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WATER QUALITY TRENDS IN THE NORTHERN COASTAL BASIN (1984-2023)

by

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EXECUTIVE SUMMARY

NORTHERN COASTAL BASIN OVERVIEW

The Northern Coastal Basin (NCB) extends from the Florida-Georgia border in Nassau County southward to the Ponce Inlet in Volusia County and contains a mix of Class II (shellfish harvest) and Class III (recreational use: swimming, fishing) surface waters. Extensive salt marshes, mangroves, and intertidal oyster reef habitats can be found throughout the NCB, which collectively support key ecosystem functions including improvements to water quality (WQ) through nutrient and pollutant filtration and sequestration (e.g., carbon) as well as shoreline stabilization. Estuarine systems throughout this region exhibit spatial variability, characterized by differences in hydrology, land use, and coastal habitat. For planning and management purposes, the NCB has been subdivided into eight Planning Units (PU) which includes the Lower St. Marys (2C), Nassau River (1A), Intracoastal Waterways (3K and 3I), Tolomato River (9D), Matanzas River (9C), Pellicer Creek (9B), and the Halifax River (9A; Fig. E.S.1). Estuaries in the northern NCB (e.g., St. Marys, Nassau) are characterized by greater tidal amplitudes and flushing, less development, and a predominance of salt marsh vegetation. In contrast, estuaries in the southern NCB (e.g., Halifax) exhibit greater development, smaller tidal amplitudes, and habitat that includes a mix of salt marsh and mangroves. These physical and ecological differences affect WQ dynamics (e.g., nutrient loading, residence time, biogeochemical cycling) across the NCB.

Good WQ is essential for healthy estuarine ecosystems and maintaining conditions that support sustainable oyster populations and resilient coastal habitats is a fundamental objective of NCB management. In general, coastal WQ concerns across Northeast Florida include impairments related to nutrients, phytoplankton (chlorophyll-*a*), dissolved oxygen (DO), heavy metals, and bacteria. Estuaries in developed watersheds face greater risk of impairment due to habitat loss, reduced ecosystem services, and other anthropogenic stressors (e.g., dredging, hydrological alterations, wastewater discharge/nutrient loading) that can degrade WQ. With populations throughout Nassau, Duval, St. Johns, Flagler, and Volusia Counties projected to increase by nearly 670,000 people across all five counties over the next 25 years (Rayer and Wang, 2024), WQ monitoring and trend analysis is imperative to identifying potential threats and developing management and resiliency strategies to protect coastal waters. The primary objective of this report is to:

i) Provide information on long-term physical (temperature, salinity, conductivity, pH, DO, Secchi) and chemical (total phosphorus, total nitrogen, dissolved nitrite and nitrate, dissolved organic carbon, color, total suspended solids, chlorophyll-*a*) WQ trends at each of the 29 monthly monitoring stations using all available data (9-39 years per station).

ii) Evaluate management concerns throughout the NCB.

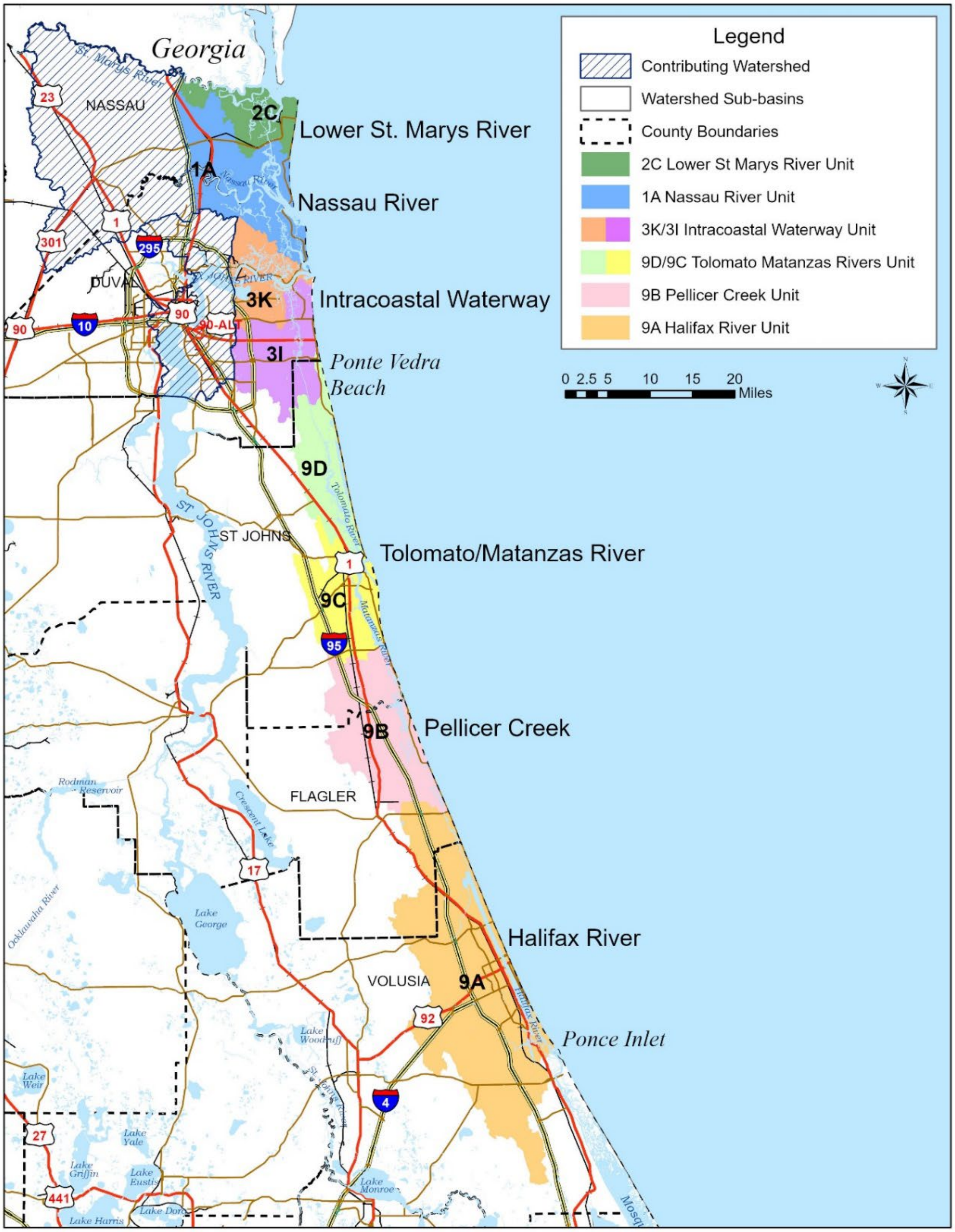

Figure E.S.1. Map depicting the eight NCB planning units which include the Lower St. Marys River (2C), Nassau River (1A), Intracoastal Waterway (3K, 3I), Tolomato and Matanzas Rivers (9D, 9C), Pellicer Creek (9B), and the Halifax River (9A).

SUMMARY OF WATER QUALITY TRENDS

Analysis of long-term WQ trends indicated management concerns related to:

i) **Increasing water temperature:** Water temperature increased in 5 of the 8 NCB PUs, with rising temperatures expected to reduce DO and intensify primary productivity and eutrophication, which may exacerbate associated WQ impairments. Warmer water temperatures can also affect estuarine community dynamics by increasing disease prevalence, altering growth and survivorship rates, and facilitating the northward expansion of subtropical species.

Salinity, pH, and DO trends varied spatially throughout the NCB, with both increases and decreases being found.

ii) **Changes in Salinity:** Salinity changes can indicate broad scale shifts in precipitation patterns and sea level dynamics, with both abrupt salinity shifts (e.g., freshwater pulses from storms) and gradual sea level rise presenting management concerns.

iii) **Declines in pH and DO:** Declines in pH can adversely affect oysters by impairing growth, development, feeding and reproduction, while also increasing the risk of shell dissolution. Similarly, low DO can elevate stress in aquatic communities, potentially leading to fish kills under hypoxic conditions, and can disrupt biogeochemical processes (e.g., nitrogen and phosphorus cycling) and alter sediment-water flux of nutrients and heavy metals. However, more information is needed to determine if changes in pH and DO are impacting oysters (e.g., health, habitat suitability) and other species across the NCB.

Overall, nutrient concentrations declined across the NCB, with some localized increases in total phosphorus (TP) or total nitrogen (TN). Chlorophyll-*a* (Chl-*a*) declines were more prevalent in the northern NCB, while increases were observed in southern NCB.

iv) **Nutrients and Chlorophyll-*a*:** Despite declining nutrient trends, many areas across the NCB remain impaired for nutrients and Chl-*a*, supporting the need for further nutrient reductions throughout the NCB. Eutrophication is a management priority since it directly contributes to the degradation of WQ. Harmful algal blooms resulting from excess nutrients can increase the risk of hypoxia, introduce toxins, decrease light attenuation, and negatively impact oysters (e.g., impaired feeding, disrupt spawning/reproduction cycles) and other aquatic animals. Eutrophication has also been linked to coastal habitat changes by contributing to salt marsh loss, increasing mangrove freeze resistance, and accelerating mangrove range-expansion in the NCB.

