting Water Quality Results into Context

4.3.1. Comparison to Stormwater Literature

Published studies on urban stormwater constituents in US cities seem to be uncommon, but some are summarized in Table 7. A recent study of stormwater contaminants in 21 urbanized areas across 17 states in the conterminous US found concentrations, loads, and yields of organic compounds were positively related to impervious surfaces and highly developed urban catchments (Masoner et al. 2019). They reported moderately low median concentrations of copper, lead, and zinc, but higher levels of total phosphorus (Table 7). They did not report data for nitrogen species or TSS.

The USGS sampled 19 headwater streams in Raleigh and found the following: (medians) DO 6.4 mg/L, pH 6.9, specific conductivity 125 µS/cm, TKN 0.28 mg/L, nitrate + nitrite 0.37 mg/L, ammonia 0.02 mg/L, and total phosphorus 0.07 mg/L (USGS 2019). Specific conductivity in these small streams in Raleigh was much lower than mean specific conductivity reported for streams in the urban Washington DC area, which ranged from around 550 to 750 µS/cm (Hopkins 2019).

A comparison of water quality data from the inlet streams during storms in this study showed concentrations to be lower than typical for urban stormwater runoff across the nation. This is probably due to a dilution effect from stormwater runoff being added to permanently running stream water, combined with the effectiveness of existing forested and otherwise vegetated buffers at reducing contaminants before they reached the study streams. Similarly, Horowitz et al. (2008) found that median contaminant concentrations within streams in Atlanta, GA—which placed first out of 18 metropolitan cities for urban tree cover in 2008—were lower than estimated median stormwater concentrations across the United States (Giarrusso and Smith 2014; Pitt et al. 2018). Much of the subcatchment areas of the intensive study wetlands contained well-established neighborhoods dating from immediately after World War II and included large trees, cumulatively extensive unfertilized lawns, and large amounts of buffer vegetation along tributaries and the mainstem of Walnut Creek. These relatively low levels of contamination were most likely attributable to the upland subcatchments' effectiveness at cleaning stormwater, and perhaps a lack of substantial parameter input from subcatchment surfaces. With the exception of the HR wetland, large amounts of impervious surfaces were generally absent from the catchment areas of the intensive study wetlands (Table 1). Below-ground leakage was also not apparent in the study site surface waters.

Table 7. Urban surface water contaminant levels reported. Asterisks (*) in first column denote means reported; otherwise values are medians. TSS = total suspended solids; TP = total phosphorus; TN = total nitrogen; TKN = total Kjeldahl nitrogen; ND = non-detect.

Source	TSS (mg/L)	TP (mg/L)	TN (mg/L)	Nitrogen Components			Copper (Cu) (µg/L)	Lead (Pb) (µg/L)	Zinc (Zn) (µg/L)
				TKN (mg/L)	Ammonia (NH_3) (mg/L)	Nitrate + Nitrite (NO_3^- + NO_2^-) (mg/L)			
Stormwater in 21 urbanized areas, 17 states (Masoner et al. 2019)		.092 (.778 max)					5.5 (68 max)	0.39 (2.1 max)	15 (189 max)
Stormwater from over 200 municipalities in the US (Pitt et al. 2018)	58	0.236		1.4	0.3	0.6	11.3	6.0	82
Urban stormwater across the US (Smullen et al. 1999)*	54.5	0.26	2.00	1.47		0.53	11.1	50.7	129

Source	TSS (mg/L)	TP (mg/L)	TN (mg/L)	Nitrogen Components			Copper (Cu) (µg/L)	Lead (Pb) (µg/L)	Zinc (Zn) (µg/L)
				TKN (mg/L)	Ammonia (NH_3) (mg/L)	Nitrate + Nitrite ($NO_3^- + NO_2^-$) (mg/L)			
Surface runoff at 10 urban sites in Atlanta, GA (Horowitz et al. 2008)		0.039 (max 0.04 to 3.2)	1.98 (max 1.6 to 18)				6.2 (max 7.1 to 110)	0.77 (max 3.1 to 13)	2.64 (max 12 to 1300)
Review of international stormwater data (Pamuru et al. 2022)*	104	0.35	2.08				45.3	22.3	144
This study: 4 urban streams (inlets) during storms	16.5	0.18	1.0	0.67	0.07	0.35	4.45	2.6	18.5

The few high outlier concentrations observed in the study wetlands were most likely attributable to the accumulation of algal biomass, which bioconcentrates metals and creates TSS. High concentrations at outlets were most likely attributable to fine sediment accumulation when beavers dammed the outlets. Below are some examples of high concentrations found in the literature and how the results of this study compare:

- Total Phosphorus: about 0.10 mg/L TP is considered low; 10 mg/L high for warm climate constructed wetlands (Kadlec and Wallace 2009; Masoner et al. 2019). Maximum TP in this study (74 mg/L; June 2020) was from ZH center with the next highest value at 13.7 mg/L (also ZH center; Feb. 2020). Median TP from all sampling in the natural urban wetlands in this study was 0.16 mg/L.
- TKN: 5 mg/L is considered the high threshold for TKN in agricultural communities in Europe (Marecos do Monte and Albuquerque 2010) and 10 mg/L for the US (US EPA 2016b). Maximum TKN in this study (91 mg/L; June 2020) was from ZH center with the next highest value at 11.0 mg/L (also ZH center; Feb. 2020). Median TKN from all sampling in the natural urban wetlands in this study was 0.70 mg/L.
- Ammonia: 2.05 to 2.3 mg/L was a level of concern for ammonia in Chicod Creek, North Carolina (O'Rear 1975), while 6.67 mg/L is considered a chronic hazard at pH 6.5 by the Iowa Administrative Code, Chapter 61 (Sawyer 2008). Maximum ammonia

concentration found in this study (1.2 mg/L; June 2020) was from ZH center with the next highest value at 0.78 mg/L (HR inlet; Feb. 2020). Median ammonia concentration from all sampling in the natural urban wetlands in this study was 0.10 mg/L.

- Copper and zinc: 40 µg/L copper and 710 µg/L zinc were high contamination levels observed in light industrial zones in the Haw River basin (Shuman et al. 1977). Maximum copper concentration in this study (540 µg/L; June 2020) was from ZH center with the next highest value at 91 µg/L (HR outlet; June 2020). Maximum total zinc concentrations in this study (2900 and 460 µg/L) were from ZH center and HR outlet in June 2020; the next highest value was 310 µg/L (ZH center; Dec. 2019). The median concentrations from all sampling in the natural urban wetlands in this study were 3.4 µg/L for copper and 17 µg/L for zinc.
- Lead: High reported values of lead in stormwater were from urban highways at 400 µg/L and from heavy industrial sites at 290 µg/L (Claytor and Schueler 1996). Maximum total lead concentrations in this study (710 and 300 µg/L) were from ZH center and HR outlet in June 2020; the next highest value was 69 µg/L (ZH center; Dec. 2019). Median total lead concentration from all sampling in the natural urban study wetlands was 2.7 µg/L.

4.3.2. Comparison to Regulatory Standards

All nine of the parameters of interest in this study—TKN, nitrate + nitrite, ammonia, oil and grease, TSS, total phosphorus, copper, lead, and zinc—are naturally occurring elements and compounds (except for the possibility of some unknown fraction of total suspended solids), and hence can be found in “background” levels in the wetland and stream waters monitored, as noted in the previous section.

Regulatory standards and non-regulatory thresholds for these substances usually revolve around two major concerns: human health and aquatic non-human biotic health (Table 8). The US EPA lists human drinking water thresholds for nitrate-nitrogen at 10 mg/L and nitrite at 1 mg/L (maximum contaminant levels), 1300 µg/L copper and 15 µg/L lead (treatment technology action levels), and 2000 µg/L zinc (lifetime health advisory level) (US EPA 2018). EPA also lists “Criterion Continuous Concentration (CCC)” or concentrations at which an aquatic community can be exposed indefinitely without injurious effect. The CCC levels at a hardness of 100 mg/L are 2.5 µg/L for lead, and 120 µg/L for zinc. Hardness in the wetland streams in the study ranged from 28 to 120 mg/L, with a median of 61 mg/L.

In North Carolina, regulatory standards vary according to the classification of surface water considered, which is based on the designated uses of that body of water. In Class C waters, secondary recreation and biological integrity—including fish and other aquatic life—are protected. For Class C waters, surface water standards for lead, zinc, and copper are calculated for dissolved metals based on hardness. Other NC fresh surface water classifications add further protections for primary recreation (e.g., swimming; Class B) and drinking water sources (Classes WS-I to WS-V). Supplemental classifications further add or modify standards in specific surface waters, such as increasing the minimum dissolved oxygen standard in designated trout (Tr) waters (NC DEQ DWR 2023). In NC, as in other states, National Pollutant Discharge Elimination System (NPDES) permitted facilities have site-specific limits for applicable parameters in discharges to ensure that surface water standards are met in receiving streams.

Regulatory bodies in other parts of the US (e.g., the New Mexico Environment Department) have promulgated similar surface water standards for the protection of aquatic life and drinking water supplies (New Mexico Administrative Code 20.6.4.900).

Median concentrations of nitrogen species, copper, and zinc in the inlets and outlets of the four intensive study wetlands were below US EPA drinking water standards as well as NC DWR Class C surface water standards (Table 8). Combined nitrate + nitrite concentrations in the wetland inlets (median: 0.46 mg/L; mean: 0.52 mg/L) and outlets (median: 0.03 mg/L; mean: 0.09 mg/L) were far below the individual NC DWR drinking water standards for nitrate (10 mg/L) and nitrite (1 mg/L). Copper (inlet median: 3.1 µL; outlet median: 4.0 µL) and zinc (inlet median: 16 µL; outlet median: 18 µL) concentrations tended to be below NC DWR Class C water standards, with the exception of a few outlier samples with particularly high concentrations. Lead concentrations (inlet median: 2.0 µL /non-detect; outlet median: 3.2 µL) were slightly higher. However, analyses for this study measured total metals concentrations, and NC hardness-based standards are calculated for dissolved metals, so direct comparison was not possible.

Table 8. Thresholds and/or regulatory standards for various substances and water types. Water quality standards are shown as maximum values unless otherwise specified.

Criterion	Nitrogen Species	Copper (Cu)	Lead (Pb)	Zinc (Zn)	Turbidity	Other
US EPA Drinking Water Maximum Contaminant Level (MCL), Treatment Technology Action Level, or Health Advisory Level	Nitrate: 10 mg/L; Nitrite: 1 mg/L	1300 µg/L	15 µg/L	2000 µg/L		
US EPA Criterion Continuous Concentration[1] (CCC, at 100 mg/L hardness)			2.5 µg/L	120 µg/L		
NC DWR Class C waters (standards apply to all NC fresh surface waters)[2]		At 61 mg/L hardness, WQS for chronic exposure is 5.9 µg/L dissolved Cu; at 100 mg/L hardness, chronic WQS is 9.0 µg/L	At 61 mg/L hardness, WQS for chronic exposure is 1.5 µg/L dissolved Pb; at 100 mg/L hardness, chronic WQS is 2.5 µg/L	At 61 mg/L hardness, WQS for chronic exposure is 77.7 µg/L dissolved Zn; at 100 mg/L hardness, chronic WQS is 118.1 µg/L	50 NTU (Nephelometric Turbidity Units) (non-trout waters)	Total arsenic: 10 µg/L; pH between 6.0 and 9.0 except swamp waters may be as low as 4.3 if due to natural conditions
NC DWR Drinking Water Supply surface water standards or in-stream target values	Nitrate: 10 mg/L; Nitrite: 1 mg/L				TDS: 500 mg/L	Chloride: 250 mg/L; Hardness: 100 mg/L
NC DWR High Quality Waters (wastewater effluent limits)	Ammonia: 2 mg/L				TSS: 10 mg/L for trout waters; 20 mg/L for non-trout waters	DO > 6 mg/L
New Mexico Environment Department surface water standards (NMAC 20.6.4.900)	Nitrate as N: 10 mg/L in drinking water supplies	At 100 mg/L hardness, chronic WQS is 8.96 µg/L dissolved Cu	At 100 mg/L hardness, chronic WQS is 2.52 µg/L dissolved Pb	At 100 mg/L hardness, chronic WQS is 121 µg/L dissolved Zn		

[1] US EPA CCC is the concentration at which an aquatic community can be exposed indefinitely without injurious effect.

[2] Median hardness for the intensive study wetland inlets and outlets was 61 mg/L $CaCO_3$. NC surface water quality standards (WQS) for dissolved metals in fresh waters increase with increasing hardness according to state regulatory formulas.

The intensive study wetlands used in this project included large areas (acres) of standing water to some depth (maxima to approx. 3.3 feet or 1 meter). Therefore, limnological thresholds for eutrophication were also considered (Table 9). Traditional limnological investigations have used the Trophic State Index (TSI), a semi-quantitative water quality scale originating from Sweden used to categorize water bodies as oligotrophic, mesotrophic, eutrophic, or hypereutrophic. The TSI can be calculated using one of three variables: chlorophyll-a, total phosphorus, or water clarity as defined by Secchi disk depth. Though based initially on characteristics of only Northern European and American temperate and sub-arctic lakes, the TSI is still used in modified forms for other surface water provenances around the world. The most widely considered US version sets the possibility of algal blooms and other submerged aquatic vegetation (SAV) problems (e.g., “scums”, bottom anoxia) occurring at total phosphorus levels above 0.048 mg/L, but possibly as low as 0.024 (Carlson 1977). This level is labelled eutrophic on the TSI scale. Burkholder (1992) considered Lake Raleigh (with total phosphorus ranging from 0.035 to 0.075 mg/L) to be mesotrophic or moderately clear most of the summer but on the edge of becoming eutrophic with increased nutrients. The USGS, in its 1992 NC stream report (Caldwell 1992), cited several authors (Sawyer 1947; Sakamoto 1966; Vollenweider 1971) who considered eutrophication conditions in streams and lakes to begin at levels above 0.3 mg/L total nitrogen and 0.1 mg/L total phosphorus. Dodds et al. (1998) modified the TSI to apply to stream environments. They considered the mesotrophic to eutrophic boundary in streams to be 0.075 mg/L for total phosphorus and 1.5 mg/L for total nitrogen. Scientists at NC DWR have used standards of total phosphorus at 0.04 mg/L and TKN at 0.5 mg/L to define the boundary between mesotrophic and eutrophic in NC lakes (Vander Borgh, M. pers. comm. 2022). By NC DWR standards for NC lakes, the median total phosphorus and TKN concentrations across all intensive study wetlands samples were indicative of a eutrophic state at wetland inlets (TP: 0.14 mg/L; TKN: 0.6 mg/L), centers (TP: 0.26 mg/L; TKN: 1.1 mg/L), and outlets (TP: 0.12 mg/L; TKN: 0.7 mg/L).