MANAGEMENT RECOMMENDATIONS

To effectively address WQ concerns and support data driven ecosystem management, there is a clear need to strengthen the current WQ monitoring network and improve how monitoring data informs ecosystem management priorities. This can be accomplished by:

i) **Identifying Data Deficiencies and Expanding Monitoring Coverage:** Establishing WQ monitoring stations in data deficient regions will fill data gaps and improve statistical analyses by strengthening result interpretation over broader spatial scales and enhancing statistical power. Collaborating with other agencies to find and fill these gaps can also prevent redundancies and leverage ongoing WQ management efforts through the alignment of shared goals between stakeholders.

ii) **Enhancing Analytical Capacity with Additional Water Quality Analyses:** Utilizing analytical methods, such as generalized additive models, which can assess changes in trend direction over time and incorporate multiple explanatory variables, can improve evaluations of management effectiveness and improve the identification of key drivers and interactions influencing WQ. Ensuring temporal consistency can also enhance the spatial comparison of trends across local and regional scales.

iii) **Oyster Reef Mapping and Increased Monitoring:** There is a critical need for up-to-date oyster habitat maps, which are essential for supporting WQ assessments and ecosystem health evaluations. Increased monitoring in Class II waters can also expand our understanding of how commercial and recreational harvest influence reef resilience and aid in management planning efforts designed to support sustainable oyster fisheries while maintaining ecosystem integrity.

iv) **Mangrove Mapping and Monitoring:** Establishing systematic mangrove mapping will support real time tracking of habitat expansion, which is essential for assessing changes in WQ resulting from salt marsh to mangrove habitat shifts and contribute to coastal resilience forecasting models.

Overall, these improvements will provide the foundation for adaptive management strategies that promote long-term WQ health and ecosystem health and resilience.

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LIST OF ACRONYMS AND ABBREVIATIONS

Chl-*a*	Chlorophyll-*a*
Cond.	Conductivity
DO	Dissolved oxygen
DOC	Dissolved organic carbon
DOM	Dissolved organic matter
EPA	Environmental Protection Agency
F.A.C.	Florida Administrative Code
FDACS	Florida Department of Agriculture and Consumer Services
FDEP	Florida Department of Environmental Protection
FIND	Florida Inland Navigation District
FLW	Florida LAKEWATCH
FWC	Florida Fish and Wildlife
ft	Feet
GADNR EPD	Georgia Department of Natural Resources Environmental Protection Division
GAM	Generalized Additive Model
GTM	Guana, Tolomato, and Matanzas
GTMNERR	Guana Tolomato Matanzas National Estuarine Research Reserve
HRPU	Halifax River Planning Unit
ICW	Intracoastal Waterway
ICWPUs	Intracoastal Waterway Planning Units
kg	Kilograms
km, km^2	Kilometer, square kilometer
KW	Kruskal–Wallis test

LSJRB	Lower St. Johns River Basin
LSMPU	Lower St. Marys Planning Unit
m	Meter
MDL	Method detection limit
mg/L	Milligrams per liter
mg/m^3	Milligrams per cubic meter
mi, mi^2	Mile, square mile
MRPU	Matanzas River Planning Unit
NCB	Northern Coastal Basin
NOx-D	Dissolved nitrite and nitrate
NRPU	Nassau River Planning Unit
OFW	Outstanding Florida Waterways
PCCSC	Palm Coast Community Service Corporation
PCPU	Pellicer Creek Planning Unit
PCU	Platinum-cobalt units
PU	Planning Unit
Sal.	Salinity
SI Unit	International System of Units
SJCUEL	St. Johns County Utility and Environmental Laboratory
SJR	St. Johns River
SJRWMD	St. Johns River Water Management District
SWIM Plan	Surface Water Improvement and Management
Temp	Temperature
TFE	Total fractional exposure

TMDL	Total maximum daily load
TN	Total nitrogen
TP	Total phosphorus
TRPU	Tolomato River Planning Unit
TS	Theil-Sen slope
TSS	Total suspended solids
UCF	University of Central Florida
US HW 17	United States Highway 17
VCEM	Volusia County Environmental Management
WBID	Waterbody identification
WQ	Water quality

CHAPTER 1. INTRODUCTION

The Northern Coastal Basin (NCB), located in northeast Florida, spans 193 km (120 mi) of coastline from the Florida-Georgia border southward through Nassau, Duval, St. Johns, Flagler, and Volusia counties, where it ends at the Ponce De Leon inlet (Fig. 1.1). The northeast coastal region of Florida is characterized by a series of shallow estuarine bays and lagoons interconnected by narrow, maintained dredged channels, and separated from the Atlantic Ocean by a barrier island system with seven inlets (the St. Marys, Nassau, Ft. George, St. Johns, St. Augustine, Matanzas, and Ponce De Leon). Tidal river and estuarine systems across the NCB include the St. Marys, Nassau, St. Johns, Guana, Tolomato, and Matanzas (GTM), Pellicer and Halifax. In Duval County, a portion of the NCB overlaps with the Lower St. Johns River Basin (LSJRB) with the St. Johns River falling within the purview of LSJRB management whereas the NCB primarily focuses on the estuarine areas to the north and south of the St. Johns River's confluence with the Atlantic Ocean.

Most rivers in northeast Florida are blackwater systems with low pH and dissolved oxygen (DO) and high color and dissolved organic carbon (DOC; Brown and Orell 1995, Gallegos 2005, Blair et al. 2009, Boning 2016). Within blackwater estuarine systems, seawater introduces salinity into tidally influenced regions and typically has higher pH and DO and lower color and DOC compared to blackwater (Williams and Kimball, 2013, Chaya et al. 2023). Hydrology is highly dynamic across the NCB, with estuarine salinity regimes, water residence times, and other water quality (WQ) parameters dependent on tidal amplitudes, freshwater inflow, precipitation, currents, wind and distance to an inlet (Phlips et al. 2004, Chaya et al. 2023). As a result, physical and chemical WQ parameters can be highly variable based on the amount of fresh and saltwater mixing, with increases in salinity typically associated with increases in pH and dissolved oxygen and decreases in color, DOC and other nutrients across NCB estuaries (Williams and Kimball, 2013, Chaya et al. 2023). Salinity regimes across the NCB estuaries and their respective tributaries include freshwater (> 0.5), oligohaline (0.5 – 5.0), mesohaline (<5.0 – 18.0), polyhaline (<18.0 – 30.0), and euhaline (< 30.0) conditions.

In addition to estuaries and inlets, another notable water feature throughout the NCB is the Intracoastal Waterway (ICW), which extends from the lower St. Marys River south to the Halifax River. The ICW was originally constructed between 1882-1912 to create a continuous inland waterway passage from Jacksonville to Miami (Abecassis 2005) and was later integrated into the Atlantic Intracoastal Waterway (Crawford 2006). The ICW within the NCB is a relatively narrow waterway with widths ranging from 27 to 45 m (90 to 150 ft), characterized by depths between 3 to 3.7 m (9.8 to 12.1 ft; Abecassis 2005, Kinnaird 1983) and has both natural and man-made portions across Nassau, Duval, St. Johns, Flagler, and Volusia counties.

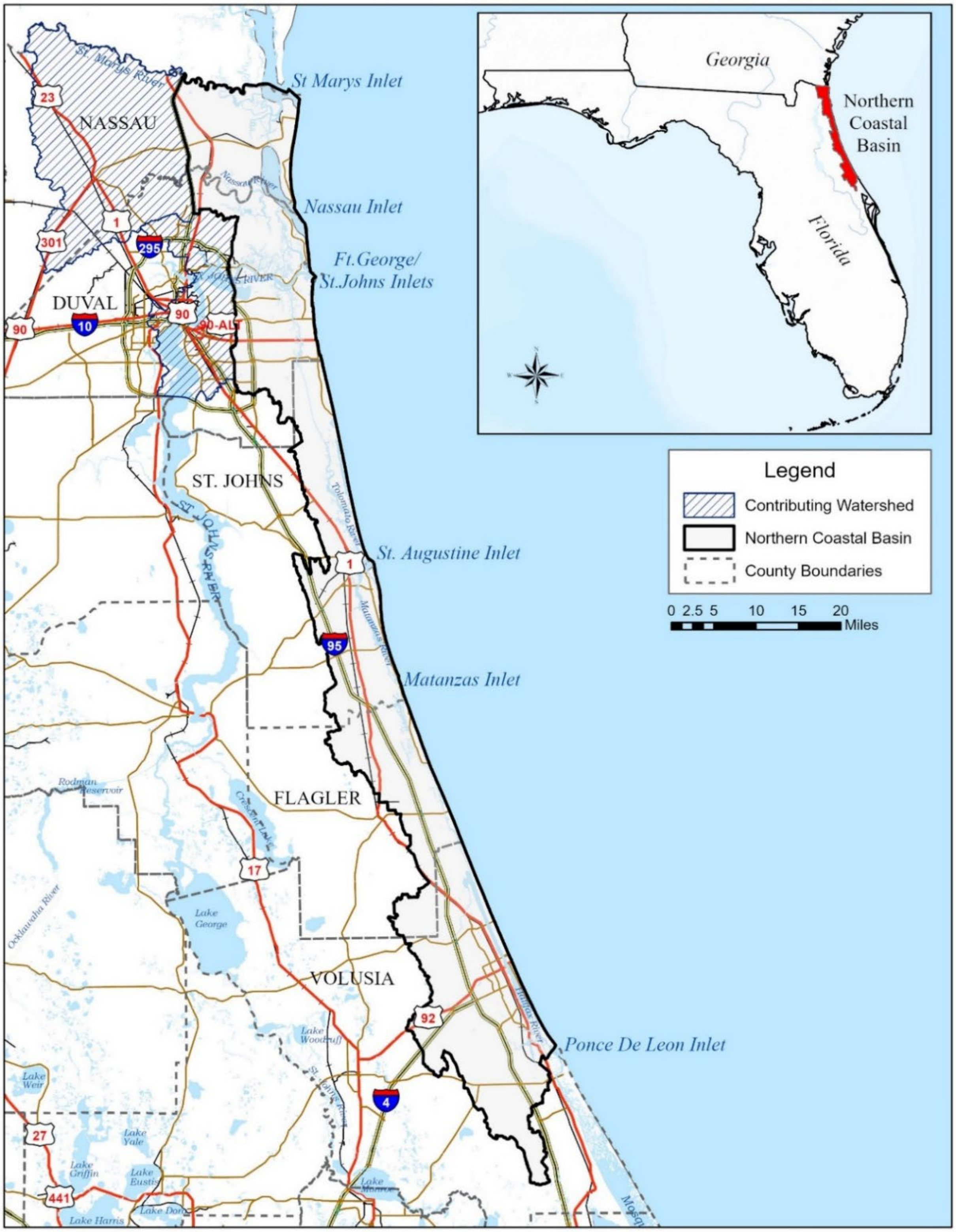

Figure 1.1.1. Map of the NCB.

Of the five counties across the NCB the two northern-most counties, Nassau and Duval, have the largest estuarine marsh system found on the east coast of Florida (Dix et al. 2021) and contain estuaries from the St. Marys, Nassau and St. Johns Rivers. The St. Marys and Nassau estuaries have large tidal amplitudes (2.0 m; 6.5 ft) near their inlets (Blair et al. 2009, Williams and Kimball 2013), which can increase daily water exchange and tidal flushing within these systems (Uncles et al. 2002). By comparison, the inlet to the St. Johns estuary and nearby portions of the ICW have a maximum tidal range of 1.5 m (4.9 ft; Bacopoulos et al. 2012, Dix et al. 2021).

The GTM Rivers, estuary, and Pellicer Creek span St. Johns and Flagler counties, which are among the fastest-growing counties in Florida (BEBR 2022). The GTM estuary has a tidal amplitude of 1.5 m (4.9 ft) near the St. Augustine inlet (Kimball and Eash-Loucks 2021), while the middle reaches of Pellicer creek have a smaller amplitude of 0.45 m (1.5 ft; Schafer et al. 2022). Loss of natural wetlands and pollution (e.g., urban runoff) due to increased population growth and development are major concerns for WQ within these estuaries. However, portions of the habitat surrounding the GTM estuary and Pellicer Creek are located within the Guana Tolomato Matanzas National Estuarine Research Reserve (GTMNERR). The GTMNERR spans 56 km (35 mi) north to south and includes 311 km^2 (120 mi^2) of undeveloped coastal and estuarine habitat (Dix et al. 2021) which is protected from future development.

In contrast, the Halifax River and estuary in Volusia County is highly developed, with nearly 80% of the County's population living within its watershed (Reiter and Cho 2020). Only a small portion of natural shoreline and wetland remains along the Halifax River, making urban runoff and associated pollutants a major concern for WQ within this system (Goolsby et al. 2016, Reiter and Cho 2020). The tidal amplitude for the Halifax is much smaller (0.6 m; 2 ft.) compared to other estuaries within the NCB such as the Lower St. Marys or Nassau, and alterations (e.g., canal construction, dragline ditching, dredging) to the watershed have had major impacts on circulation, tidal exchange, and water transport within this system (Cho et al. 2020).

1.1. NORTHERN COASTAL BASIN HABITATS

The temperate estuarine habitats found throughout the NCB are comprised of extensive salt marshes, mangroves, and intertidal oyster reefs which provide important ecosystem services including WQ improvement (e.g., nutrient assimilation, sediment trapping), carbon sequestration, shoreline protection (e.g., wave attenuation, erosion reduction, elevation maintenance) and fisheries habitat (Grabowski et al. 2012, Blair et al. 2015, Widney et al. 2018).

Salt Marsh and Mangroves

Extensive salt marsh and mangrove habitat can be found throughout the NCB, with Nassau and Duval counties dominated by salt marsh while St. Johns, Flagler, and Volusia counties in the south contain a mix of salt marsh and mangrove vegetation (Dix et al. 2019). Salt marsh and mangroves are highly productive wetland habitats which occupy similar geomorphic positions within tidal estuarine systems (Cahoon et al. 2020). From a functional perspective, salt-marshes and mangroves serve as foundation habitats, controlling ecosystem dynamics and facilitating the development of entire ecological communities. The types of ecosystem services (e.g., coastal protection, erosion control, water purification, carbon sequestration, fisheries habitat) provided by mangrove forests and marsh habitats differ in regard to the quality and level of service provided (Barbier et al. 2011). The ecotonal boundary between mangrove and salt marsh fluctuates within the NCB in response to environmental conditions. Cold–sensitive mangroves die back during freeze events and expand during warm periods, creating a temporally and spatially dynamic ecosystem (e.g., Cavanaugh et al. 2014, Osland et al. 2017). The current ecotone between mangroves and salt marshes in northeast Florida has shifted between mangrove and salt marsh dominance at least 6 times between the late 1700s and 2017 due to decadal-scale fluctuations in the frequency and intensity of extreme cold events (Cavanaugh et al. 2019). Additionally, coastal eutrophication has been shown to increase freeze resistance (Feller et al. 2023) and accelerate mangrove range-expansion in this system (Dangremond et al. 2020). Changes in dominant plant cover of this magnitude have the potential to significantly alter habitat structure, function, and landscape ecosystem services (Kelleway et al. 2017) which may also affect WQ in these areas.

Both habitats can enhance WQ through processes such as wave attenuation and removal of nutrients and other pollutants (Barbier et al. 2011, Blair et al. 2015, Widney et al. 2018). For example, wave attenuation promotes sediment deposition and retention which can improve water clarity (Sánchez-Núñez et al. 2020). Additionally, removal of nutrients (e.g., phosphorus, nitrogen) and other pollutants such as heavy metals can counteract eutrophication, reduce the potential occurrence of harmful algal blooms and mitigate the possible negative effects of heavy metals on aquatic life and human health (Yadav et al. 2023, Borgström et al. 2024). Conversely, coastal eutrophication is an issue of serious global concern and although nutrient subsidies can enhance the primary productivity of coastal wetlands, they can be detrimental to their long-term maintenance. These systems have been shown to increase productivity and alter biomass allocation when nutrient limitation is alleviated (Feller 1995; Feller et al. 2003; Lovelock et al. 2004; Reef et al. 2010; Deegan et al. 2012), which may reduce resilience and recovery when exposed to other environmental stressors and extreme events (Lovelock et al. 2009; Feller et al. 2015).

Oysters

The eastern oyster, *Crassostrea virginica*, is a native bivalve species typically found along intertidal zones throughout the eastern U.S., including northeast Florida (Dix et al. 2019). Oyster reefs are a living resource that contribute to coastal resilience by providing valuable ecosystem services including improvement of WQ (e.g. removal of nutrients) and water clarity, protection for coastal ecosystems, shorelines, and waterfront communities, carbon sequestration, as well as food and habitat for estuarine animals (Coen et al. 2007; Scyphers et al. 2011, Kellogg et al. 2014, Volety et al. 2014, Garvis et al. 2020). For example, within the GTM estuary in St. Johns County, oyster reefs filter approximately 60% of the estuary's total water volume within a single water residence period and play an important role in maintaining and improving WQ within the system (Gray et al. 2021). Additionally, oysters are also an important component of northern Florida's recreational and commercial fisheries economy. In 2023, oysters had an annual estimated value of $174,426 for St. Johns County based on Florida Fish and Wildlife's (FWC) Commercial Fisheries Landing Summary.

Globally, oyster populations are declining with an estimate loss of 85% of the world's oyster reefs resulting from stressors such as anthropogenic hydrologic alterations, eutrophication, boating traffic (e.g., boat wake), increasing water temperature, acidification and overharvesting (Lotze et al. 2006, Beck et al. 2011, Knoell et al. 2021, Kimmel et al. 2024). In Florida, freshwater diversion from upland watersheds has contributed to the loss of over 90% of historic oyster beds on the Atlantic coast (Beck et al. 2011). The severe loss of oyster reef coverage, coupled with shellfish harvesting closures due to WQ concerns, has encouraged conservation, mapping, monitoring and restoration efforts throughout Florida. In 2015, SJRWMD and University of Central Florida (UCF) mapped oysters to document the existing footprint within the region, which can be used to examine negative impacts to reef area over time. Mapping resulted in the identification of 17,953 individual reefs, covering 652 ha, with a mean reef area of 1,618 m^2 across the NCB (Table 1.1.1; Garvis et al. 2020). Of these reefs, 6.1% were classified as dead, all of which were along important boating channels (Dix et al. 2019, Garvis et al. 2020). Additionally, the GTMNERR has conducted several oyster monitoring projects in the GTM Rivers. From 2014 to 2016, the GTMNERR conducted monitoring and observed an average oyster density of 1,621 m^2 across the system (Marcum et al. 2018). An oyster monitoring program was subsequently implemented which split the GTM estuary into upper, middle and lower regions, with data collected for each section once every three years (Kimmel et al. 2024).