Table 9. Comparison of thresholds for identifying trophic states in lakes and streams.

Source	Trophic State	Location/Type of Water Body	Total Phosphorus Level
Burkholder 1992	Mesotrophic/ Eutrophic boundary	Lake Raleigh, Raleigh, NC	0.035 to 0.075 mg/L
Carlson 1977	Eutrophic (algal bloom/SAV problems)	Northern temperate to subarctic lakes	Above 0.048 mg/L but possibly as low as 0.024 mg/L
Caldwell 1992 (multiple sources)	Eutrophic	North Carolina streams and lakes	Above 0.1 mg/L (and 0.3 mg/L total nitrogen)
Dodds et al. 1998	Mesotrophic/ Eutrophic boundary	Streams	0.075 mg/L (and 1.5 mg/L total nitrogen)
NC Division of Water Resources scientists	Mesotrophic/ Eutrophic boundary	North Carolina lakes	0.04 mg/L (TKN 0.5 mg/L)

4.4. Challenges in Studying Water Quality in Natural Wetlands in Urbanized Watersheds

One inherent challenge in studying wetlands is the complexity and variability in the interaction of environmental parameters. There are many factors influencing how wetlands interact with inflowing water, including watershed and site-specific background concentrations, vegetation types and coverage, local climate and seasonality, geology, internal chemical and stochastic environments in both the sediments and standing water, internal hydraulic regime including structures and bathymetry, hydraulic retention time and flow rates, wildlife use, microbial communities, area and depth of the wetland, carbon availability, the levels and variability of concentrations and species of incoming contaminants, and others.

In addition to this inherent complexity, geochemical and hydrodynamic long-term research on urban wetlands in Raleigh was very difficult due to a host of human and beaver impingements, including flashy behavior of Walnut Creek and its tributaries with overbank floods of the mainstem sometimes reaching several feet above wetland surfaces. Wetland site structures were impacted by an unexpected factor; a sizeable beaver population throughout the study area made major and random (from a human perspective) changes to the hydrodynamics in all four intensive study wetlands, especially toward the last third of the data collection period. Dam construction occurred in the tributary channels, sometimes dramatically changing surface water flow direction, strength, and consistency across wetland interiors and streams between data collection site visits. The dynamic between standing water levels and standing wetland vegetation—crucial in sequestering suspended material in influent—was rapidly and drastically changed by beaver activity.

Several experiences with this study highlighted the challenges of working in an urban setting where ongoing human activity constantly affects natural areas. Additional impingements included unexpected human mechanical interference redirecting water and destroying monitoring equipment, changes to the geomorphology and groundwater topography due to construction and impervious surfaces, and the ephemeral, fragile nature of many wetland tributaries to the mainstem of Walnut Creek. These factors contributed to the small number of wetland sites that were ultimately selected as usable for the study, mainly in terms of viability of perennial stream flow through a given wetland into Walnut Creek.

Other challenges to analysis included the small sample size of storm events that satisfied the antecedent dry period criterion and the limited ability to sample per event because grab sampling was required (instead of the desired multiple automatic samples over the course of individual storms). Grab sampling meant heavily relying on weather forecasts to time sampling events, and frequently weather forecasts were inaccurate in terms of the amount and location of rain predicted, especially in this area where thunderstorms can be very localized. Staff availability and the COVID-19 pandemic also hampered efforts to collect data.

Additionally, new construction of a large school and associated impervious parking and driving areas adjacent to ZH with a culvert directing stormwater into the inlet stream was completed at the end of the study period. This new stormwater input could have contributed sediment during construction and oil and grease, nutrients, etc. after construction. Significant site structural changes were also brought on by unanticipated human activity in one of the intensive study wetlands (WP site) toward the end of data collection for this project which made further scientific data collection at the site impossible. While we can report on what occurred during the study period at these wetlands, the nature of weather patterns, unanticipated human activities, and animal activity (particularly beavers) make it difficult to predict how a natural urban wetland will perform in the future.

4.5. Major Findings

- Baseflow and storm event concentrations in wetland inlet and outlet streams at all sites were low compared to typical stormwater concentrations reported in the literature for the US.
- The relatively low levels of contamination in the study sites' influent water at baseflow and during storms were most likely attributable to the upland subcatchments' effectiveness at cleaning stormwater, along with the apparent lack of substantial parameter input from subcatchment surfaces. These factors may be due to relatively lower impervious surface areas and denser vegetated buffers in the study area than those in many urbanized cities.
- Observed high concentrations of metals at the centers of the intensive study wetlands relative to inlets are likely from the accretion and holding of metal-carrying sediment by vegetation and the bioabsorption and bioaccumulation (die-off) of in-situ algal colonies in the wetland centers. Legacy contamination of lead, copper, and zinc exists in the urban study wetlands, but mostly appears bound to the wetlands' soils.
- The wetlands were not generally contributing appreciable pollutants to the mainstem of Walnut Creek, because outlet concentrations were low and the flow rates of water discharged into the Creek from the wetlands were low relative to the typical flow of the Creek, leading to a dilution of contaminants leaving the wetlands in the Creek.
- The study wetlands showed significant trapping of sediment and macro-pollution (plastic trash – not measured; Appendix A: Figure A-2) which has been kept out of the lower Walnut Creek basin and hence the greater Neuse River Basin.

The findings of this project cannot determine the approximate concentration thresholds of urban contaminants that, when exceeded, the natural wetlands of Raleigh would become overwhelmed and unable to keep these contaminants from entering Walnut Creek. Therefore, it is not possible to directly compare stormwater uplift capability of these wetlands with operating constructed stormwater BMPs for a similar suite of contaminants. However, it is clear that Raleigh City wetlands are performing invaluable ecological and water-quality-enhancing services to the City without any direct or indirect cost, particularly for nutrients and sediment. Nitrate + nitrite in waters, many cubic yards of sediment, high amounts of heavy metals in the soils, and large quantities of trash are being kept from the lower Walnut Creek watershed and by extension the Neuse River Basin by the self-maintaining, trapping action of these forested marshes and ponds. Further, this study gives strong on-the-ground evidence of the effectiveness and necessity of maintained stream and wetland buffers. Developing and infilling parcels in downtowns of cities is often desired, but it is critical that planners and developers carefully avoid disturbing established buffers that enable contamination capture.

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APPENDICES

Appendix A

Supplementary Photographs

Figure A-1. Example of beaver dam at wetland site (APH center).

Figure A-2. Trash being trapped in wetland (ZH).

Appendix B

Rapid Assessment Results

Table A-1. NC Wetland Assessment Method (NCWAM) results and Ohio Rapid Assessment Method (ORAM) results.

Wetland Site	NCWAM Rating	ORAM Score (out of 90)
APH	Medium	71
HR	Medium	70.5
WP	Low	65
ZH	Medium	67.5

Rapid assessments were performed using the NC Wetland Assessment Method (NCWAM) and the Ohio Rapid Assessment Method (ORAM) for each of the four wetland study sites. The NCWAM is focused on various water-oriented functions of wetlands, particularly in relation to water storage and water filtration capacity. It also considers several habitat variables, such as wetland size, connectivity, level of alteration, and diversity, composition, and structure of the vegetation.

The ORAM is a rapid assessment method for wetland quality that holistically takes into account wetland size, surrounding land use, hydrology variables, habitat alteration, and plant community information.

Appendix C

Water Chemistry Medians and Ranges

Table A-2. Parameter median and range values for baseflow water quality in four urban wetland inlets (I), centers (C), and outlets (O); ND = non-detect.

Collection Location	Total Suspended Solids (TSS) (mg/L)	Total Phosphorus (TP) (mg/L)	Ammonia (NH_3) (mg/L)	Nitrate + Nitrite (NO_3^- + NO_2^-) (mg/L)	Total Kjeldahl Nitrogen (TKN) (mg/L)
APH-I	6.2/ND (6.2/ND to 130)	0.08 (0.04 to 0.37)	0.11 (0.03 to 0.52)	0.32 (0.02/ND to 0.87)	0.47 (0.36 to 0.86)
APH-C	23 (12 to 370)	0.19 (0.14 to 0.33)	0.31 (0.2 to 0.56)	0.05 (0.02/ND to 0.37)	0.88 (0.72 to 1.2)
APH-O	14 (6.2/ND to 350)	0.20 (0.09 to 0.78)	0.28 (0.18 to 0.64)	0.07 (0.02/ND to 0.42)	0.96 (0.48 to 1.8)
HR-I	28 (12 to 396)	0.15 (0.07 to 1.4)	0.24 (0.12 to 0.76)	1.20 (0.03 to 2.2)	0.76 (0.53 to 3.4)
HR-C	59 (19 to 2,400)	0.145 (0.07 to 1)	0.06 (0.02/ND to 0.28)	0.02/ND (0.02/ND to 0.2)	0.99 (0.44 to 4.1)
HR-O	13.5 (6.2/ND to 2,050)	0.07 (0.02/ND to 0.83)	0.08 (0.03 to 0.24)	0.05 (0.02/ND to 0.12)	0.40 (0.2/ND to 1.8)
WP-I	22 (6.2/ND to 176)	0.18 (0.1 to 0.76)	0.03 (0.02/ND to 0.24)	0.57 (0.1 to 0.71)	0.35 (0.31 to 1.1)
WP-C	54.5 (15 to 631)	0.24 (0.07 to 0.68)	0.03 (0.02/ND to 0.22)	0.02/ND (0.02/ND to 0.69)	1.30 (0.68 to 3.3)
WP-O	33.5 (13 to 638)	0.20 (0.13 to 0.66)	0.14 (0.02/ND to 0.39)	0.02/ND (0.02/ND to 0.2)	0.94 (0.6 to 1.6)
ZH-I	7.6 (6.2/ND to 27)	0.09 (0.05 to 0.16)	0.12 (0.02/ND to 0.29)	0.52 (0.46 to 0.69)	0.55 (0.35 to 0.86)
ZH-C	306 (27 to 24,600)	1.45 (0.26 to 74)	0.085 (0.02/ND to 1.2)	0.02/ND (0.02/ND to 0.2)	1.70 (0.59 to 91)
ZH-O	38 (6.3 to 94)	0.09 (0.03 to 0.28)	0.06 (0.03 to 0.2)	0.02/ND (0.02/ND to 0.1)	0.61 (0.3 to 0.99)

Collection Location	Oil and Grease (mg/L)	Copper (Cu) (µg/L)	Lead (Pb) (µg/L)	Zinc (Zn) (µg/L)
APH-I	10/ND (10/ND to 15)	2/ND (2/ND to 5.9)	2/ND (2/ND to 9.6)	10/ND (10/ND to 33)
APH-C	10/ND (10/ND to 12)	2.1 (2/ND to 3.7)	3.1 (2/ND to 7.9)	12 (10/ND to 19)
APH-O	10/ND (10/ND to 12)	2/ND (2/ND to 16)	2/ND (2/ND to 32)	10/ND (10/ND to 69)
HR-I	10/ND (10/ND to 13)	3.2 (2/ND to 13)	2/ND (2/ND to 12)	33 (16 to 95)
HR-C	10/ND (10/ND to 12)	7.5 (2.1 to 21)	5.8 (2/ND to 39)	40 (14 to 150)
HR-O	10/ND (10/ND to 12)	7.65 (2/ND to 91)	4.25 (2/ND to 300)	22.5 (18 to 460)
WP-I	10/ND (10/ND to 14)	2.25 (2/ND to 24)	2/ND (2/ND to 19)	10/ND (10/ND to 83)
WP-C	10/ND (10/ND to 13)	6.65 (2.5 to 14)	12.5 (4.2/ND to 45)	40.5 (10/ND to 97)
WP-O	10/ND (10/ND to 12)	6.2 (2/ND to 39)	7.45 (3 to 59)	21 (10/ND to 130)
ZH-I	10/ND (10/ND to 15)	2/ND (2/ND to 3.5)	2/ND (2/ND to 6)	10/ND (10/ND to 22)
ZH-C	10/ND (10/ND to 42)	4.7 (2/ND to 540)	7.8 (2/ND to 710)	35 (10/ND to 2,900)
ZH-O	10/ND (10/ND to 12)	2.8 (2/ND to 7)	2.7 (2/ND to 7.5)	10/ND (10/ND to 22)

Collection Location	pH	Dissolved Oxygen (DO) (%)	Dissolved Oxygen (DO) (mg/L)	Specific Conductivity (µS/cm)	Water Temp (°C)
APH-I	7.15 (6.85 to 7.48)	90.5 (38 to 106.5)	9.96 (3.01 to 11.38)	231.7 (153 to 310.3)	16.4 (5.98 to 24.9)
APH-C	6.85 (6.68 to 7.23)	69.3 (8.8 to 97.7)	5.49 (0.74 to 9.20)	219.7 (185 to 300)	18.0 (6.55 to 27.1)
APH-O	6.96 (6.73 to 7.26)	68.7 (46.3 to 91.4)	7.38 (3.91 to 8.93)	211.3 (92.4 to 300.5)	18.1 (4.16 to 26.6)
HR-I	6.56 (6.10 to 6.87)	36.7 (5.8 to 63.1)	3.27 (0.70 to 7.57)	361.0 (125 to 716.1)	14.1 (4.9 to 22.6)
HR-C	6.43 (6.06 to 6.76)	27.3 (1.4 to 76)	3.35 (0.12 to 10.08)	261.3 (125.8 to 539.9)	10.7 (3.2 to 22.58)
HR-O	6.49 (5.97 to 6.81)	33.05 (15.1 to 41.5)	3.39 (1.66 to 5.22)	245.4 (124.3 to 881.5)	10.75 (4.62 to 22.31)
WP-I	7.17 (6.66 to 7.53)	83.7 (75.8 to 99.4)	9.55 (6.40 to 11.42)	202.0 (181.9 to 325.8)	14.7 (8.04 to 25.2)
WP-C	6.69 (5.87 to 6.99)	32.0 (12.5 to 55.7)	3.68 (1.04 to 6.01)	132.7 (80.2 to 236.3)	13.9 (2.9 to 28.3)
WP-O	6.78 (6.66 to 6.94)	36.3 (0.6 to 102.6)	5.57 (0.83 to 11.03)	149.2 (110.5 to 262.2)	15.05 (7.38 to 24.3)
ZH-I	7.01 (6.03 to 7.22)	63.5 (39.5 to 81.3)	6.10 (3.46 to 9.80)	312.0 (260 to 472.6)	14.2 (7.61 to 23.12)
ZH-C	6.66 (5.94 to 7.36)	9.9 (0.1 to 54.2)	1.18 (0.01 to 7.10)	289.4 (118 to 668.5)	14.8 (3.65 to 25.01)
ZH-O	6.45 (5.86 to 7.3)	40.4 (2.9 to 76)	4.40 (0.25 to 7.84)	217.2 (132.3 to 489.1)	11.6 (2.01 to 25.73)