Table 1.1.1. Summary of NCB oyster reef coverage and area by county.

County	Total Extent (ha)	Reefs (#)	Mean Reef Area (m^2)
Nassau	158.11	6,813	232.07
Duval	103.67	3,561	291.13
St. Johns	331.81	4,848	684.43
Flagler	18.13	1,099	164.97
Volusia	40.13	1,632	245.89
Total	**651.85**	**17,953**	**1618.49**

Data table adapted from Garvis et al. 2020. Total extent, reefs, and mean reef area include live (aggregate and continuous) and dead oyster reefs.

1.2. Northern Coastal Basin Management History

The Northern Coastal Basin (NCB) Project was first initiated by the St. Johns River Water Management District (SJRWMD) in 1995 in response to closures of shellfish harvesting in St. Johns County and general WQ concerns across the NCB. The potential for negative impacts to WQ related to current and anticipated population growth and coastal development highlighted a need for a dedicated program to monitor and address surface water concerns within the NCB (SJRWMD 1997). In 2003 SJRWMD developed the NCB Surface Water Improvement and Management (SWIM) Plan (Haydt and Frazel 2003) to guide efforts to restore, protect, and manage WQ and coastal resources within the Basin.

The original NCB project and 2003 SWIM Plan established a WQ monitoring network from Ponte Vedra Beach to Ponce Inlet, with monitoring stations being spread across four planning units (PUs). The PUs are subdivisions of the 10 major basins managed by SJRWMD, consisting of individual or grouped tributary basins with similar characteristics, and are used for organizing planning and management efforts (Adamus et al. 1997). Each PU is assigned a number corresponding to its respective major basin and letter which distinguishes it from other subdivisions. The four PUs originally included in the 2003 SWIM Plan were the Tolomato River (9D), Matanzas River (9C), Pellicer Creek (9B), and Halifax River (9A; Fig. 1.2.1). In 2014, the geographic range of the NCB was expanded northward from Ponte Vedra Beach to the Florida-Georgia border as part of a strategic priority initiative in the SJRWMD 2015-2019 Strategic Plan. The expansion of the NCB boundary included four additional PUs; Lower St. Marys (2C), Nassau River (1A), and Intracoastal Waterways (3K, 3I; Fig. 1.2.1).

The NCB boundary currently includes eight PUs, spanning five counties encompassing various local, state, and federal lands with overlapping jurisdictional boundaries and responsibilities. In addition to SJRWMD there are multiple agencies, entities and stakeholders that have programs or plans that directly relate to the core initiatives of the 2003 NCB SWIM Plan (e.g., GTMNERR and Volusia County Environmental Management (VCEM)). The 2003 NCB SWIM Plan initiatives include i) WQ, ii) watershed master planning, iii) stormwater retrofit and master plan implementation, iv) compliance and rules and v) resource assessment, protection and restoration. The goal of the WQ initiative was to evaluate available WQ data, assess monitoring efforts and determine whether gaps in the data collection and synthesis exist, identify areas for potential enhancement and management and establish targets for WQ and habitat protection. Additional goals established in the SJRWMD 2015-2019 Strategic Plan, which were driven by the 2003 NCB SWIM Plan's WQ initiative, were to expand the NCB's WQ motoring network and address threats (e.g., WQ impairments, degraded oyster habitat) to estuarine WQ and habitats. Ongoing monitoring and analysis of WQ data in the NCB is essential for tracking spatial and temporal changes and identifying potential impairments and threats to coastal waters and habitats in northeast Florida.

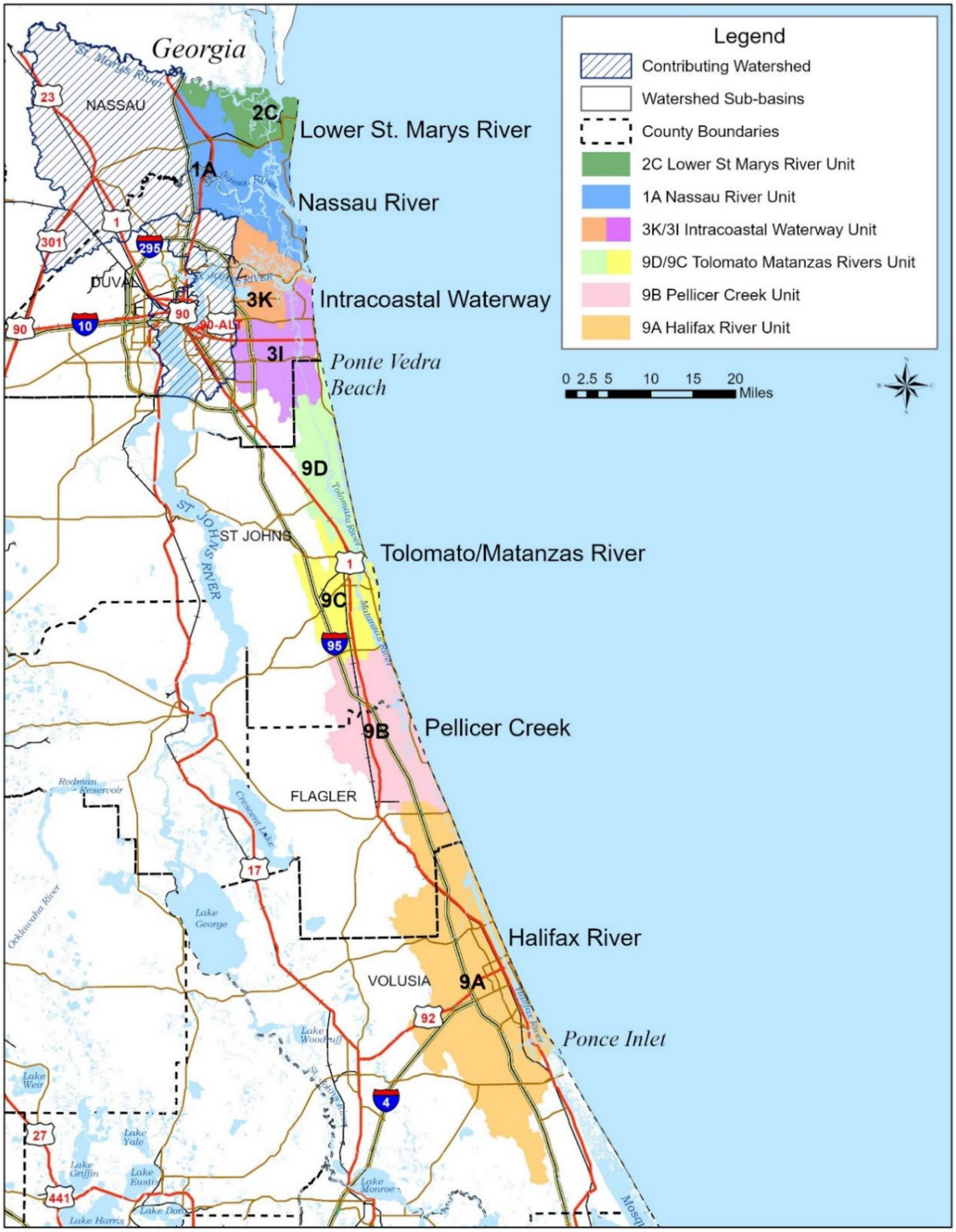

Figure 1.2.1. Map depicting the eight NCB PUs which include the Lower St. Marys River (2C), Nassau River (1A), Intracoastal Waterway (3K, 3I), Tolomato and Matanzas Rivers (9D, 9C), Pellicer Creek (9B), and the Halifax River (9A).

1.3. Impaired Waterbodies

Protecting and restoring WQ is a core mission of the SJRWMD and a primary objective of the 2003 NCB SWIM Plan. Monitoring WQ is necessary to identify WQ impairments and provide a foundation for resource management aimed at maintaining, protecting and restoring Florida's estuarine waters. The collection and analysis of WQ information throughout the NCB is accomplished through multiple state and regional level monitoring programs including SJRWMD, Florida Department of Environmental Protection (FDEP) and Florida Department of Agriculture and Consumer Services (FDACS). Together, these programs help determine if waterbodies meet current surface WQ standards, assess which waterbodies are at risk of impairment and target specific monitoring needs to address local concerns.

A waterbody (or waterbody segment) is considered impaired if it does not currently meet applicable surface WQ standards based on its intended use (Table 1.3.1). In addition to these classifications, certain waterbodies, like Pellicer Creek Aquatic Preserve, are designated as Outstanding Florida Waterways (OFW) under Florida Administrative Code (F.A.C.) Rule 62-302.700. This designation, given to waterbodies with excellent WQ or exceptional ecological, social, educational, or recreational value, imposes stricter requirements for permitted activities such as wastewater discharge or dredging. The goal of the OFW designation is to protect and preserve high WQ conditions and prevent degradation of these exceptional waterbodies.

Table 1.3.1. List of Florida surface waterbody classifications.

Class I	Potable water supplies.
Class I - Treated	Treated potable water supplies.
Class II	Shellfish propagation or harvesting.
Class III	Fish consumption; Recreation, propagation and maintenance of a healthy, well-balanced population of fish and wildlife.
Class III - Limited	Fish consumption; Recreation or limited recreation; and/or propagation and maintenance of a limited population of fish and wildlife.
Class IV	Agricultural water supplies.
Class V	Navigation, utility and industrial use.