Table A-3. Parameter median and range values for stormflow water quality in four urban wetland inlets (I), centers (C), and outlets (O); ND = non-detect

Collection Location	Total Suspended Solids (TSS) (mg/L)	Total Phosphorus (TP) (mg/L)	Ammonia (NH_3) (mg/L)	Nitrate + Nitrite (NO_3^- + NO_2^-) (mg/L)	Total Kjeldahl Nitrogen (TKN) (mg/L)
APH-I	14 (6.2/ND to 127)	0.14 (0.05 to 0.24)	0.06 (0.04 to 0.16)	0.30 (0.20 to 0.63)	0.68 (0.33 to 0.93)
APH-C	36 (32 to 40)	0.21 (0.15 to 0.26)	0.17 (0.05 to 0.28)	0.15 (0.06 to 0.24)	1.16 (0.92 to 1.4)
APH-O	27 (12 to 331)	0.18 (0.14 to 0.4)	0.22 (0.12 to 0.32)	0.16 (0.10 to 0.16)	0.82 (0.71 to 1.1)
HR-I	18 (11 to 69)	0.22 (0.1 to 0.39)	0.12 (0.06 to 0.48)	0.19 (0.13 to 1.1)	0.46 (0.24 to 1.6)
HR-C	8.6 (7.2 to 10)	0.07 (0.03 to 0.11)	0.02/ND (0.02 to 0.02)	0.03 (0.02/ND to 0.03)	0.43 (0.33 to 0.52)
HR-O	13.9 (6.2/ND to 61)	0.08 (0.04 to 0.21)	0.03 (0.02/ND to 0.06)	0.16 (0.04 to 0.44)	0.46 (0.26 to 0.81)
WP-I	29 (8 to 169)	0.33 (0.16 to 1.2)	0.04 (0.02/ND to 0.1)	0.37 (0.21 to 0.46)	0.60 (0.36 to 2.1)
WP-C	140.5 (35 to 246)	0.41 (0.37 to 0.45)	0.20 (0.02/ND to 0.37)	0.02/ND (0.02 to 0.02)	2.90 (2.70 to 3.1)
WP-O	71 (22 to 206)	0.24 (0.08 to 0.41)	0.14 (0.02/ND to 0.26)	0.02/ND (0.02 to 0.02)	1.12 (0.54 to 1.5)
ZH-I	8 (6.5 to 25)	0.15 (0.13 to 0.24)	0.09 (0.06 to 0.18)	0.35 (0.16 to 0.6)	0.80 (0.65 to 0.85)
ZH-C	138.5 (82 to 195)	0.92 (0.44 to 1.4)	0.05 (0.04 to 0.06)	0.02/ND (0.02 to 0.02)	1.65 (0.70 to 2.6)
ZH-O	22 (11 to 51)	0.10 (0.07 to 0.21)	0.03 (0.02/ND to 0.04)	0.06 (0.02/ND to 0.27)	0.62 (0.37 to 0.72)

Collection Location	Oil and Grease (mg/L)	Copper (Cu) (µg/L)	Lead (Pb) (µg/L)	Zinc (Zn) (µg/L)
APH-I	10/ND (10/ND to 10)	4.9 (2/ND to 8.6)	4 (2/ND to 14)	17 (10/ND to 40)
APH-C	10/ND (10/ND to 10)	3.9 (3.9 to 4)	4.4 (4.1 to 4.7)	13 (12 to 14)
APH-O	10/ND (10/ND to 10)	2.9 (2.4 to 7.9)	3.2 (2/ND to 13)	10.5 (10/ND to 34)
HR-I	10/ND (10/ND to 10)	6.4 (2.9 to 8.7)	2.7 (2/ND to 3.9)	36 (16 to 120)
HR-C	10/ND (10/ND to 10)	3.6 (3.1 to 4)	2.2 (2/ND to 2.4)	26.5 (25 to 28)
HR-O	10/ND (10/ND to 10)	5.7 (3.1 to 8.1)	2.8 (2/ND to 14)	18.5 (16 to 30)
WP-I	10/ND (10/ND to 10)	4.1 (2/ND to 4.3)	3.3 (2/ND to 6.1)	13 (10/ND to 19)
WP-C	10/ND (10/ND to 10)	5.3 (2/ND to 8.6)	10.0 (2/ND to 18)	35.5 (13 to 58)
WP-O	10/ND (10/ND to 10)	12.0 (4.5 to 13)	13.5 (5.6 to 35)	45 (15 to 170)
ZH-I	10/ND (10/ND to 10)	6.7 (3.1 to 8.9)	2/ND (2/ND to 2.5)	23 (18 to 33)
ZH-C	10/ND (10/ND to 10)	6.5 (2/ND to 11)	12 (2/ND to 22)	45.5 (12 to 79)
ZH-O	10/ND (10/ND to 10)	3.6 (2.5 to 7.2)	2/ND (2/ND to 7.7)	10.5 (10/ND to 18)

Collection Location	pH	Dissolved Oxygen (DO) (%)	Dissolved Oxygen (DO) (mg/L)	Specific Conductivity (µS/cm)	Water Temp (°C)
APH-I	7.01 (6.40 to 7.34)	79.8 (69.5 to 88.3)	6.94 (5.72 to 10.54)	152.5 (112.4 to 219.7)	21.2 (7.5 to 26)
APH-C	7.00 (6.85 to 7.15)	70.45 (50.2 to 90.7)	5.58 (4.13 to 7.03)	133.2 (131.8 to 134.6)	26.5 (24.8 to 28.1)
APH-O	6.85 (6.67 to 6.92)	61.0 (55.1 to 87.7)	5.14 (4.48 to 10.69)	138.2 (80.9 to 156.0)	23.2 (6.7 to 27.2)
HR-I	7.05 (6.73 to 8.50)	45.8 (35.3 to 97.2)	4.31 (2.92 to 11.67)	103.7 (35.2 to 277.5)	23.1 (7.1 to 24.9)
HR-C	6.51 (6.38 to 6.63)	15.75 (6.2 to 25.3)	1.31 (0.53 to 2.09)	176.3 (164.8 to 187.7)	23.5 (22.5 to 24.5)
HR-O	6.63 (6.52 to 6.82)	50.8 (34.8 to 79.5)	4.41 (2.94 to 9.61)	183.4 (111.4 to 208.3)	22.2 (6.8 to 24)
WP-I	7.09 (6.79 to 7.47)	82.1 (71.2 to 99.0)	7.63 (5.73 to 11.97)	181.0 (47.9 to 183.5)	22.9 (6.9 to 26.4)
WP-C	6.42 (6.32 to 6.51)	25.35 (22.7 to 28)	2.08 (1.81 to 2.35)	138.5 (137.1 to 139.9)	25.9 (24.6 to 27.1)
WP-O	6.52 (5.83 to 6.81)	27.05 (15.8 to 78.6)	2.32 (1.28 to 9.87)	125.6 (73.3 to 175.2)	22.8 (5.5 to 25.8)
ZH-I	6.94 (6.81 to 7.10)	40.3 (13.6 to 93.7)	3.77 (1.14 to 11.08)	159.1 (59.5 to 287.5)	22.7 (7.7 to 24.4)
ZH-C	6.57 (6.55 to 6.58)	10.5 (6.0 to 15.0)	0.87 (0.52 to 1.22)	216.4 (130.9 to 301.9)	24.3 (23 to 25.6)
ZH-O	6.58 (6.41 to 6.85)	41.7 (35.6 to 77.3)	3.49 (3.00 to 9.50)	149.3 (89.1 to 190.9)	22.2 (6.2 to 24.4)

Appendix D

Hydrographs for Intensive Study Sites

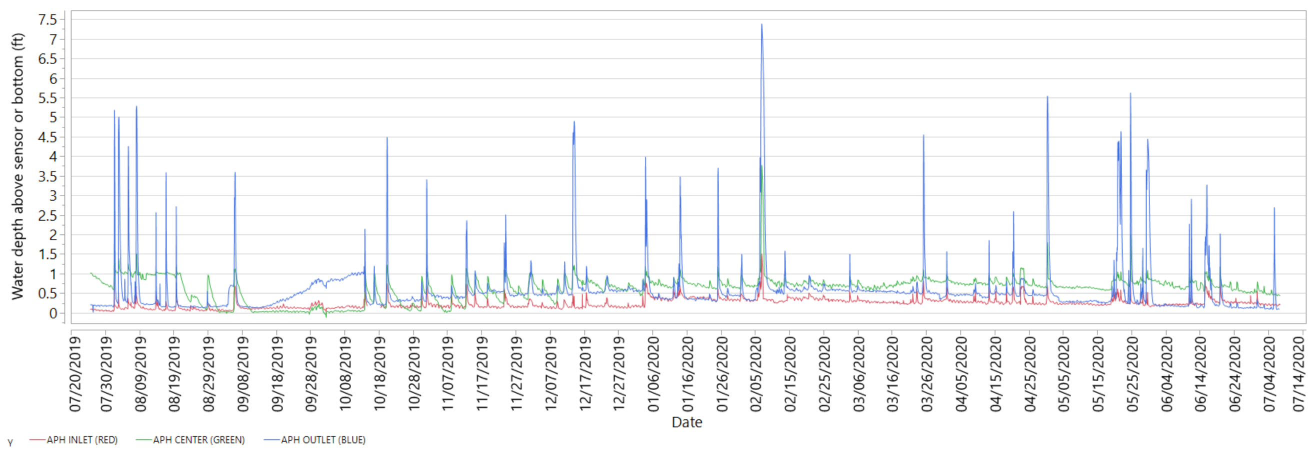


Figure A-3. Hydrograph for Apollo Heights inlet (red), center (green), and outlet (blue) July 25, 2019 through July 07, 2020.

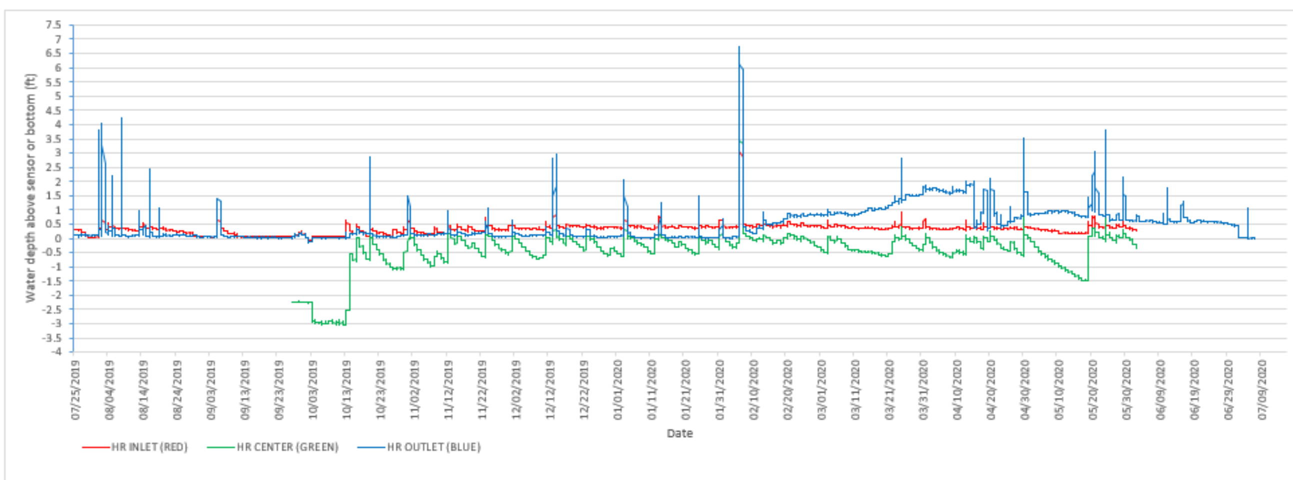


Figure A-4. Hydrograph for Hammond Road inlet (red), center (green), and outlet (blue) July 25, 2019 through July 07, 2020.

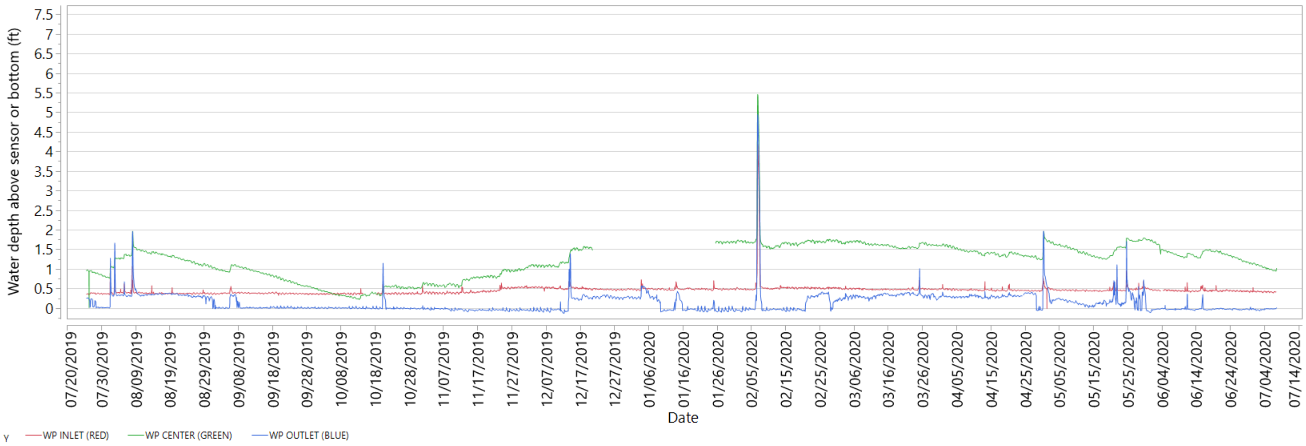


Figure A-5. Hydrograph for Women's Prison inlet (red), center (green), and outlet (blue) July 25, 2019 through July 07, 2020.