Table adapted from 62-302.400. F.A.C.

Section 303(d) of the U.S. Clean Water Act requires that states submit a list of waters that do not meet sufficient WQ standards [62-302, F.A.C.] for their designated uses to the U.S. Environmental Protection Agency (EPA) for approval every two years. In Florida, FDEP assesses WQ standards every two years across waterbody identification (WBID) segments within each of the four major NCB PU basins (St. Marys River, Nassau River, Lower St. Johns River, and Northern Coastal). Based on FDEP's 2022-2024 Comprehensive Verified List and 2022-2024 WQ Assessment, TMDLs, and BMAPs, there are 65 impaired WBIDs throughout the NCB PUs, which fall under four general categories; bacteria (e.g., fecal coliform), nutrients/eutrophication (e.g., nitrogen, phosphorus, Chl-*a*), metals (e.g., aluminum, iron, copper) and DO.

Impaired water bodies (WBIDs) that do not meet WQ standards are recommended to a have total maximum daily load (TMDL) developed and implemented based on FDEP's prioritization framework (TMDL Prioritization 2.0 | Florida Department of Environmental Protection). A TMDL is the maximum determined amount of a pollutant that can be absorbed by a given surface waterbody while still meeting surface WQ standards. It is designed to reduce target pollutants and restore the affected waterbody for its intended class. Out of the eight NCB PUs there are TMDLs within three; the Intracoastal Waterway (3K, 3I), Pellicer Creek (9B), and Halifax River (9A) PUs (Table 1.3.2). Additionally, there are nutrient TMDLs within the lower St. Johns River which partly overlap with the Intracoastal Waterway (3K, 3I) PUs. However, the St. Johns River is not a focal point of this report.

Table 1.3.2. List of impaired water bodies (WBIDs) with total maximum daily loads (TMDLs) within the four NCB PUs.

PU (ID)	WBID	TMDL
Intracoastal Waterway (3K, 3I)	Sherman Creek (2227)	Fecal coliforms
	Open Creek (2299)	Fecal coliforms
	Hopkins Creek (2266)	Fecal coliforms
	Greenfield Creek (2240)	Fecal coliforms
Pellicer Creek (9B)	Pellicer Creek (2580B)	Fecal coliforms
	Palm Coast (2363D)	Nutrient (Chl-*a*)
Halifax River (9A)	Tomoka River (2634)	Nutrient (TN, TP)
	Spruce Creek (2674)	Fecal coliforms
	Spruce Creek (2674A)	DO and nutrient (TP)
	Halifax River (2363B)	Nutrient (TN)

1.4. WATER QUALITY MONITORING PROGRAM

Since the NCB Project was first initiated in 1995 the number of active WQ monitoring stations has expanded and contracted as program priorities, initiatives, and partnerships have evolved. When initially determining the locations of monitoring stations, areas with pre-existing WQ data from other projects or agencies (e.g., FDEP) dating back to the 1980s were prioritized. In the 1990s SJRWMD developed cooperative agreements with multiple counties and agencies, including the St. Johns County Utility and Environmental Laboratory (SJCUEL), Palm Coast Community Service Corporation (PCCSC), and Volusia County (VCEM) to establish coordinated efforts to monitor WQ with standardized methods, procedures, and equipment across the four original NCB PUs. Partnerships were first established in 1993 with VCEM, and in 1997 with PCCSC and SJCUEL (Holub and Associates, 1998). By 1997 there were 59 active WQ monitoring stations across the Tolomato River (9D), Matanzas River (9C), Pellicer Creek (9B), and Halifax River (9A) PUs. Out of the 59 stations, WQ data was collected by SJRWMD at seven stations, SJCUEL at six stations, PCCSC at 14 stations, and VCEM at 32 stations. While many stations were newly established by SJRWMD and partner organizations throughout the 1990s, data collection at several WQ monitoring stations in the Matanzas River (9C), Pellicer Creek (9B), Tolomato River (9D), and Halifax River (9A) (n = 3) PUs date as far back as the early 1980's.

The SJRWMD 2015-2019 Strategic Plan expanded the geographic footprint of the NCB and established eight new WQ monitoring stations throughout the Lower St. Marys (2C), Nassau River (1A), and Intracoastal Waterways (3K, 3I) PUs and continued monitoring efforts at an additional two stations with pre-existing data from the early 1990s. These stations were added to improve the WQ monitoring network throughout the NCB and aid in the maintenance, protection and restoration of coastal waters in northeast Florida. As of 2024, there are 29 active SJRWMD WQ monitoring stations across the eight NCB PUs and contributing watersheds (Fig. 1.4.1). Out of the 29 stations, 25 are located within the NCB boundaries and 4 are located within the contributing watersheds.

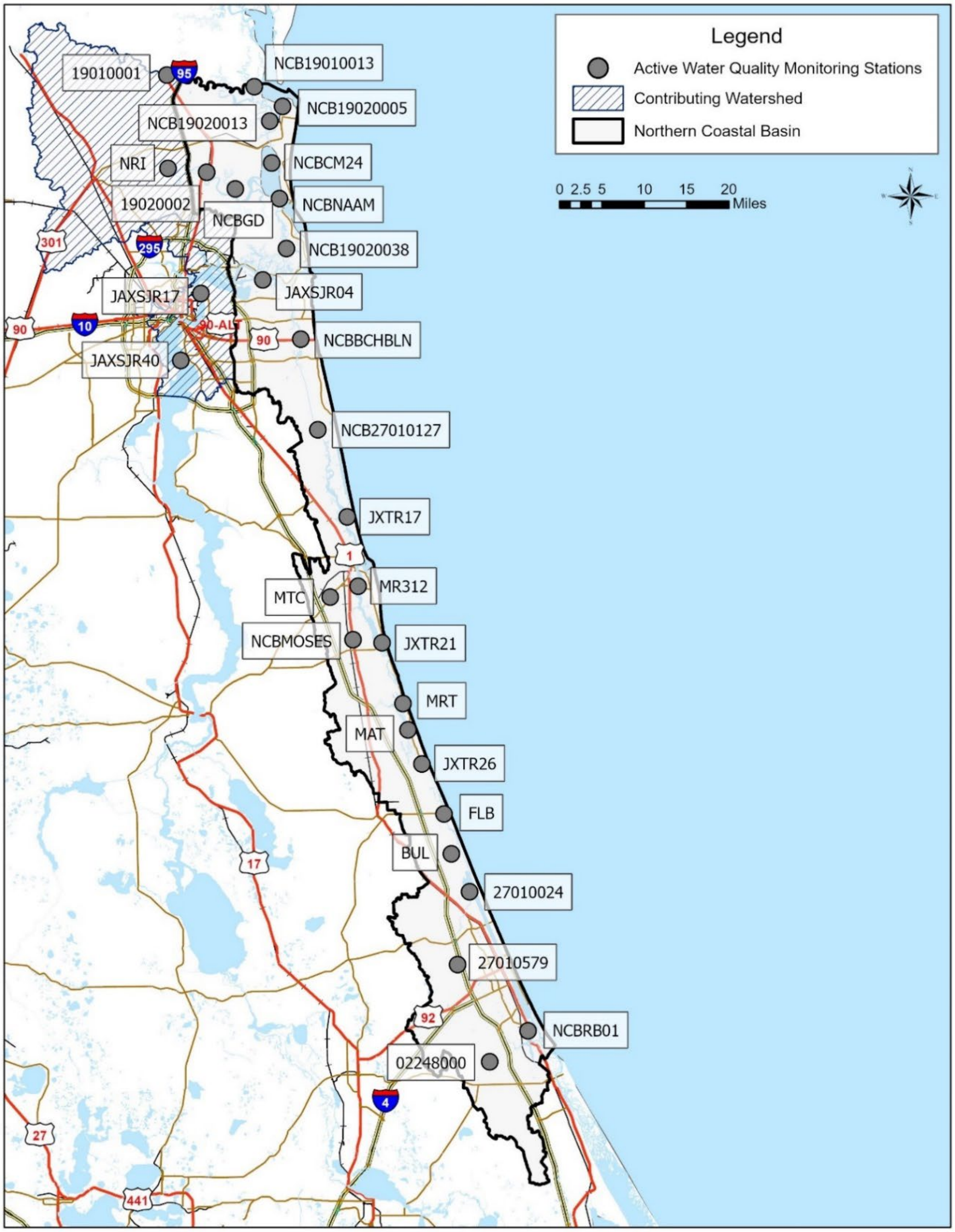

Figure 1.4.1. Map depicting 2024 active SJRWMD WQ monitoring stations across the NCB.

1.5. Goals and Objectives

Good WQ is a key component of healthy estuarine ecosystems. Maintaining good WQ supports diverse communities of plants and wildlife and can enhance ecosystem services, recreational activities, and other socioeconomic factors (e.g., healthy fisheries). Physical and chemical WQ parameters such as nutrients (nitrogen and phosphorus), Chl-*a*, water transparency, DO, and pH can be used as indicators of stress or poor WQ (Sheldon and Alber 2011). Consistent monitoring, processing and analyzing WQ data is essential in determining impairments and potential threats to respective waterbodies, and for developing management and strategies to enhance resilience.

The NCB is a dynamic system with varying hydrological characteristics, habitats, and land uses. The overarching goal of this report is to evaluate WQ trends of thirteen physical and chemical parameters within each of the eight NCB PUs based on available data from SJRWMD. Additional objectives are to i) disseminate information on existing conditions and evaluate WQ management concerns throughout the NCB and ii) provide recommendations for future management and monitoring of NCB WQ and coastal habitats which contribute towards maintaining good WQ and healthy ecological conditions.

CHAPTER 2. METHODS AND ANALYSES

2.1. WATER QUALITY PARAMETERS

Analyses focused on key physical and chemical parameters (Table 2.1.1) which have been highlighted as important indicators for evaluating estuarine WQ and other hydrological conditions (Coffin et al. 1992, Sheldon and Alber 2011, Dix et al. 2021). Parameters such as temperature, salinity (or conductivity) and DO can influence the spatial distribution and metabolic processes (e.g., growth, biological oxygen demand) of plants and animals within estuaries. Additionally, pH can impact the growth and survivorship of aquatic organisms such as oysters (Waldbusser et al. 2011a, Keppel et al. 2016) and influence the solubility of metals. Excess nutrients, especially nitrogen and phosphorus, can lead to harmful algal blooms and hypoxia within estuarine systems (Howarth et al. 2011), and alter growth of wetland plant species (Dangremond et al. 2020, Feller et al. 2023). Parameters such as color, TSS, Secchi depth and Chl-*a* can also be used as indicators for water clarity which influences aquatic vegetation health (Smith et al. 2006), bivalve filtration rates (Galimany et al. 2020), predator-prey interactions and abundance of fish (Minello et al. 1987, Benfield and Minello, 1996, Lunt and Smee 2014).

Table 2.1.1. List of physical and chemical parameters used to evaluate WQ across the NCB.

Physical Parameters	Chemical Parameters
Water temperature (Temp., °C)	Total phosphorus (TP, mg/L)
Salinity (Sal.)	Total nitrogen (TN, mg/L)
Conductivity (Cond., μmhos/cm)	Dissolved nitrite and nitrate (NOx-D, mg/L)
pH (SI units)	Dissolved organic carbon (DOC, mg/L)
Dissolved oxygen (DO, mg/L)	Color (PCU)
Secchi depth (m)	Total suspended solids (TSS, mg/L)
	Chlorophyll-*a* (Chl-*a*, mg/m^3)

Respective abbreviations and measurement units are shown in parentheses. Salinity values are presented using the unitless Practical Salinity Scale (Lewis 1980).

2.2. Sample Collection

At each active SJRWMD WQ monitoring station, ambient physical and chemical data was collected monthly, with stations in the St. Marys (2C), Nassau (1A), and Intracoastal Waterway (3K) PUs being sampled on the outgoing tide. Physical parameters were recorded at 0.5 m (1.6 ft) above the sediment surface and at 0.5 m below the water surface. Water samples for chemical analyses were collected at 0.5 m below the water's surface.

2.3. DATA ANALYSIS

Linear regressions and correlations (e.g., Pearson's *r*, Spearman's rho, and Kendall's tau) are among the most widely used statistical approaches for evaluating WQ data (Schreiber et al. 2022). While parametric linear regression models can be powerful analytical tools, they are also constrained by the assumptions of linearity, homoscedasticity (equal variance) and normal distribution of residuals (Helsel et al. 2020). Field data often violates one or more of these assumptions, meaning that alternative analyses, such as non-parametric tests, should be considered (Schreiber et al. 2022). For example, the non-parametric Mann-Kendall test does not assume linearity or normal distribution of residuals (Mustapha 2013, Helsel et al. 2020) and may be used to determine if a given WQ parameter is monotonically (consistently) increasing or decreasing over time. Since Mann-Kendall tests are rank-based they are robust to outliers and Seasonal Kendall tests can be utilized to account for data with seasonal variability (Mustapha 2013, Helsel et al. 2020).

For NCB WQ data, trends in physical and chemical parameters across years were investigated for individual stations through Mann-Kendall and Seasonal Kendall tests in R (version 4.3.1) using the "kendallTrendTest" and "kendallSeasonalTrendTest" functions through the package EnvStats (Millard 2013). Trends in WQ data across years were initially investigated using linear regressions, however most analyses violated test assumptions. Prior to running the Kendall trend tests, a Kruskal-Wallis (KW) test was used to determine if there were significant differences in WQ parameters across months using the built in R function "kruskal.test." If no differences were detected ($p \geq 0.05$) a Mann-Kendall test was run or a Seasonal Kendall if significant differences were found ($p < 0.05$). If a Seasonal Kendall test was run, the test assumption of homogeneity in trend direction across seasons (months) was also checked using a Van Belle-Hughes Heterogeneity Test. If significant heterogeneity ($p < 0.05$) was found, individual Kendall trends were run for each month. The strength of monotonic changes (trends) based on Kendall's tau were interpreted as low for tau values $< |0.3|$, moderate for tau values between $|0.3| \leq$ and $< |0.5|$, and strong for values $\geq |0.5|$ (Helsel et al. 2020, Schaeffer et al. 2022).

After running the KW and respective Mann-Kendall or Seasonal Kendall tests, trendlines were plotted for water temperature, salinity, DO, TP, TN and Chl-*a* for each station. These parameters were plotted for the purpose of comparing DO, TP, TN, and Chl-*a* trendlines to WQ and nutrient criteria values, which can aid in determining if a location may be at risk of developing WQ impairments or should be prioritized for specific management goals. Emphasis was also placed on the trendlines and data for water temperature and salinity at each station, since water temperature can impact DO concentrations and metabolic rates (e.g., phytoplankton growth) and freshwater inflow can also influence DO, nutrient and Chl-*a* concentrations within blackwater estuarine systems.

Based on the U.S. EPA and other scientific studies (Vaquer-Sunyer and Duarte 2008, Schmidt et al. 2019), hypoxic conditions within estuaries begin when DO concentrations fall between 2.0-3.0 mg/L. Therefore, a hypoxia stress value of 3.0 mg/L was used as the WQ criteria value for DO. Nutrient criteria values for TP, TN and Chl-*a* were based on F.A.C. Rule 62-302.531(2)(c) for streams and F.A.C. Rule 62-302.532 for estuaries. At several stations nutrient criteria values were not reported because values were expressed as annual nutrient loads rather than specified concentrations or they had not been established (e.g., streams typically do not have Chl-*a* criteria). In this report, Chl-*a* is presented as mg/m^3 which is equivalent to µg/L (1:1 conversion) – the unit used in the F.A.C. regulations. Additionally, information on impaired WBIDs throughout the NCB was based on FDEP's 2022-2024 Comprehensive Verified List and ArcGIS shapefiles for FDEP's 2022-2024 WQ Assessment, TMDLs, and BMAPs (downloaded on October 14, 2024).

For the plots, trendlines for Mann-Kendall tests were based on the Theil-Sen Estimator (Sen 1968, Theil 1992) for slope and the Conover Estimator (Granato 2006) for intercept, which are included in the "kendallTrendTest" output in R. Both estimators for slope and intercept are median-based measures and robust against outliers (Granato 2006, Helsel et al. 2020). Trendlines for Seasonal Kendall tests were based on the Hirsch et al. (1982) modification of the Theil-Sen Estimator for slope, which is computed as the median of all two-point slopes computed within each season and is included in the "kendallSeasonalTrendTest" output in R. Conover's Estimator (Granato 2006) for intercept was also used for the trendlines of the Seasonal Kendall tests.

Data for individual stations and parameters were inspected for data gaps before running the KW test and respective trend analyses. While Kendall tests can handle some missing data, large amounts of missing data, systematic differences in data collection, or changes in data distribution (variability) can influence test results (Helsel et al. 2020). If a large gap in data collection was found (e.g., ≥ 3 years missing), then a subset of data from the most recent sampling period following the missing data gap was used. Several stations throughout the NCB had systematic differences in sampling frequency, with data collection beginning as bimonthly (every other month) and later switching to monthly. In cases where majority of the data collected was bimonthly relative to the period of record, data was aggregated into bimonthly bins (e.g., January and February, March and April, etc.). Following Helsel et al. (2020), the median value was calculated for the bimonthly bins during years when WQ data was collected at a higher frequency (monthly). If only a small amount of data (e.g., < 3 years) was collected as bimonthly relative to the period of record, then the bimonthly data was removed.

Additionally, values for chemical WQ parameters that were analyzed but not detected (non-detects) were reported at the method detection limit (MDL), which is acceptable for datasets with non-detection rates below 25% (Croghan and Egeghy 2003). MDLs can change based on sample preparation (e.g., dilution) and instrumental capabilities, which may be an important consideration when interpreting trend results as exogenous factors that influence data variability can affect the power (ability) of Kendall tests to detect changes over time (Meals et al. 2011, Helsel et al. 2020). Details on any data that was removed due to data gaps, missing data, or systematic differences in data collection and/or notable changes in data distribution across time will be discussed in Chapter 3 within the respective PU sub-chapters.