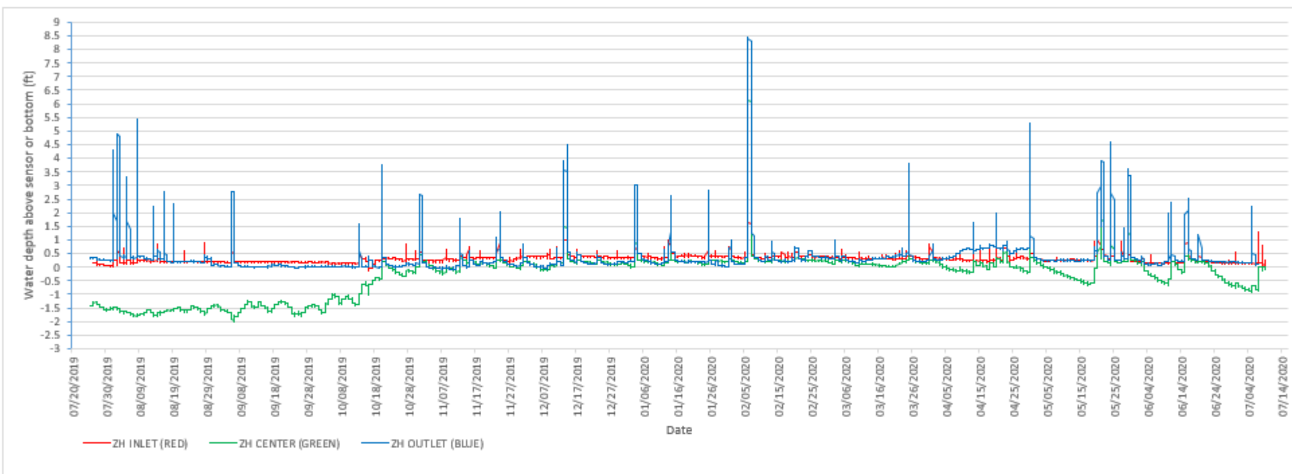


Figure A-6. Hydrograph for Zhang inlet (red), center (green), and outlet (blue) July 25, 2019 through July 07, 2020.

Continuous water level data were obtained from July 25, 2019 through July 7, 2020 for each wetland. Gaps in the data for WP center from December 2019 to January 2020 and at the end of the timeframe for center and inlet at HR were caused by equipment malfunction.

Peak water levels at our wetland outlets ranged from 4.75 feet above base of the stream outlet (WP) to 8.5 feet above base of stream outlet (ZH) during a February 2020 storm. Flooding events in the wetlands represented substantial water coming into the wetlands from

Walnut Creek (at WP) and also high water levels in Walnut Creek preventing stormflow from leaving some wetlands (at APH, HR, ZH). Water level sensors showed water levels rising in the outlets before they rose in the inlets, indicating that water was coming into the wetlands from Walnut Creek during larger storms.

Appendix E

Sediment Input Tracking Calculations for ZH Wetland

ZH Sediment Delta Tracking

Change at stake base (cm) per inspection date

(established Feb 18 2019)

Delta area = 952 sq. meters/0.24 acres

Station I.D.	Lat	Long.	Storm	Feb 25 2019	May 20 2019	July 26 2019		Oct 18 2019			Nov 26 2019			Jan 8 2020		Feb 18 2020		July 22 2020	Sept 10 2020	Oct 15 2020	Nov 25 2020	April 5 2021	March 15 2022
			Note	2.4 in rain after 5 days	0.4 in since May 12	4 storms totalling 1.8 in since June 10		2 storms @ 1.2 in	Note		0.9 in storm two days before	Note		1.4 in past 7 days	Note	about 4 in rain past 12 days							
Delta Edge D.E. 1	35.761432	-78.621631	small trib. center	0	4	0		3.8	small trib		1	very slurried		1	very slurried	6		4.75	3.75	-2	0	2	1
D.E. 2	35.756225	-78.621732	next to bucket	0	4	1		0	bucket floated 30 ft to NE		*-2 mm			1		0		0	-4	3	-2	3	3
D.E. 3	35.756205	-78.621873	in trib.; sand ends 75	0	3.5	1.5		~ 4			*-5mm			4	appx: sludgey	neg 1 to 0		5	-3	-2.5	3	0.5	0
Delta Interior D.I. 1	35.755925	-78.621879		0	0	0		x	compromised by tree fall		~0	reset stake after being hit by deadfall		1.5		2		2.5	2.75	3.5	3.5	2.5	4.5
D.I. 2	35.75601	-78.621962		0	3	2		3.3			4			3.5		3.5		5	5	6.5	5.5	5.5	5
D.I. 3	35.756043	-78.622008		0	0	1		2.8			2.5			2		2.5		2.5	5.5	8	6	6.2	7.5
Delta Back D.B. 1	-35.755768	-78.622037		0	0	1		2.8			2			1.75		4		4.5	2	7	2.5	3.2	6.5
D.B. 2	35.7558	-78.622059		0	3	2		4.05			6			3.25		6		6	8	9	7.5	8	8.5
D.B. 3	35.755893	-78.622131		0	4	2.4		4.3			4.5			6.5		0	newly reestablished	1.5	2.5	5	3	3.75	4

avg = 4.44444

: depth x area=

41.88 m3 sediment accumulated

Walnut Creek Overbank Deposit Into ZH Wetland Tracking

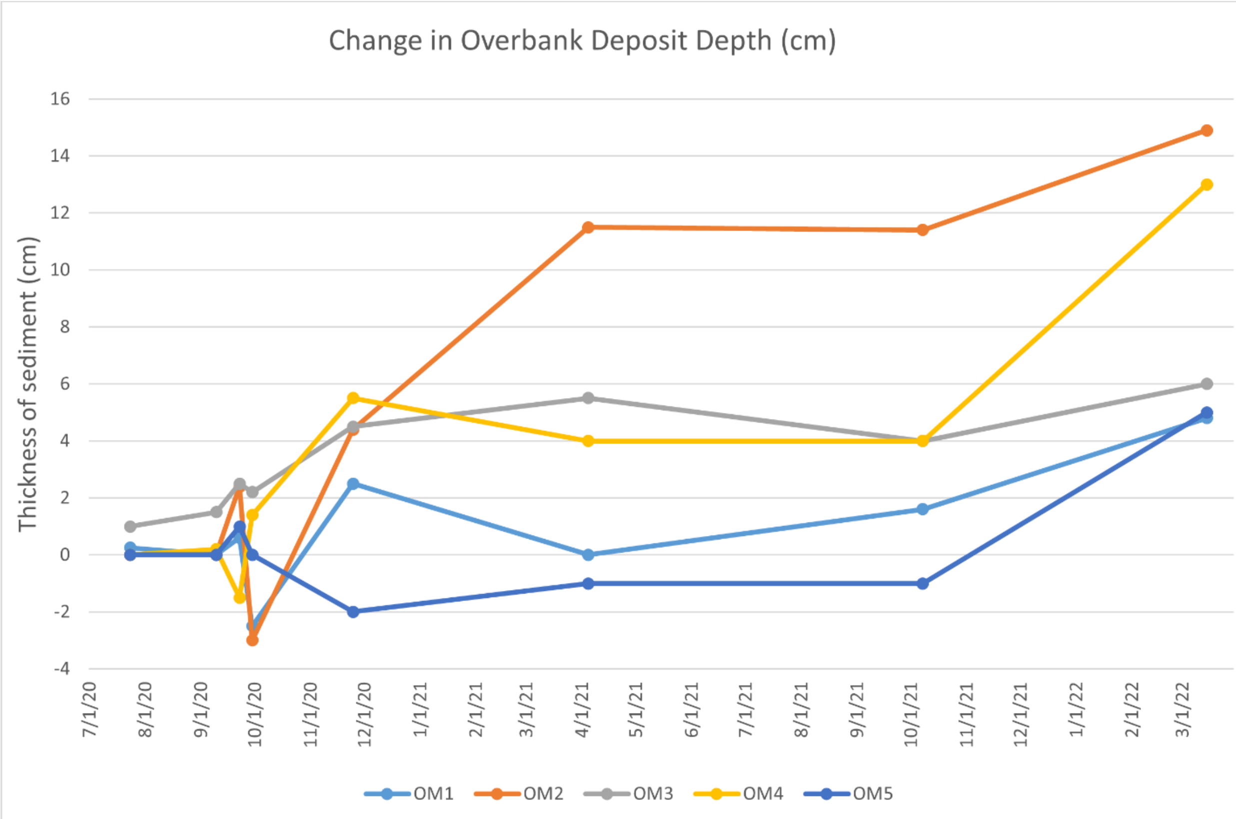
	Cumulative change in elevation of sediment (cm)							
Marker	7/24/20	9/10/20	9/23/20	9/30/20	11/25/20	4/5/21	10/8/21	3/15/22
Overbank marker (OM) 1	0.25	0	0.6	-2.5	2.5	0	1.6	4.8
OM2	0	0	2.4	-3	4.4	11.5	11.4	14.9
OM3	1	1.5	2.5	2.20	4.5	5.5	4	6
OM4	0	0.2	-1.5	1.4	5.5	4	4	13
OM5	0	0	1	0	-2	-1	-1	5

distance from 0m 1 to om 5 = 51.7 m

distance from 0m5 to bank = 40 m

markers established 7/22/20

8.74 avg overbank deposit depth accumulated in 601 days
(spread over an estimated 7000 m2)
: 612 m^3 of mostly sand with some silt


Change in Overbank Deposit Depth (cm)
Thickness of sediment (cm)
16
14
12
10
8
6
4
2
0
-2
-4
7/1/20
8/1/20
9/1/20
10/1/20
11/1/20
12/1/20
1/1/21
2/1/21
3/1/21
4/1/21
5/1/21
6/1/21
7/1/21
8/1/21
9/1/21
10/1/21
11/1/21
12/1/21
1/1/22
2/1/22
3/1/22
OM1
OM2
OM3
OM4
OM5

Appendix F

Sediment Sampling Results

Site	Sample Location	Total Exchange Capacity (meq/100 g)	pH	Organic Matter (%)	Estimated Nitrogen Releas (lbs N/acre)	S* (ppm)	P* (mg/kg)	Bray II P (mg/kg)	Ca* (mg/kg)	Mg* (mg/kg)	K* (mg/kg)	Na* (mg/kg)
HAMMOND ROAD	INLET	7.04	6.2	5.91	105	30	26	47	935	106	55	30
HAMMOND ROAD	CENTER	10.6	6.2	9.8	124	15	21	30	1475	129	55	43
HAMMOND ROAD	OUTLET	10.84	6.2	7.41	112	12	27	48	1499	141	61	34
LAKE RALEIGH	INLET	3.95	6.5	1.49	50	8	15	25	544	69	40	15
LAKE RALEIGH	CENTER	3.14	6.6	1.98	60	7	12	18	449	51	20	19
LAKE RALEIGH	OUTLET	4.84	5.9	3.67	87	9	18	31	598	64	38	18
LITTLE JOHN	INLET	8.78	5.7	5.4	102	10	23	40	1004	103	67	22
LITTLE JOHN	CENTER	10.32	5.4	10.28	125	18	5	29	952	115	114	51
LITTLE JOHN	OUTLET	7.29	6.3	7.05	110	11	9	54	956	111	101	43
WAKECOTRNSPTCNTR	INLET	3.03	6.2	1.37	47	13	17	36	376	56	35	17
WAKECOTRNSPTCNTR	CENTER	7.85	5.5	4.65	96	14	16	52	771	103	40	41
WAKECOTRNSPTCNTR	OUTLET	3.32	6.3	0.74	30	8	17	19	398	76	37	19
WCWWETLANDCENTER	INLET	1.61	6.7	0.27	11	5	10	8	227	28	12	14
WCWWETLANDCENTER	CENTER	8.15	5.2	9.69	123	18	22	55	732	64	44	22
WCWWETLANDCENTER	OUTLET	1.41	6.9	0.4	16	5	11	13	193	31	13	15
WOODMEADOW	INLET	1.05	6.5	0.17	7	3	7	4	133	21	12	11
WOODMEADOW	CENTER	7.43	5.3	10.53	125	20	2	39	680	62	73	33
WOODMEADOW	OUTLET	6.29	5.2	4.77	98	12	33	75	486	82	65	27
ZHANG	INLET	3.8	6.1	0.84	34	36	11	11	428	96	22	21
ZHANG	CENTER	11.31	5.4	14.9	127	9	7	65	1003	153	74	81
ZHANG	OUTLET	8.55	5.5	4.14	91	11	16	30	893	94	44	18

Site	Sample Location	B* (mg/kg)	Fe* (mg/kg)	Mn* (mg/kg)	Cu* (mg/kg)	Pb (mg/kg)	Zn* (mg/kg)	Al* (mg/kg)	NO3-N (ppm)	NH4-N (ppm)	Nitrogen (%)	Bulk Density (g/cm3)
HAMMOND ROAD	INLET	0.63	667	31	< 0.20	119	25.99	533	< 0.5	14	0.19	0.77
HAMMOND ROAD	CENTER	0.55	450	66	6.52	106.3	25.17	764	2.0	5.8	0.41	0.64
HAMMOND ROAD	OUTLET	0.53	466	40	9.26	99.7	41.77	621	0.8	5.4	0.32	0.74
LAKE RALEIGH	INLET	0.52	588	88	0.93	12.6	7.92	340	< 0.5	6	0.05	1.16
LAKE RALEIGH	CENTER	0.58	744	77	0.75	9.1	6.79	302	< 0.5	10.1	0.06	1.19
LAKE RALEIGH	OUTLET	0.69	858	50	0.55	17	12.32	389	< 0.5	17.3	0.1	0.98
LITTLE JOHN	INLET	0.64	697	51	4.50	84.7	38.3	542	< 0.5	7	0.23	0.78
LITTLE JOHN	CENTER	0.70	1140	56	2.01	136	36.85	550	< 0.5	17.1	0.47	0.59
LITTLE JOHN	OUTLET	0.86	1398	312	1.65	83.6	32.88	381	< 0.5	141.1	0.26	0.73
WAKECOTRNSPTCNTR	INLET	0.65	1170	68	1.43	35.9	17.5	340	< 0.5	5.6	< 0.05	1.26
WAKECOTRNSPTCNTR	CENTER	0.71	1253	59	0.91	25.9	21.01	461	< 0.5	9.2	0.19	0.89
WAKECOTRNSPTCNTR	OUTLET	0.64	773	61	0.43	17.3	4.44	330	< 0.5	23.5	< 0.05	1.31
WCWWETLANDCENTER	INLET	< 0.20	128	8	1.08	18.3	4.81	121	< 0.5	2.3	< 0.05	1.37
WCWWETLANDCENTER	CENTER	0.73	660	39	14.46	441.5	74.26	777	< 0.5	8.6	0.38	0.62
WCWWETLANDCENTER	OUTLET	< 0.20	171	11	1.39	19.4	6.77	108	< 0.5	1.5	< 0.05	1.38
WOODMEADOW	INLET	< 0.20	90	9	0.34	<5/ND	1.26	105	< 0.5	1.5	< 0.05	1.45
WOODMEADOW	CENTER	0.95	1824	44	1.00	42.6	15.61	475	< 0.5	21	0.38	0.62
WOODMEADOW	OUTLET	0.42	557	37	8.14	302.5	30.05	953	< 0.5	2.5	0.37	0.84
ZHANG	INLET	0.38	448	18	0.76	<5/ND	7.84	177	< 0.5	2.7	< 0.05	1.35
ZHANG	CENTER	0.66	1264	49	1.33	88.4	25.28	380	< 0.5	9.8	0.53	0.52
ZHANG	OUTLET	0.51	499	11	12.11	113.7	45.38	506	14.1	3.4	0.14	0.91

* Mehlich III extractable elements

Appendix G

Ancillary Wildlife Observations

Table A-4. Ancillary vertebrate wildlife observations during site visits (HR=Hammond Rd, WP=Women's Prison, ZH=Zhang Property, APH=Apollo Heights). *recorded on camera traps nearby at Walnut Creek Wetland Center.