CHAPTER 3: WATER QUALITY

3.1. LOWER ST. MARYS PLANNING UNIT

Summary

- There are four SJRWMD WQ monitoring stations located within the Lower St. Marys Planning Unit (LSMPU) (2C) and watershed.
- Mean WQ parameters were similar among the three eastern stations in the lower region of the estuary. Mean salinity at these three stations indicated marine (euhaline) conditions.
- Mean conditions at the station in the upper region of the estuary were oligohaline and displayed characteristics (low pH, high DOC, and high color) typical of blackwater estuarine systems.
- Each station showed at least two WQ trends, but the changes over time were weak (tau < 0.3). Trends include an increase in water temperature and decreases in salinity, DO, TN, and TSS at the station in the upper region of the estuary and decreases in DO, TP, TN, TSS and/or Chl-*a* at one or more stations in the lower region of the estuary.
- Increasing water temperature can be explained by an increase in warm weather conditions and decreasing salinity to changes in freshwater inflow. Decreases in nutrients and Chl-*a* are most likely explained by ongoing management efforts to reduce point and non-point sources of pollution into LSMPU.
- Increasing water temperature and decreasing DO concentrations can affect biological communities (e.g., growth, survivorship) and ecosystem function. Continued monitoring of these parameters is important since several regions of the estuary are impaired for DO and warmer water conditions can exacerbate naturally low DO concentrations in the upper estuary.
- Despite evidence of declining nutrient and Chl-*a* concentrations, continued management efforts to reduce nutrient loading are needed to address current DO, TP, and Chl-*a* impairments in the lower estuary.

Introduction

The St. Marys River is a 209 km (130 mi) long blackwater river system, with its head waters originating in the Okefenokee Swamp in Georgia (GADNR EPD 2002). The 3,367 km^2 (1,300 mi^2) basin drains into Atlantic Ocean through the St. Marys Inlet which is located along the Florida-Georgia state border near Cumberland and Amelia Islands (GADNR EPD 2002). The lower third of the St. Marys River is brackish, with maximum freshwater discharge up to 170 m^3/sec. (6,000 ft^3/sec.) at low tide and a spring tidal range of 1.9 m (6.4 ft; Blair et al. 2009, Dix et al. 2021). The river is home to 52 species of fish including the endangered Atlantic Sturgeon (*Acipenser oxyrinchus oxyrinchus*; GADNR EPD 2002, Fox et al. 2018) and the habitat surrounding the watershed is home to a myriad of animals including the Florida Black Bear (*Ursus americanus floridanus*), Southeastern American Kestrel (*Falco sparverius paulus*), red cockaded woodpecker (*Leuconotopicus borealis*), and gopher tortoise (*Gopherus polyphemus*; Blair et al. 2009). Additionally, the St. Marys Inlet is an important passageway for naval, commercial and recreational vessels (Blair et al. 2009) and is designated as Class III waterbody by FDEP.

The Lower St. Marys River Planning Unit (LSMPU) (2C) is confined to the northern portion of Nassau County, with its northern boundary extending to the Florida-Georgia state border and southern boundary through the northern portion of Amelia Island (Fig.3.1.1). The western LSMPU boundary borders I-95, however the Lower St. Marys watershed, and brackish region of the estuary, extends past this boundary. Extensive salt marsh, dominated by *Spartina alterniflora*, can be found throughout the Lower St. Marys watershed (Blair et al. 2009). Fort Clinch State Park, located on the northern tip of Amelia Island, comprises 5.7 km^2 (2.2 mi^2) of maritime hammock and coastal habitat. Much of the upland forest surrounding the Lower St. Marys watershed has been used for silviculture, with slash pine (*Pinus elliottii*) or loblolly pine (*Pinus taeda*) being grown for paper production (Blair et al. 2009). Overall, the LSMPU has seen less development compared to other NCB PUs like the Halifax River, with urban development primarily occurring along Amelia Island (e.g., Fernandina Beach).

Methods

For a full description of WQ parameters and analyses performed refer to Chapter 2. Several deviations to data management are as follows: At the westernmost station (19010001) sampling frequency of physical and chemical WQ parameters differed by period, occurring bimonthly from 1995-1997 and monthly from 1998-2023. To maintain consistency in trend analysis, bimonthly data from 1995-1997 was removed, and Chl-*a* data analysis was limited to 2007-2023 due to a large data gap from 1999-2006. While the westernmost station is outside the LSMPU boundary and is considered a stream based on F.A.C. regulations, it was included in the analyses since it overlaps with the brackish region of the St. Marys River.

Figure 3.1.1. Map of LSMPU (2C) (green). Red circles indicate SJRWMD WQ monitoring stations within the PU and contributing watershed.

Results

Mean Water Quality Conditions

There are four SJRWMD WQ monitoring stations located within the LSMPU and watershed (Fig.3.1.1). The westernmost station (19010001) located in St. Marys River west of US Highway (HW) 17 is the longest running station with data starting in 1998, while the other three stations began data collection in late 2014 (Table 3.1.1, Table 3.1.2). Of the four WQ stations, those within the PU located near Cumberland Sound (NCB19010013), Amelia River north (NCB19020005) and Amelia River south (NCB19020013), hereinafter referred to as the “NCB stations,” had similar physical and chemical WQ parameters with mean salinity across stations being euhaline (Table 3.1.1, Table 3.1.2). At the westernmost station, mean water temperature, DO, TP and NOx-D were similar to the three NCB stations, while salinity (oligohaline), conductivity, pH, Secchi depth, TSS, and Chl-*a* were lower, and TN, DOC, and color were higher (Table 3.1.1, Table 3.1.2).

Monthly Variability

Results from the KW tests indicated there was variation in one or more WQ parameters across months at each station. For the westernmost station significant differences ($p < 0.05$) were found between two or more months for all physical WQ parameters except Secchi depth and for TP, TN, and TSS (Table 3.1.3). At the three NCB stations, KW tests were significant for water temperature, DO, and NOx-D (Table 3.1.3). Additionally, differences across months were found at Cumberland Sound station for TN and TSS, at the northern Amelia River station for salinity and conductivity, and at the southern Amelia River station for pH, TP, TN, and Chl-*a*. KW tests were marginally significant for conductivity at the Cumberland Sound station and TSS at the northern Amelia River station.

Station Water Quality Trends and Criteria

Trend analyses at the **westernmost station (19010001)** detected an upward trend in water temperature over time (Table 3.1.3), with values increasing by 0.9°C (21.5°C to 22.4°C) from 1998 to 2023 (Fig. 3.1.2A). Declines in salinity, DO, TN, and TSS were also observed (Table 3.1.3), with salinity decreasing from 1.5 in 1998 to 1.0 in 2023 (Fig. 3.1.2B). The DO trendline and most recorded concentrations remained above the 3.0 mg/L hypoxia threshold (Fig. 3.1.2C). Most individual TP and TN concentrations were below their respective numeric nutrient criteria values (Fig. 3.1.2D-F), with TN decreasing from 1.1 mg/L in 1998 to 0.8 mg/L in 2023. No trend was found for Chl-*a*, with most individual concentrations being below 7.5 mg/m^3.

Table 3.1.1. Mean (±) standard deviation of physical WQ parameters for stations within the LSMPU.

Station	Years	Temp. (°C)	Sal.	Cond. (µmhos/cm)	pH	DO (mg/L)	Secchi (m)
19010001	1998 - 2023	21.45 ± 6.03	4.05 ± 5.25	6900.08 ± 8587.69	6.22 ± 0.66	5.73 ± 1.97	0.52 ± 0.24
NCB19010013	2014 - 2023	22.58 ± 5.87	31.42 ± 3.49	48137.92 ± 4965.08	7.82 ± 0.17	6.49 ± 1.14	1.01 ± 0.50
NCB19020005	2014 - 2023	22.54 ± 5.74	32.19 ± 2.58	49299.11 ± 3594.21	7.84 ± 0.15	6.37 ± 1.14	0.91 ± 0.38
NCB19020013	2014 - 2023	22.93 ± 5.73	31.21 ± 2.66	47943.11 ± 3691.93	7.69 ± 0.14	5.86 ± 1.35	0.78 ± 0.29

Table 3.1.2. Mean (±) standard deviation of chemical WQ parameters for stations within the LSMPU.

Station	Years	TP (mg/L)	TN (mg/L)	NOx-D (mg/L)	DOC (mg/L)	Color (PCU)	TSS (mg/L)	Chl-*a* (mg/m^3)
19010001	1998 - 2023	0.05 ± 0.01	1.06 ± 0.33	0.05 ± 0.03	31.00 ± 10.10	329.15 ± 181.72	10.06 ± 6.73	1.89 ± 1.80
NCB19010013	2014 - 2023	0.07 ± 0.03	0.43 ± 0.17	0.03 ± 0.03	6.43 ± 5.48	45.61 ± 70.63	16.97 ± 11.46	5.06 ± 2.47
NCB19020005	2014 - 2023	0.07 ± 0.03	0.42 ± 0.13	0.03 ± 0.02	4.46 ± 2.01	21.63 ± 21.30	20.87 ± 13.36	5.67 ± 2.50
NCB19020013	2014 - 2023	0.10 ± 0.04	0.60 ± 0.19	0.03 ± 0.03	5.88 ± 1.74	30.59 ± 18.99	35.08 ± 18.01	6.52 ± 2.70

Table 3.1.3. Summary statistics for physical and chemical parameters at WQ monitoring stations in the LSMPU.