Class	Common Name	Scientific Name	HR	WP	ZH	APH
Amphibian	American bullfrog	*Lithobates catesbeianus*		X		
Amphibian	Green tree frog	*Hyla cinerea*		X		X
Amphibian	Southern leopard frog	*Lithobates sphenocephalus*		X		
Amphibian	Southern toad	*Anaxyrus terrestris*	X			
Amphibian	Two-toed amphiuma	*Amphiuma means*	X			
Amphibian	Unidentified frogs			X	X	
Reptile	Common snapping turtle	*Chelydra serpentina*				X
Reptile	Eastern box turtle	*Terrapene carolina carolina*	X		X	X
Reptile	Painted turtle	*Chrysemys picta*		X		
Reptile	Southeastern mud turtle	*Kinosternon subrubrum*		X	X	
Reptile	Unidentified turtles			X		
Reptile	Common water snake	*Nerodia sipedon*		X		
Bird	American robin	*Turdus migratorius*			X	
Bird	Barred owl	*Strix varia*	X			
Bird	Blue-gray gnatcatcher	*Polioptila caerulea*		X		X
Bird	Brown-headed nuthatch	*Sitta pusilla*		X		
Bird	Canada goose	*Branta canadensis*		X	X	
Bird	Carolina chickadee	*Poecile carolinensis*				X
Bird	Carolina wren	*Thryothorus ludovicianus*	X	X	X	X
Bird	Common grackle	*Quiscalus quiscula*		X		
Bird	Cooper's hawk	*Accipiter cooperii*			X	X
Bird	Downy woodpecker	*Dryobates pubescens*		X	X	
Bird	Eastern bluebird	*Sialia sialis*	X		X	
Bird	Eastern phoebe	*Sayornis phoebe*		X		X
Bird	Great blue heron	*Ardea herodias*			X	
Bird	Mallard duck	*Anas platyrhynchos*		X		
Bird	Mourning dove	*Zenaida macroura*			X	
Bird	Northern mockingbird	*Mimus polyglottos*			X	
Bird	Pileated woodpecker	*Dryocopus pileatus*		X		
Bird	Red-shouldered hawk	*Buteo lineatus*	X	X	X	X
Bird	Unidentified hawk	*Buteo* sp.			X	X
Bird	Red-winged blackbird	*Agelaius phoeniceus*		X		
Bird	Ruby-throated hummingbird	*Archilochus colubris*	X			
Bird	Turkey vulture	*Cathartes aura*			X	
Bird	Wood duck	*Aix sponsa*		X		
Mammal	American beaver	*Castor canadensis*	X	X	X	X
Mammal	Bobcat	*Lynx rufus*		*		
Mammal	Common raccoon	*Procyon lotor*			X	
Mammal	Coyote	*Canis latrans*		*		

Class	Common Name	Scientific Name	HR	WP	ZH	APH
Mammal	Eastern Grey Squirrel	*Sciurus carolinensis*	X	X	X	X
Mammal	Muskrat	*Ondatra zibethicus*		X		
Mammal	Nutria	*Myocastor coypus*		*		
Mammal	River otter	*Lontra candensis*		*		
Mammal	White-tailed deer	*Odocoileus virginianus*	X	X	X	X

The intensive urban study wetlands are year-round sources of food, water, shelter, and nesting habitat for a wide variety of birds (resident and migratory), amphibians, reptiles, and mammals (Table A-4). Fish such as mosquitofish (*Gambusia* sp.) were also observed, as well as invertebrates such as crayfish. The utilization of these wetlands by longer-lived animals like turtles also showed the wetlands have been providing habitat over a long period of time. The unique suite of species observed at each wetland also emphasizes the fact that individual wetlands cannot satisfy habitat requirements for all wildlife species, so heterogeneity in type, vegetation structure, and hydrology is essential to maintain full habitat function overall in urban settings. Connectivity to other habitat types and other wetlands is also crucial.

Appendix H

Literature Review

Literature Review on the Removal of Urban Stormwater Contaminants using Constructed Wetlands and Comparisons with Natural Wetlands

Submitted to the US Environmental Protection Agency by

The NC Department of Environmental Quality

Division of Water Resources

Water Sciences Section

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List of Abbreviations

DO	Dissolved Oxygen
FWS	Free Water Surface
HSSF	Horizontal Subsurface Flow
NC DEQ	North Carolina Department of Environmental Quality
NC DWR	North Carolina Division of Water Resources
NH_3	Ammonia
$NO_3^- + NO_2^-$	Nitrate + Nitrite
TKN	Total Kjeldahl Nitrogen
TN	Total Nitrogen
TP	Total Phosphorus
TSI	Trophic State Index
TSS	Total Suspended Solids
US EPA	United States Environmental Protection Agency
USGS	United States Geological Survey
VFCW	Vertical Flow Constructed Wetland

Purpose of Literature Review

The primary objective of this study was to quantify the capability of existing natural riparian wetlands in an urban setting to effect stormwater pollution, while the secondary objective was to compare results to those reported for manmade wetlands or wetland-like systems constructed to treat similar materials known to exist in their inflowing surface waters. To accomplish the latter, it was necessary to define the compromised water quality that may occur in watersheds of various scales within urbanized landscapes. Further, an understanding was needed of the existing status of water quality in undisturbed land use contexts to enable calibration of expectations about what would quantitatively constitute water quality uplift of urbanized (i.e., “disturbed”) surface water flowing through the intensive study wetlands. This literature review included studies on constructed wetlands and/or wetland water quality in both urbanized and undisturbed settings, focusing on the parameters selected for this project. The following is a summary of the background research that addressed the above stated objective and its application to the findings for this study’s wetland sites.

1.0 Contaminants in Stormwater

1.1. General Background and State Area Contaminant Characterization Studies

“Natural” background levels of surface waters are influenced firstly by atmospheric deposition. In 2021, mean annual wet deposition for total inorganic nitrogen (ammonium and nitrate combined) across 199 National Atmospheric Deposition Program monitoring locations was 3.0 ± 1.5 kg/ha (NADP 2021). In North Carolina, mean inorganic nitrogen wet deposition for 2021 ranged from 2.0 to 3.6 kg/ha (NADP 2021). In 2017, global wet nitrogen deposition ranged from 0.02 to 102 kg N per ha per year, with most global regions averaging roughly 3–20 kg N per ha per year (Zhang et al. 2021). Mean annual deposition in the US for 2017 was 3.2 ± 1.8 kg N per ha per year (Zhang et al. 2021). Kuenzler et al. (1977) reported rainfall nitrogen concentrations of 0.36 mg/L and phosphorus at 0.06 mg/L in eastern North Carolina. Therefore, depending on the waters of interest, depositional amounts through rain may have a substantive impact on nutrient concentrations in stormwater.

The second non-human contributor of material constituents to surface waters is natural processes that include the interaction of surface geology and biotic life cycles. Mineralogical, soil-building, and plant processes produce typical concentrations of organic nitrogen, TKN, and total nitrogen of 1.5 mg/L and total phosphorus of 0.022 mg/L, based on a study of 85 relatively undeveloped basins in the US (Kadlec and Wallace 2009). Background levels of nutrients in Louisiana swamps have been reported at $\leq$ 3 mg/L for nitrogen and $\leq$ 1 mg/L for phosphorus (Hunter et al. 2009). The USGS reported background concentrations in undisturbed streams in North Carolina for total nitrogen at about 0.3 mg/L with the highest stream sample at 0.79 mg/L (Caldwell 1992). The maximum nitrate reported in that study was 0.44 mg/L, ammonia slightly higher than 0.02 mg/L, and total phosphorus concentrations at 0.03 mg/L for baseflow and 0.04 mg/L for storm flow conditions. Mean copper and lead levels reported were below 10 µg/L and mean zinc concentrations were less than 20 µg/L. One outlier sample of lead, reported at 64 µg/L, was understood as vehicle-exhaust effect. The USGS report (Caldwell 1992) also contended that atmospheric deposition was a major source of metals but provided no quantitative data. Kadlec and Wallace (2009) reported average metals concentrations around the world's freshwaters at about 10 µg/L copper and 0.2 µg/L lead, though underlying geology can contribute higher amounts at the local level.

Eleven (11) stream sites across NC that are regularly monitored by NC DEQ with similar land use characteristics in their watersheds were recently reviewed in regard to the pollutants of concern for this study (US EPA 2020). The 2017 sampling results showed mean TKN at 0.52 mg/L (maximum of 2.09 mg/L), mean total phosphorus at 0.06 mg/L (maximum of 0.58 mg/L), mean ammonia at 0.056 mg/L (maximum 0.43 mg/L), and mean nitrate at 0.36 mg/L (maximum of 4.07 mg/L). These stream sites were sampled monthly in their study period. The USGS collected grab samples from 19 headwater streams in Raleigh, NC in 2019 to analyze nutrients and found median specific conductivity of 151.8 µS/cm, median TKN of 0.03 mg/L (range 0.01 to 0.07 mg/L), median nitrate + nitrite of 0.74 mg/L (range 0.04 to 2.46 mg/L), and median total phosphorus of 0.05 mg/L (range 0.016 to 0.166 mg/L) (Hopkins pers. comm. 2021 unpubl. data).

Historical (1970s) sampling in Rocky Branch (a south-central Raleigh, NC stream) yielded mean TKN at 0.54 mg/L, nitrate at 0.83 mg/L, ammonia at 0.11 mg/L, phosphate at 0.69 mg/L, copper at 2.8 µg/L, and zinc at 8.25 µg/L (National Water Quality Monitoring Council 2019). An older 1977 study of the Haw River (arguably the mostly industrialized subbasin in NC) found copper levels between 40 and 100 µg/L, and zinc levels between 70 and 190 µg/L (Shuman et al. 1977). Most of this contamination was caused by known industrial point sources.

A study of stream channelization in the inner coastal plain of NC concentrated on a stream with 45% of its basin under crop, 1% pasture, 10% "miscellaneous", and the rest woodland (Swift Creek in Ayden, NC) (O'Rear 1975). That study reported nitrogen (particulate and dissolved organic) in the range of 0.075 to 0.75 mg/L, nitrate from 1.0 to 1.2 mg/L, ammonia 2.05 to 2.3 mg/L, and nitrite 0.015 to 0.2 mg/L. Phosphorus ranged from 0.25 to 2.8 mg/L. Dissolved metals studied included zinc (10–70 µg/L), lead (0.009–0.015 µg/L), and copper (< 9 µg/L). Channelization, seasonality, and storm flow versus baseflow conditions affected instantaneous results. Gambrell et al. (1975) found nitrate runoff concentrations from well-drained farm soils in the coastal plain of NC ranging from 7 to 15 mg/L (46 kg/ha per year).

A study of a Raleigh reservoir by NC State University found total phosphorus levels mainly at or below 0.05 mg/L and maximum total nitrogen levels at 1.2 mg/L (Burkholder 1992). A 2015 NC DEQ basin report found levels of total phosphorus in another nearby Raleigh area reservoir (Lake Benson) between 0.04 and 0.10 mg/L in the photic zone, with TKN levels between 0.60 and 1.10 mg/L (NC DEQ 2015).

The above literature findings are summarized in Table 1.

Table 1. Background levels reported for various North Carolina surface waters. Numbers are means unless otherwise noted. TP = total phosphorus; TN = total nitrogen; TKN = total Kjeldahl nitrogen; ND = non-detect.
***The Haw River had known industrial point sources for metal contamination.**
****Median.**

Source	TP (mg/L)	TN (mg/L)	Nitrogen components				Copper (Cu) (µg/L)	Lead (Pb) (µg/L)	Zinc (Zn) (µg/L)
			TKN (mg/L)	Ammonia (NH_3) (mg/L)	Nitrate (NO_3^-) (mg/L)	Nitrite (NO_2^-) (mg/L)			
11 similar NC streams with similar watersheds (US EPA 2020)	0.06(0.58 max)		0.52 (2.09 max)	0.056 (0.43 max)	0.36 (4.07 max)				
Rocky Branch, Raleigh, NC (National Water Quality Monitoring Council 2019)	0.69		0.54	0.11	0.83		2.8		8.25
19 headwater streams in Raleigh, NC (USGS 2019)**	0.05 (range 0.016 to 0.166)	0.75	0.34 (range 0.041 to 2.46)	0.03 (range 0.01 to 0.07)	0.4 (range 0.13 to 2.44)	0.01 (range 0.001 to 0.014)			
Undisturbed streams in NC (Caldwell 1992)	0.03 baseflow; 0.04 storm flow	0.3 (0.79 max)		0.02 max	0.44 max		<10	<10 (outlier of 64)	<20
Haw River, NC (Shuman et al. 1977)*							40 to 100		70 to 190
Streams in Inner Coastal Plain of NC (O'Rear 1975)	0.25 to 2.8	0.075 to 0.75		2.05 to 2.3	1.0 to 1.2	0.015 to 0.2	< 9	0.009 to 0.15	10 to 70
Runoff from farm soils in Coastal Plain of NC (Gambrell et al. 1975)					7 to 15				
Reservoir in Raleigh, NC (Burkholder 1992)	$\leq$ 0.05	1.2 max							
Lake Benson, Raleigh, NC (NC DWQ 2015)	0.04 to 0.10		0.60 to 1.10						
This study: 4 urban streams (inlets) into 4 natural wetlands (NC DEQ DWR 2023)	0.22 (0.18 median)	1.2 (1.0 median)	0.71 (0.67 median)	0.14 (0.07 median)	0.52 combined (0.35 median)		4.2 (4.5 median)	23.7 (2.6 median)	3.5 (18.5 median)

1.2. Urban Area Stormwater Characterizations

A recent study of stormwater contaminants in 21 urbanized areas across 17 states in the conterminous US found concentrations, loads, and yields of organic compounds were positively related to impervious surfaces and highly developed urban catchments (Masoner et al. 2019).

The study reported moderately low median concentrations of copper, lead, and zinc (Table 2). Data were not reported for nitrogen species or TSS.

The National Stormwater Quality Database v4.02 calculated median concentrations from stormwater runoff data collected from more than 9,000 runoff events from about 200 municipalities across the US for TSS (58 mg/L), total phosphorus (0.236 mg/L), total Kjeldahl nitrogen (1.4 mg/L), nitrate + nitrite (0.6 mg/L), ammonia (0.3 mg/L), Cu (11.3 µg/L), Pb (6.0 µg/L), and Zn (82 µg/L) (Pitt et al. 2018). A study of TSS, metals, and nutrients in surface runoff in the Atlanta, Georgia area at 10 urban sites returned the following results: copper median concentration was 6.2 µg/L with maxima between 7.1 and 110 µg/L; lead median at 0.77 µg/L, maxima between 3.1 and 13 µg/L; and zinc median at 26.4 µg/L, maxima between 12 and 1300 µg/L (Horowitz et al. 2008). Total nitrogen had a median of 1.98 mg/L, ranging from 1.6 to 18 mg/L for storm maxima, and total phosphorus had a median of 0.0385 mg/L with maxima between 0.04 and 3.2 mg/L. Mean specific conductivity in streams in the urban Washington DC area ranged from around 550 to 750 µS/cm between 2005 and 2019 (Hopkins 2019). In a review of thousands of storm events (mostly from the US) reported to the International Stormwater Best Management Practices database and the National Stormwater Quality Database, Pamuru et al. (2022) calculated mean stormwater concentrations of TSS (104 mg/L), total phosphorus (0.35 mg/L), total nitrogen (2.08 mg/L), Cu (45.3 µg/L), Pb (22.3 µg/L), and Zn (144 µg/L).

The above literature findings are summarized in Table 2.

Table 2. Urban surface water contaminant levels reported. Asterisks in first column denote medians reported; otherwise values are means. TSS = total suspended solids; TP = total phosphorus; TN = total nitrogen; TKN = total Kjeldahl nitrogen; ND = non-detect.

Source	TSS (mg/L)	TP (mg/L)	TN (mg/L)	Nitrogen Components			Copper (Cu) (µg/L)	Lead (Pb) (µg/L)	Zinc (Zn) (µg/L)
				TKN (mg/L)	Ammonia (NH_3) (mg/L)	Nitrate + Nitrite (NO_3^- + NO_2^-) (mg/L)			
Stormwater in 21 urbanized areas, 17 states (Masoner et al. 2019)*		.092 (.778 max)					5.5 (68 max)	0.39 (2.1 max)	15 (189 max)
Stormwater from over 200 municipalities in the US (Pitt et al. 2018)*	58	0.236		1.4	0.3	0.6	11.3	6.0	82
Urban stormwater across the US (Smullen et al. 1999)	54.5	0.26	2.00	1.47		0.53	11.1	50.7	129
Surface runoff at 10 urban sites in Atlanta, GA (Horowitz et al. 2008)*		0.039 (max 0.04 to 3.2)	1.98 (max 1.6 to 18)				6.2 (max 7.1 to 110)	0.77 (max 3.1 to 13)	2.64 (max 12 to 1300)
Review of international stormwater data (Pamuru et al. 2022)	104	0.35	2.08				45.3	22.3	144
This study: 4 urban streams (inlets) into natural wetlands	45.5 (15.0 median)	0.22 (0.14 median)	1.23 (1.08 median)	0.71 (0.57 median)	0.14 (0.11 median)	0.52 (0.46 median)	4.2 (3.1 median)	23.7 (2.0/ND median)	3.5 (16 median)

1.3. Acceptable Contamination Levels: Understanding Pollution Presence in Stormwater in Wetlands

Major contaminants in surface waters include total suspended solids (TSS), metals, phosphorous, multiple nitrogen species, and oil/grease. All of the parameters of interest in this study—phosphorus, total Kjeldahl nitrogen, nitrate + nitrite, ammonia, oil and grease, TSS, copper, lead, and zinc—are naturally occurring elements and compounds (except for the possibility of some unknown fraction of total suspended solids), and hence can be found in background levels in the wetland and stream waters monitored, as noted in the previous section.

Regulatory standards for these substances usually revolve around two major concerns: human health and aquatic non-human biotic health (Table 3). The US EPA lists action levels for human drinking water for nitrate-nitrogen at 10 mg/L and nitrite at 1 mg/L (maximum contaminant levels), 1300 µg/L copper and 15 µg/L lead (treatment technology action levels), and 2000 µg/L zinc (lifetime health advisory level) (US EPA 2016, 2018, 2022). EPA has set a secondary maximum contaminant level (a non-mandatory standard set for aesthetic considerations) for zinc at 5000 µg/L, above which drinking water may have a metallic taste. EPA also lists "Criterion Continuous Concentration (CCC)" or concentrations at which an aquatic community can be exposed indefinitely without injurious effect. The CCC levels at a hardness of 100 mg/L are 2.5 µg/L for lead, and 120 µg/L for zinc. Hardness in the wetland streams in the current study ranged from 28 to 120 mg/L, with a median of 61 mg/L.

In North Carolina, regulatory thresholds vary according to the classification of water considered, which is based on the designated uses of that body of water. In Class C waters, secondary recreation and biological integrity—including fish and other aquatic life—are protected. For Class C waters, surface water standards for copper, lead, and zinc are calculated for dissolved metals based on hardness. Other NC fresh surface water classifications add protections for primary recreation (e.g., swimming; Class B) and drinking water sources (Classes WS-I to WS-V). Supplemental classifications further add or modify standards in specific surface waters, such as increasing the minimum dissolved oxygen standard in designated trout (Tr) waters and providing additional protections for designated High Quality Waters (HQW). Surface water standards may be numeric (e.g., for Class C: standards for turbidity are 50 NTU in streams, 25 NTU in lakes and reservoirs, and 10 NTU in Tr waters) or narrative, such as that oils, grease, and other wastes "shall not render the water injurious to public or aquatic health…" (NC DEQ DWR 2023). In NC, as in other states, National Pollutant Discharge Elimination System (NPDES) permitted facilities have site-specific limits for applicable parameters in discharges to ensure that surface water standards are met in receiving streams (US EPA 2020).

Regulatory bodies in other parts of the US (e.g., the New Mexico Environment Department) have promulgated similar surface water standards for the protection of aquatic life and drinking water supplies (New Mexico Administrative Code 20.6.4.900). Various states use total nitrogen thresholds to determine if streams or lakes meet their nutrient criteria (Table 4).

Table 3. Thresholds and/or regulatory standards for various substances and water types. Water quality standards are shown as maximum values unless otherwise specified.

Criterion	Nitrogen Species	Copper (Cu)	Lead (Pb)	Zinc (Zn)	Turbidity	Other
US EPA Drinking Water Maximum Contaminant Level (MCL), Treatment Technology Action Level, or Health Advisory Level	Nitrate: 10 mg/L; Nitrite: 1 mg/L	1300 µg/L	15 µg/L	2000 µg/L		
US EPA Criterion Continuous Concentration[1] (CCC, at 100 mg/L hardness)			2.5 µg/L	120 µg/L		
NC DWR Class C waters (standards apply to all NC fresh surface waters)[2]		At 61 mg/L hardness, WQS for chronic exposure is 5.9 µg/L dissolved Cu; at 100 mg/L hardness, chronic WQS is 9.0 µg/L	At 61 mg/L hardness, WQS for chronic exposure is 1.5 µg/L dissolved Pb; at 100 mg/L hardness, chronic WQS is 2.5 µg/L	At 61 mg/L hardness, WQS for chronic exposure is 77.7 µg/L dissolved Zn; at 100 mg/L hardness, chronic WQS is 118.1 µg/L	50 NTU (Nephelometric Turbidity Units) (non-trout waters)	Total arsenic: 10 µg/L; pH between 6.0 and 9.0 except swamp waters may be as low as 4.3 if due to natural conditions
NC DWR Drinking Water Supply surface water standards or in-stream target values	Nitrate: 10 mg/L; Nitrite: 1 mg/L				TDS: 500 mg/L	Chloride: 250 mg/L; Hardness: 100 mg/L
NC DWR High Quality Waters (wastewater effluent limits)	Ammonia: 2 mg/L				TSS: 10 mg/L for trout waters; 20 mg/L for non-trout waters	DO > 6 mg/L
New Mexico Environment Department surface water standards (NMAC 20.6.4.900)	Nitrate as N: 10 mg/L in drinking water supplies	At 100 mg/L hardness, chronic WQS is 8.96 µg/L dissolved Cu	At 100 mg/L hardness, chronic WQS is 2.52 µg/L dissolved Pb	At 100 mg/L hardness, chronic WQS is 121 µg/L dissolved Zn		

[1] US EPA CCC is the concentration at which an aquatic community can be exposed indefinitely without injurious effect.

[2] Median hardness for the intensive study wetland inlets and outlets was 61 mg/L $CaCO_3$. NC surface water quality standards (WQS) for dissolved metals in fresh waters increase with increasing hardness according to state regulatory formulas.

Table 4. Total nitrogen numeric criteria values in place in a variety of states in the US. Source: US EPA 2023.

State	Total Nitrogen Values/Range
Arizona – human health & aquatic life	cold/warm lakes, specific rivers; 0.6 to 3.0 mg/L
California	site specific by HUC; 0.05 to 4.0 mg/L
Colorado	lakes >25 acres cold (0.426 mg/L) and warm (0.91 mg/L)
Florida	vary by region; 0.67 to 1.87 mg/L
Georgia	lake specific; 3.0 to 4.0 mg/L
Massachusetts – aquatic life criterion	0.38 mg/L
Missouri – aquatic life	Lake specific; 0.20 to 0.616 mg/L
Montana	ecoregion specific; 0.25 to 1.3 mg/L
Nebraska – aquatic life	0.8 to 1.0 mg/L
South Carolina	large lakes >40 acres, by ecoregion: Piedmont (1.5 mg/L), Mountains (0.35 mg/L), Coastal Plain (1.5 mg/L)

The intensive study wetlands used in this project included large areas (acres) of standing water to some depth (maxima to approx. 1 meter). Therefore, limnological standards for contamination were also considered (Table 5). Traditional limnological investigations have used the Trophic State Index (TSI), a semi-quantitative water quality scale originating from Sweden used to categorize water bodies as oligotrophic, mesotrophic, eutrophic, or hypereutrophic. The TSI can be calculated using one of three variables: chlorophyll-a, total phosphorus, and water clarity as defined by Secchi disk depth. Though based initially on characteristics of only Northern European and American temperate and sub-arctic lakes, the TSI is still used in modified forms for other surface water provenances around the world (Table 5). The most widely considered US version sets the possibility of algal blooms and other submerged aquatic vegetation (SAV) problems (e.g., “scums”, bottom anoxia) occurring at total phosphorus levels above 0.048 mg/L, but possibly as low as 0.024 (Carlson 1977). This level is labelled eutrophic on the TSI scale. Burkholder (1992) considered Lake Raleigh (with total phosphorus ranging from 0.035 to 0.075 mg/L) to be mesotrophic or moderately clear most of the summer but on the edge of becoming eutrophic with increased nutrients. The USGS, in its 1992 NC stream report (Caldwell 1992), cited several authors (Sawyer 1947; Sakamoto 1966; Vollenweider 1971) who considered eutrophication conditions in streams and lakes to begin at levels above

0.3 mg/L total nitrogen and 0.1 mg/L total phosphorus. Scientists at NC DWR have used standards of total phosphorus at 0.04 mg/L and TKN at 0.5 mg/L to define the boundary between mesotrophic and eutrophic in NC lakes (Vander Borgh, M. pers. comm. 2022). Dodds et al. (1998) modified the TSI to apply to stream environments; they considered the mesotrophic to eutrophic boundary in streams to be 0.075 mg/L for total phosphorus and 1.5 mg/L for total nitrogen.

Table 5. Comparison of standards for identifying trophic states in lakes and streams.

Source	Trophic State	Location/Type of Water Body	Total Phosphorus Level
Burkholder 1992	Mesotrophic/ Eutrophic boundary	Lake Raleigh, Raleigh, NC	0.035 to 0.075 mg/L
Carlson 1977	Eutrophic (algal bloom/SAV problems)	Northern temperate to subarctic lakes	Above 0.048 mg/L but possibly as low as 0.024 mg/L
Caldwell 1992 (multiple sources)	Eutrophic	North Carolina streams and lakes	Above 0.1 mg/L (and 0.3 mg/L for total nitrogen)
Dodds et al. 1998	Mesotrophic/ Eutrophic boundary	Streams	0.075 mg/L (and 1.5 mg/L for total nitrogen)
NC Division of Water Resources scientists	Mesotrophic/ Eutrophic boundary	North Carolina lakes	0.04 mg/L (TKN 0.5 mg/L)

In the constructed treatment wetland discipline, effluent target levels of approximately 5 mg/L of nitrate (and/or other nitrogen constituents) and 1 mg/L of phosphorus are considered extremely low and difficult to achieve, as are levels of copper and zinc at or below 10 µg/L and 2 µg/L for lead (the detection limit of most laboratories) (Petrasek and Kugelman 1983; Marecos do Monte and Albuquerque 2010; Tao et al. 2012; Wu et al. 2015; Bosak et al. 2016; Stefanakis 2020). Targets for TSS of approximately 10 mg/L or slightly higher in treatment effluents are also considered to be very low.

2.0 Processes That Can Change Contaminant Levels in Interior Wetland Surface Waters

2.1. Total Suspended Solids (TSS)

Total suspended solids (TSS; i.e., sediment) in excess amounts can be injurious to surface water ecosystems in a mechanical sense and some toxins can bind to and be transported by the solids. In North Carolina, regulatory thresholds for effluent discharges are limited to 20 mg/L TSS in designated High Quality Waters, and 10 mg/L TSS in designated trout (Tr) waters. These levels are often greatly surpassed in urban stormwater runoff (Pitt et al. 2018; Pamuru et al. 2022).

Processes affecting TSS levels within wetlands include sedimentation (particularly at the surface inlet area[s]), periphyton litterfall into the water body, chemical precipitation of solid compounds created from influent and/or existing interior water, plankton litterfall, macrophyte litterfall, resuspension of interior sediments/soils, and macrophyte surface interception of flowing particulates (Kadlec and Wallace 2009). Wetlands can generate more TSS than is loaded in influent because of the interaction of plant life cycles with nutrient-containing sediments and dissolved constituents, as well as the non-biotic geochemical processes that may be occurring at the same time. A large fraction of the effluent TSS at any time therefore can be wetland generated. Kadlec and Wallace (2009) concluded that "the solids leaving the wetland will very often not be related to the solids entering but rather to the detrital fragments originating internal to the system" (page 217). Interactions between influent and internal storages can also be highly variable by season and diurnal conditions, so modeling removal rates for TSS is highly complex (Kadlec and Wallace 2009).

2.2. Metals

Typical sources of metal contamination include runoff from galvanized roof materials, car exhaust and other car-part sources, metals processing (i.e., plating), products introduced to rain and dryfall, and other road and building materials contacted by water (e.g., lead paint). Copper and zinc are essential to life but become toxic at certain levels depending on the tolerance of the ingesting life form. Lead is natural but not needed for life and is considered toxic when encountered in biotic life pathways in appreciable amounts. North Carolina regulatory limits for chronic exposure in Class C waters at 61 mg/L hardness are 5.87 µg/L dissolved copper, 77.71 µg/L dissolved zinc, and 1.46 µg/L dissolved lead. These limits have been exceeded in numerous urban watersheds across the United States (Horowitz et al. 2008; Kadlec and Wallace 2009; Masoner et al. 2019).

The two major modes by which wetlands remove and retain metals from surface water are uptake by plants and sorption by organic molecules (e.g., ligands) (Vymazal and Richardson 1995; Kadlec and Wallace 2009; Williams, A. pers. comm. 2020). The uptake of metals by vascular plants mostly occurs in the rootzone and not in the upper plant body (Nolte and Associates 1998). Algae can often uptake concentrations of metals in order[s] of magnitude greater than vascular plants. Living algal tissues can concentrate copper 5 to 10 times more than vascular plants, with typical dry weights of 100–1000 µg/g for a number of algal species (Kadlec and Wallace 2009). Lead can be taken up by algae at levels 200 times higher than macrophytes and zinc is "effectively" sorbed by algal necro mass (Fielding et al. 2022). Highly organic (peat) soils in wetlands can sorb copper to a very high concentration: 100 mg/g (Kadlec and Wallace 2009). Additionally, metals can settle to the bottom of wetland standing water through transformations and precipitations with other substances, but these change easily with changing water and substrate conditions (e.g., redox). Predictive modeling fails because conditions for chemical complexing of metals are highly variable in wetland waters and cannot be replicated by the controlled abiotic and thermodynamically stable conditions of lab experiments (Kadlec and Wallace 2009).

2.3. Phosphorus

Phosphorus (P) is an essential element that is often at such low concentrations in natural environments that it is considered “limiting” to plant growth and nourishment. It occurs inorganically as particulate P and phosphate, PO_4-P (orthophosphate). It is usually measured in environmental investigations including wetland studies as total phosphorus (TP): inorganic and organic forms together. NC DWR defines a lake with levels of TP above 0.04 mg/L and total Kjeldahl nitrogen (TKN) above 0.5 mg/L to be eutrophic. In studies of urban surface water and stormwater, TP levels have been recorded below (Horowitz et al. 2008) as well as hundreds of times above this level (Horowitz et al. 2008; Kadlec and Wallace 2009; Masoner et al. 2019).

Wetlands are considered good systems for dealing with high phosphorus water because they provide environments where the ten different naturally occurring phases of P can be interconverted, with the eventual sink being the wetland solid elements. The P phases, therefore, become a significant fraction of the mass of wetland plants, detritus, microbes, wildlife, and soils, although they normally occur in amounts 10 times lower than nitrogen compounds (Kadlec and Wallace 2009). Kadlec and Wallace (2009) suggest that the plant cycle returns most of the captured phosphorus back to the donating wetland waterbody but relegates enough to long-term soil accretion to provide net removal on a supra-seasonal time scale; thereby providing eventual reductions in levels of total phosphorus. Because microbial and algal masses are smaller and embedded or bound to soil compared to large plants, the dominant fraction of wetland P is contained in soils and a lesser amount in living plants and litter. However, algae, plankton, and other microbes can store significant amounts of P and place them into the soil-sediment system, thus acting as an entry way to long-term soil storage. Forested wetlands can improve phosphorus-impaired water quality over other vegetation types. A study of 76 wetlands found that the phosphorus removal rate constant in forested wetlands was 13.1 m/year as opposed to 3.1 m/year in unforested wetlands (Kadlec and Knight 1996). Overall, biotic cycling of P is extremely difficult to quantitatively track and variability across seasons and in influent waters can have dramatic effects. For instance, drying out wetland soils immobilizes P by oxidation and then rewetting remobilizes much of this fraction back into standing water (Kadlec and Wallace 2009).

2.4. Nitrogen Species

Like phosphorus, nitrogen is an essential element to life, but can be even more difficult and complex to understand as a surface water contaminant. This is because nitrogen occurs frequently in more species or phases than phosphorus in surface water, including gaseous components. Some of these are considered harmful to human and/or environmental health, although concentrations harmful to environmental health vary from site to site. Inorganic forms of nitrogen include ammonia (NH_3), nitrite (NO_2^-), nitrate (NO_3^-), and nitrous oxide (N_2O;

considered, along with methane, to be a greenhouse gas contributor). Nitrogen and phosphorus, though essential elements for plant life cycling, are considered problematic in surface waters when levels are high enough to overstimulate algae growth. As previously noted, NC DWR defines lakes with levels of TKN above 0.5 mg/L and total phosphorous (TP) above 0.04 mg/L to be eutrophic.

Ammonia and nitrate are considered toxic to human and animal life at certain levels. Thus, they are considered contaminants in surface water. These nitrogen species are introduced by human action into watersheds primarily from the handling of domestic sewage (ammonia) and from agricultural runoff (nitrate). Though nitrate levels are typically very low in sewage and treated effluents, nitrate is produced in the zone above saturated groundwater in agricultural areas due to the oxidation of ammonia-based fertilizers, possibly reaching levels of up to 40 mg/L (Kadlec and Wallace 2009). Ammonia and nitrate can occur in urban runoff when onsite wastewater treatment systems that include nitrification fail and from grass/lawn fertilizer excesses. The NC DWR effluent limitation for ammonia in HQW waters is 2 mg/L.

Once in a wetland, nitrogen species cycle physically and chemically similarly to phosphorus, but with more complex relationships with plants, microbes, and algae, and with volatilization that can transform and vent nitrogen species directly into the atmosphere. A wetland is a natural system uniquely suited to deal with nitrogen species, as it contains biota and physical structures that can process nitrogen into harmless dinitrogen gas, uptake the nitrogen into biologic structures (e.g., emergent herbs and trees) which also provide filtering for particulates, and store nitrogen long-term in accreting sediments (Kadlec and Wallace 2009). For example, the denitrification cycle that transforms ammonia and nitrite to N_2 gas in a series of steps involving microbes is replicated in wastewater treatment plants, but a recently discovered obscure and ancient bacteria natural to surface waters can directly convert ammonium and nitrite into N_2 and H_2O. This process, dubbed "annamox", results in a major turnover cycle of nitrogen in marine environments but was not suspected or described in wetlands until 1999 (Tao et al. 2012). Other nitrogen-transforming processes that occur in wetlands include settling, diffusion, plant translocation, litterfall, sorption, assimilation, nitrogen-fixation, and decomposition. Natural wetlands also have the advantage of being able to respond quickly to loading changes, provided the rates and concentrations do not overwhelm the subecosystems, a flexibility not usually shared by highly engineered filtering systems. Wetland nitrogen cycling is highly effected by seasonal changes, so the nitrate and ammonia loads in standing water can change significantly throughout the year. However, in all cases (per wetland, nitrogen species, and time-period of monitoring), a general net reduction in nitrogen should be accomplished by wetlands from influent to effluent (Kadlec and Wallace 2009).

3.0 Constructed Wetland Design and Performance

3.1. Design and Construction

Though natural wetlands have been documented to intentionally treat wastewater as far back as 1912, completely human-made wetlands installed to treat specific pollutants in contaminated water were not theoretically possible until 1952, when researchers at the Max Planck Institute developed the necessary technology (Bastian and Hammer 1993). During the late 1980s into the 1990s, treatment wetland construction became widespread, with most of the application being final treatment of secondary wastewater and some stormwater. These early constructed wetlands consisted of an inlet bringing water over selected media layers planted with herbaceous plants (but not trees) in a shallow water area allowing continuous flow-through to an outlet and infiltration to groundwater below. This type of constructed wetland is labelled free water surface (FWS) and has proven to be effective in removing TSS, biological oxygen demand (BOD), and to lesser degrees nitrogen, pathogens, and heavy metals (Stefanakis 2020). Phosphorus removal is limited. Since then, constructed wetland designs have developed into several morphotypes and configurations (outlined below) and have been applied to treat almost all types of wastewater streams, from individual domestic effluents to large industrial plants. It is important to note that as late as 2015, wetland engineering authors agreed that there were no widely accepted guidelines to designing these systems due to the fact that wetland geochemical and hydraulic processes that impinge on long-term pollutant removal are so staggeringly complex (Kadlec and Wallace 2009; Stefanakis 2020).

The second type of constructed wetland developed is the Horizontal Subsurface Flow (HSSF) wetland, which involves a substrate of gravel and sand with a bottom barrier (thus inhibiting infiltration to groundwater). The bottom is graded slightly to allow gravity movement of the water from inlet to outlet, as in FWS wetlands. The surface of the wetland water body is kept 5 to 15 cm below ground (media upper) surface. Influent must occur across the entire engineered bed width (through a perforated pipe) (Stefanakis 2020). Typical emergents planted in FWS wetlands are also planted in the HSSF substrate surface, including cattail (*Typha* spp.), common reed (*Phragmites australis*), bulrushes (*Scirpus* spp.) and reed canary-grass (*Phalaris arundinacea*). HSSF construction favors development of a biofilm in the substrate that enhances removal of organic matter and other TSS, but nitrogen and phosphorus removal may be variable depending on location and plant species (Thalla et al. 2019; Stefanakis 2020). An advantage of HSSF is that it requires a smaller area than FWS for the same contaminant load. This is significant since sufficient area to treat influent concentrations at usual flows is crucial in constructed wetland design and can be a detriment for FWS wetland removal performances. Both FWS and HSSF wetlands are now commonly being utilized in the US.

A third construction design, more common in Europe, was developed to increase the amount of oxygen delivered to the body of the wetland, thereby enhancing nitrogen removal processes. This type is termed Vertical Flow Constructed Wetland (VFCW) and features substrate layers of increasing coarseness over a bottom barrier, planted with common reed or similar emergents at the top. Water is applied to the top of the wetland through vertical pipes in aliquots; these are expected to percolate downward, thereby sucking in fresh air into the profile. Removal of phosphorus is limited in these constructed wetland types because the percolation times are short. VFCWs are used commonly to treat typical municipal wastewaters but also landfill leachate, dairy waste, and food processing water (Stefanakis 2020).

Hybrid designs that combine FWS, HSSF, and VFCW have evolved that try to replicate the advantages of each in one construction design. The subtype wetland areas are placed in-line and use different filter media, including zeolite, and sometimes waste, such as fly-ash. Different types of compost are also introduced into the top layers.

Floating wetlands have recently been developed to treat waste streams and eutrophic rivers and lakes. Emergents are planted on plastic floating platforms; roots create biofilm areas and uptake nutrients from the water column (Wu et al. 2015; Stefanakis 2020). How effective they can be is relatively unknown, especially given their size limitations in relation to the sizes of the overall waterbodies. Performance data are limited.

Other refinements have been added to constructed treatment wetlands. For instance, because contaminant removal generally increases with temperature, artificially heated components have been added in cooler climates. Settling basins, UV disinfection cells, trickling filters, and "biotowers" (a system that mixes nitrified water with effluents [Skancke 2007]) have also been developed and applied to existing or new constructed wetlands.

When there are sufficient conditions for successful constructed wetlands (e.g., adequate area for the loading levels), the most common barrier to continued successful functioning is bed-clogging due to lack of maintenance. Water flow through substrate media pores is reduced; consequently, oxygen input is decreased and can lead to “short-circuiting” or bypassing of influent directly to the effluent area downgradient of the wetland (Stefanakis 2020). Location-based challenges of successful treatment wetlands include areas with high groundwater tables and areas with steep slopes.

3.2. Performance

There are several ways to characterize a wetland’s ability to remove contaminants including inlet to outlet concentration drop, inlet to outlet loadings drop, outlet concentration versus areal input loading, rate constant versus areal input loading, and various mass balance models that determine amounts of material over time or over area over time (e.g., Brinson et al. 1981; Vepraskas et al. 2016). Kadlec and Wallace (2009) consider only the first and third conceptualizations "useful". In the literature, the most commonly reported performance characterization is concentration change from inlet to outlet as a percent.

There are many layers of complexity that make understanding and comparing wetlands' positive effects on water quality an inexact science. Because a wetland is a living assemblage of subecosystems that interact with different target water constituents differently, while also interacting with each other, a wetland cannot be viewed as a linear geochemical and biological process that produces an output based on an input as if it was a technological filtering system. Wetlands are better viewed as landscape features that interact with and transform watershed water moving downgradient, sometimes releasing more of a certain material than they receive on a short-term (usually seasonal) basis but slowly building up storage of most materials considered pollutants in detrital sediments over multiple growing seasons. As previously discussed (see Background Information and Purpose of main report), there are many factors influencing how wetlands interact with inflowing water across spatial and temporal scales. Additionally, the stochasticity of certain environmental and chemical processes makes predicting the effects of wetlands on water quality parameters even more inexact.

The concentrations of incoming materials that a wetland can deal with is not limitless. For example, Kadlec and Wallace (2009) maintained that wetlands are not particularly efficient at obtaining oxygen in sufficient quantities to deal with heavy pollutant loads. Therefore, constructed wetlands need aerators, vertical flow systems, etc. to complement oxygen from inflow, oxygen transfer down plant stems, and/or photosynthesis from underwater plants. In their evaluation of techniques to enhance constructed wetland performance, Wu et al. (2015) were less encouraged than other authors on the state of world-wide constructed wetland effectiveness. They concluded that, at the time of publication, constructed wetlands did not adequately remove nutrients to meet the world's environmental standards, lacked long-term data on performance and cost-benefit analyses, and were understudied in their effectiveness at removing heavy metals and micropollutants.

Two factors have complicated, if not held back, maximizing constructed wetlands' removal performances: an under-appreciation of the complexity of wetland processes (a steep learning curve that the industry is still climbing) and inconsistent or lacking long-term maintenance after initial success. This is illustrated in Skancke's 2007 evaluation of 11 FWS and HSSF constructed wetlands in New Mexico. These wetlands were emplaced in the early 1990s to treat light urban wastewater including water from housing developments, a resort and golf course, and a public school. All had pretreatment in the form of septic tanks. Wetland maintenance ranged from none to occasional harvesting of above-ground vegetation. Three wetlands consistently met state standards for nitrate and chloride effluents. Four others occasionally met standards, one never met standards, and three were abandoned to be retooled into fully artificial/non-wetland systems. The three sites meeting standards had additional components emplaced after the original treatment design was constructed and put online: powered fountains, a "biotower", and a trickling filter. Skancke (2007) concluded that the other systems, though properly sited and sized for anticipated hydraulic loading rates, did not allow for unanticipated additional aerobic incorporation. Another unexpected difficulty for future monitoring and evaluation was year-round evaporation at some of the wetlands, where the entire water body dried up for multiple seasons. Skancke does not discuss the possible effects of rewetting and mobilization on effluents. Nitrate removal in Skancke's study wetlands exhibited a wide range from 9.6 to 90.2 percent.

Experimental HSSF beds were emplaced to treat municipal wastewater in Cova da Beira, Portugal to lower contamination to levels suitable for reuse as irrigation (Marecos do Monte and Albuquerque 2010). The wetlands were successful in the sense that the low to moderate concentrations of nitrate, ammonia, and phosphorus were reduced by about 50%, even under very low (1.2 mg/L) dissolved oxygen conditions. However, transient but heavy loads of organic matter and solids clogged the bed substrate, an effect of unanticipated local agricultural discharge into the sewer system. This resulted in non-removal of TSS and Chemical Oxygen Demand (COD). Overall, the treated water was suitable for irrigation and drinking water as far as nitrate, but not for other regulated contaminants.

Both of these examples point out important common considerations when evaluating a wetland's water quality improving performance. The first is that the constraints of the regional environment and contributing watershed must be factored in to understand what is physically possible to expect from the constructed or natural wetland. A desert climate may cause total evaporation yearly; whereas seasonal dry-out may occur only a small percent of the year in a hot humid climate. Initial rewetting and outflow concentrations may be temporarily much higher than previous inputs after a drought has broken. A high desert location may include freezing conditions that will not occur in a more tropical locale. A wetland in an intensely farmed area should be designed to target nutrient reduction but may be incapable of substantially improving other less-regulated constituents (e.g., COD, TSS). Chloride control (measured as total dissolved solids [TDS]) is an important concern in surface and groundwater management in arid areas such as New Mexico due to the geology and climate but not in humid subtropical inland areas such as central North Carolina.

A second factor is that, even if maintenance is considered, construction design must anticipate problems that might not be associated with target constituents or show evidence at the initiation of wetland construction and operation. Constructed wetlands, like natural wetlands, change with time, so (unmodelled) changes in external or internal conditions can reduce performance over the long term, whereas initial performance may far outstrip that of a natural wetland.

Thirdly, a wetland may be considered "successful" by one stakeholder group but not another. Contaminant perception and understanding of impacts are constantly evolving. There are proven constructed wetlands that have dealt successfully with long-studied materials such as sediment, nutrients, and metals but their effectiveness at pathogens, petroleum, and pesticide removal has rarely been studied or reported. Emerging compounds of concern, such as PFAS and waste pharmaceuticals, have barely begun to be studied in any capacity, let alone in wetland-water interactions.

The following is an overview of how Kadlec and Wallace (2009) and others viewed a cross section of constructed wetlands' abilities to deal with the contaminants studied for this project.

3.2.1. Total Suspended Solids

Most treatment wetlands are "overdesigned" for TSS reduction. However, a large fraction of internal TSS at any point is due to factors that create TSS in the wetland itself (e.g., detrital biosystems, resuspension). Therefore, performance can only be measured by modeling particulate settling rates for specific waste waters on a case-by-case basis, making sure to account for internal generation and subtracting these background amounts from effluent amounts. High inputs of TSS have the tendency to clog HSSF and VFCW wetlands with excessive biological growth in substrate pores. In these cases, a settling pond is recommended for pretreatment before discharging into a wetland area proper. Examples of TSS reduction results from constructed wetlands found in the literature include:

- A site in Tarrant, Texas with three wetlands where TSS influent was 46, 37, and 28 mg/L and outflow was 6, 11, and 6 mg/L (79% reduction on average) (Kadlec and Wallace 2009).
- A Brawley, California wetland reported 216 mg/L influent TSS and 12 mg/L effluent TSS (94% reduction; with settling pond) while an Imperial, California wetland lowered TSS from 200 mg/L to 7 mg/L (97% reduction; with settling pond) (Kadlec and Wallace 2009).
- A literature review by Biswal and Balasubramanian (2022) reported TSS removal rates from 41 to 96% for a variety of types of constructed wetlands including HSSF and VFCW wetlands.

3.2.2. Metals

Wetlands are effective at retaining significant levels of several trace metals, primarily in the sediments. However, they are limited by toxicity thresholds of resident and downstream sensitive biota; when these limits are reached, the sediments need to be removed or the site abandoned. Reaching this limit can take decades or longer. Harvesting of above-ground wetland plant crop can be ineffective at removing certain accumulated metals. Metal accumulation in below-ground plant tissue is higher than in above-ground tissue for many metals, but may be lower for some metals, such as zinc (Vymazal and Březinová 2016). For 26 reported FWS systems, median concentration reduction in copper was 66%. Median reduction in lead concentrations was 62%, and median reduction in zinc concentrations was 68%. HSSF and VFCW systems generally had high removal rates for these metals; however, HSSF systems occasionally showed increases in metal concentrations (Kadlec and Wallace 2009):

- Copper removal ranged from 46 to 75% (median 62%) in VFCW; -27 to 84% (median 81.5%) in HSSF; overall median 63.5%.
- Lead removal ranged from 15 to 80% (median 46%) in VFCW; -220 to 98% in HSSF (median 25%); overall median 40%.
- Zinc removal ranged from -99 to 97% (median 77%) across the two system types.

3.2.3. Phosphorus

Phosphorus removal has been documented for over 250 FWS wetlands with influent concentrations ranging from 0.02 to 100 mg/L (Kadlec and Wallace 2009). As input concentrations increase, phosphorus load removal also increases, from 0.1 g/m^2/year to over 100 g/m^2/year. However, a wetland may take one to two years to respond to changing input in phosphorus. Subsurface flowing (non-FWS) wetlands rely on sorption sites to remove phosphorus, so they require periodic excavation and bed replacement to remove phosphorus on a long-term basis. Thus, subsurface flow wetlands are rarely designed with phosphorus retention as a primary objective and typically have limited phosphorus removal compared to FWS (Kadlec and Wallace 2009). For example, two HSSF wetlands in Londonderry, Ireland reduced median total phosphorus concentrations of 4.5 and 4.3 mg/L (inlets) to 3.9 and 4.0 mg/L (outlets), respectively, while one HSSF from Leicestershire, England reduced median phosphorus concentrations from 6.6 mg/L to 3.2 mg/L (Constructed Wetland Association 2006).

Although forested wetlands can improve phosphorus-impaired water quality, few constructed wetlands use trees because of the perception that time to maturity is too long (Kadlec and Wallace 2009). Additionally, some engineered forested wetlands have suffered negative impacts over years of high loadings, i.e., with tree die-off and replacement with emergents, such as cattail (Guntenspergen et al. 1980).

3.2.4. Nitrogen

In wetlands that are heavily loaded with nitrogen species, microbial cycling and removal processes dominate over vegetative processes. A majority of the nitrogen bound in a wetland's biota is released into the water and mobile “floc” layer with death and decay, but a small amount is permanently stored in soil (around 10 g/m^2/year) (DeBusk 1999; Kadlec and Wallace 2009). Effluent loads may change daily, monthly, seasonally, or annually. A large number (72) of FWS wetlands studied for nitrogen loading had nitrate as their main pollutant of concern (Kadlec and Wallace 2009). Median concentration reduction was 4.0 to 1.4 mg/L. Maximum nitrate influent was 121 mg/L, which was reduced to 66 mg/L with constructed wetland treatment.

Kadlec and Wallace (2009) comment that nitrogen chemical transformations are a challenge to ecological engineering, which makes quantitatively comparing an individual constructed wetland's nitrogen-handling ability with other constructed and natural wetlands tenuous.

3.3.Stormwater Wetlands versus Continuous Flow Wetlands

Constructed stormwater wetlands catch rain event flows, release them after a "nominal detention time", and at some point before the next event, stop flowing. Because the antecedent water aliquot is treated by the wetland in a segregated fashion ("batch-held"), Kadlec and Wallace (2009) considered stormwater wetlands to have too high a degree of variability to compare with other constructed treatment wetlands even though the processes of contaminant removal are essentially the same. Carleton et al. (2001) thought it possible to compare them with continuous treatment wetlands using plug-flow modelling but urged caution when applying. Even the idea that the ratio of the area of the stormwater wetland to the catchment area is crucial is considered by Kadlec and Wallace (2009) as having "not been tightly quantified" (page 571). Stormwater wetlands must have a settling pre-basin to operate well.

An important note is that non-vegetated ponded systems do not exhibit the same nitrogen-processing abilities as wetlands do. The interplay between aerobic and anaerobic zones and phases that occurs in different wetland areas, layers, and individual wetland plants during seasonal drying/wetting cycles creates the conditions for reducing and oxidizing necessary for nitrogen transformations. Plants also contribute essential carbon in the denitrification transformation. Particulate settling is not nearly as prevalent in the nitrogen cycle in surface waters as with the phosphorus cycle.

4.0 Comparing Natural Urban Wetlands with Artificial Wetlands Constructed to Treat Contaminated Streams

4.1. General Differences Between Natural Wetlands and Constructed Wetlands

Several notable differences exist between natural and constructed wetlands in general, which make meaningful comparisons tenuous. Comparing constructed wetlands with natural wetlands necessitates considering caveats including:

4.1.1. Differences in Hydraulic Regimes

Many natural wetlands (including the sites studied in this project) experience a combination of continuous low flows (baseflows) and pulse flows (storm flows) from the contributing watersheds over the long term. For example, during most of the year, the intensive study wetlands operated under baseflow conditions but at some point during the summer or fall, part or all of the flow occasionally stopped, depending on rainfall. Constructed wetlands experience either constant flow situations or pulse flow situations, but not usually both. For instance, bioretention swales (i.e., small constructed wetlands sited in parking lots or other impervious environments) will contain water only during storms periods, if properly sized and maintained, and therefore will not constantly drain between storm events. A bioretention swale could not be connected, for example, to a constant tertiary wastewater stream, in addition to dealing successfully with stormwater run-off.

A common problem in siting constructed wetlands is lack of land available, so many constructed wetlands operate with the minimal areal extent considered viable. Therefore, the time that new influent water spends inside the wetland before being discharged (i.e., hydraulic retention time) is usually on the order of hours or days. For example, a recently constructed stormwater wetland treating runoff in an urban park setting in China was only a fraction of an acre (0.13) in size with an average retention time of 2.6 days (Li et al. 2017). It is reasonable to assume that retention time in natural wetlands exceeds this short time period. Natural wetlands are usually much larger than 0.13 acre, are always the “correct” size for their watersheds (as long as they are not artificially impinged upon) and will adapt to higher loadings to some maximal extent before degrading. Hydraulic retention times were not calculated for the natural wetlands in this study but were estimated to be, during baseflow conditions, on the order of weeks.

4.1.2. Differences in Contamination Regimes

Except for a few instances found in the literature, point-source wastewaters with high flows and pollutant concentrations are not discharged directly into natural wetlands. One of these rare, documented examples involved wastewater (including high-concentration phosphate cleaners) from Kincheloe Air Force Base in Kinross, Michigan that discharged directly into a natural spruce bog as late as 1977 (Kadlec and Bevis 1990). In general, natural wetlands interact with a wider range of lower concentration watershed pollutants for longer time periods of capture (hydraulic retention time) compared to constructed wastewater or stormwater wetlands. An example of extremely high loadings flowing into a constructed wetland, and far exceeding "normal" levels, is the washwater from a potato farm in Ontario, Canada being piped into a constructed wetland with average biological oxygen demand (BOD) of 1113 mg/L, TSS of 4400 mg/L, TN of 311 mg/L, and TP of 42.5 mg/L (Bosak et al. 2016).

4.2. Differences Between Natural Intensive Study Wetlands and Constructed Wetlands

Based on the literature reviewed above, typical reported ranges of successful constructed wetland reduction percentages from inlets to outlets are 25 to 75% reduction, depending on the parameter. Attempts at estimating pollutant loadings made it clear that natural wetlands can display a wide range of loadings behavior from event to event versus engineered wetlands where the physical environment can be under much more direct human control (e.g., inlet and outlet sizes and configurations [e.g., pipes], hydraulic vectors impinging on the wetland area). The natural wetlands studied in this project exhibited a wide range of sizes and structures: forested (approximately 50%), open water/emergent (25%), cattail/grass/arrow arum marsh (25%). Wetland areas ranged from 3.3 to 19 acres. In contrast, constructed wetlands are sized and designed for the specific ranges and types of contaminants known to be in the influent.

This project's natural intensive study wetlands differ from typical constructed wetlands by being hydraulically connected to a major stream (Walnut Creek) through overbank flooding. This creek contributed sediment to the wetlands through overbank flooding. This hydraulic connection is in addition to the perennial and intermittent stream inflows and outflows intersecting the study wetland areas.

Another factor contrasting this study's wetlands with engineered treatment wetlands is that the study site stream inlets and outlets were in some cases highly erodible. Constructed wetlands, on the other hand, do not experience appreciable amounts of bank erosion due to the construction and maintenance of inlets and outlets, including the use of pipes. Consistent resuspension of bottom sediments (due to storm flooding or wind action) was a major factor only in the one intensive study wetland with a large (12 acre) and deep (> 1 meter) pond, especially during the non-growing season when emergents (*Peltandra virginica*) had died off. However, temporary resuspension probably occurred to some smaller degree during major overbank flooding events in the other three intensive study wetlands. This phenomenon likely occurs in constructed wetlands as well.

The four natural wetlands intensively studied in this project encountered numerous disturbances that altered the hydrology and sediment loads of the wetlands. These disturbances included damming by a sizeable beaver population, as well as significant site structural changes from unanticipated human activity. Constructed wetlands are not exposed to these unplanned modifications and if changes do occur on this order, maintenance work is possible to rectify them. Lastly, the natural wetlands of this study do not require the intensive maintenance and upkeep required to sustain constructed wetland performance.

With the above differences in mind, concentrations of the studied stormwater contaminants in the influent to the intensive study wetlands were so low that it is unknown how the study wetlands would handle contaminant concentrations that constructed wetlands are exposed to.

4.3. Similarities Between Natural Intensive Study Wetlands and Constructed Wetlands

The natural urban wetlands in this study share an important characteristic with constructed wetlands, especially in the center of the wetlands. The types of vegetation found in marsh-like conditions along the major flow-paths of surface water in the study sites are similar to those used in constructed wetlands. Broadleaf cattail (*Typha latifolia*), green arrow arum (*Peltandra virginica*), common rush (*Juncus effusus*), softstem bulrush (*Schoenoplectus tabernaemontani*), and other emergents dominated, and these plant structures remove TSS through particulate interception during flow-through episodes or time periods. These species are popular in constructed treatment wetlands, though the common reed (*Phragmites australis*), a plant used extensively in engineered wetland sites, was missing from the natural wetlands in this study. The common reed is not generally considered native to the United States but is native to some countries that use it in their treatment wetlands. Another plant commonly found in both man-made wetlands and natural wetlands is the black willow (*Salix nigra*) used in “zero-discharge” treatment systems for small town sewage effluent (Kadlec and Wallace 2009). This plant was found in varying quantities in all four intensive study wetlands. Engineered treatment wetlands often display higher values of plant tissue phosphorous concentrations than natural wetlands because they are typically loaded with higher phosphorous influents.

This literature review showed that much work has already been done, and that continued study of both natural and constructed wetlands will be valuable. Such study will increase understanding and appreciation of physical and chemical transformations and other ecosystem services provided by natural wetlands, and will inform further improvements in systems designed, constructed, and maintained to mimic these services for water quality uplift.

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