	Temp. (°C)	Sal.	Cond. (µmhos/cm)	pH	DO (mg/L)	Secchi (m)	TP (mg/L)	TN (mg/L)	NOx-D (mg/L)	DOC (mg/L)	Color (PCU)	TSS (mg/L)	Chl-*a* (mg/m^3)
	Station 19010001 (western St. Marys River)												
KW (p)	**< 0.0001**	**0.0004**	**0.0004**	**0.0183**	**< 0.0001**	0.0633	**< 0.0001**	**0.0002**	0.3016	0.0948	0.0673	**0.0128**	0.0613
Kendall tau	0.101	-0.103	-0.081	-0.021	-0.126	-0.008	-0.064	-0.217	-0.181	-0.112	0.014	-0.244	-0.036
Kendall (p)	**0.0190**	**0.0217**	0.0705	0. 6789	**0.0044**	0.8584	0.1744	**< 0.0001**	0.0645	0.2607	0.8271	**< 0.0001**	0.5246
	Station NCB19010013 (Cumberland Sound)												
KW (p)	**< 0.0001**	0.0835	**0.0435**	0.1187	**< 0.0001**	0.9716	0.1624	**0.0067**	**0.0028**	0.1859	0.1337	**0.0209**	0.1225
Kendall tau	-0.015	-0.028	-0.015	-0.039	-0.162	0.030	-0.190	-0.103	-0.117	0.058	0.049	-0.160	-0.091
Kendall (p)	0.8219	0.6685	0.7235	0.5623	**0.0495**	0.6655	**0.0081**	0.3611	0.4548	0.4002	0.4763	0.0956	0.1733
	Station NCB19020005 (Amelia River north)												
KW (p)	**< 0.0001**	**0.0247**	**0.0088**	0.1331	**< 0.0001**	0.7595	0.3083	0.4437	**0.0015**	0.8918	0.8062	**0.0441**	0.1243
Kendall tau	0.025	0.034	0.039	0.026	-0.062	0.094	-0.153	-0.111	-0.104	0.035	-0.003	-0.056	-0.186
Kendall (p)	0.7985	0.8970	0.8984	0.6985	0.4242	0.1729	**0.0372**	0.1144	0.9558	0.6102	0.9655	0.6317	**0.0058**
	Station NCB19020013 (Amelia River south)												
KW (p)	**< 0.0001**	0.1135	0.0662	**0.0007**	**< 0.0001**	0.2197	**0.0161**	**0.0001**	**0.0003**	0.2530	0.2276	0.0787	**0.0016**
Kendall tau	-0.151	-0.029	-0.016	-0.009	-0.213	-0.051	-0.233	-0.242	-0.022	-0.063	-0.031	-0.191	-0.279
Kendall (p)	0.0529	0.6624	0.8096	0.8482	**0.0084**	0.4608	**0.0084**	**0.0035**	0.8549	0.3564	0.6407	**0.0049**	**0.0002**

If $p < 0.05$ for the Kruskal–Wallis (KW) test, a Seasonal Kendall test was performed. If $p \geq 0.05$ for the KW test, a Mann-Kendall test was performed. Significant test results ($p < 0.05$) are shown in bold.

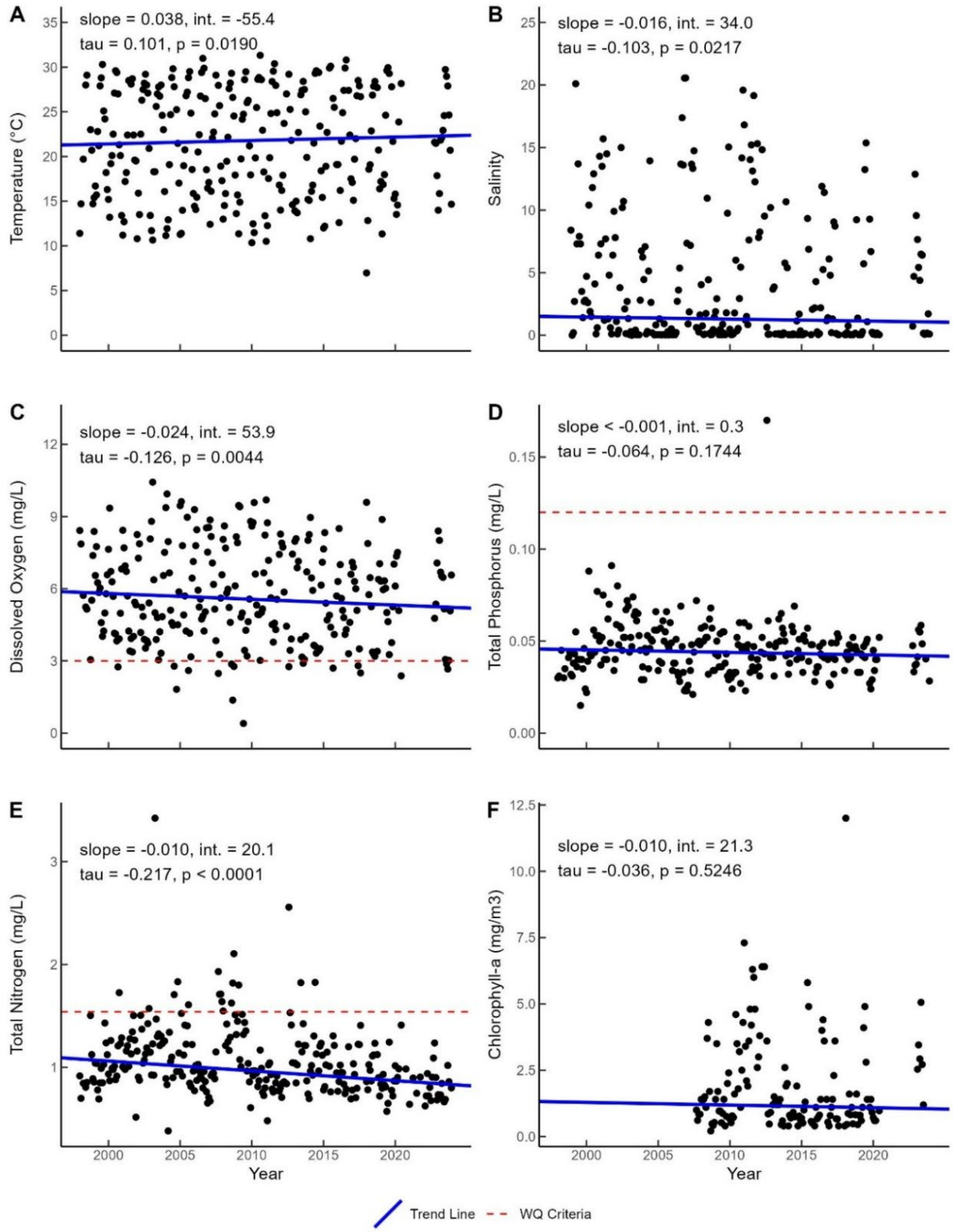

Figure 3.1.2 Trends for water temperature, salinity, DO, TP, TN and Chl-*a* (A-F) at the westernmost St. Marys River station (19010001). Seasonal Kendall = temperature, salinity, DO, TP, TN; Non-seasonal Kendall = Chl-*a*. The solid blue line depicts the monotonic trend for each parameter, with $p < 0.05$ denoting significant trends. The dashed red line indicates WQ criteria based on the 3.0 mg/L hypoxia stress threshold for DO and annual geometric mean values for TP and TN outlined in F.A.C. 62-302.531(2)(c). Streams do not have Chl-*a* criteria values.

At the **Cumberland Sound station (NCB19010013),** a decrease in TP and marginally significant decrease in DO were found (Table 3.1.3). Water temperature remained around 23.5°C, and salinity near 33 (euhaline) from 2014 to 2023 (Fig. 3.1.3A-B). The trendline for DO, and all individual concentrations were above 3.0 mg/L (Fig. 3.1.3C). Over the 9 years, TP decreased from 0.07 mg/L to 0.05 mg/L, with most individual concentrations below the numeric nutrient criteria value for this location (Fig. 3.1.3D). Additionally, all TN concentrations and most Chl-*a* concentrations remained below their respective numeric nutrient criteria values (Fig. 3.1.3E-F).

Decreases in TP and Chl-*a* were observed at the **northern Amelia River station (NCB19020005),** with the trend for water temperature near 23.6°C and salinity around 33 (euhaline) across the 9 years (2014-2023) (Table 3.1.3, Fig. 3.1.4A-B). Both the trendline and all individual concentrations for DO were above 3.0 mg/L (Fig. 3.1.4C). Concentrations for TP decreased from 0.08 mg/L in 2014 to 0.06 mg/L in 2023 with all concentrations below the numeric nutrient criteria value (Fig. 3.1.4D). While the trend for TN was not significant, most individual concentrations were below the nutrient criteria value (Fig. 3.1.4E). Across the 9-year period, Chl-*a* decreased from 6.0 mg/m^3 to 4.3 mg/m^3, with only two concentrations exceeding the nutrient criteria value (Fig. 3.1.4F).

Additionally, declining DO, TP, TN, TSS, and Chl-*a* concentrations were detected at the **southern Amelia River station (NCB19020013)** (Table 3.1.3). Water temperature remained around 24.2°C, and salinity near 32 (euhaline) from 2014 to 2023 (Fig. 3.1.5A-B). The trendline as well as all individual DO values recorded were above 3.0 mg/L (Fig. 3.1.5C). Over the 9 years, TP decreased from 0.11 mg/L to 0.08 mg/L and TN decreased from 0.65 mg/L to 0.49 mg/L (Fig. 3.1.5D-E). Most individual TP concentrations were below the nutrient criteria value whereas individual TN concentrations periodically exceeded the criteria value. Concentrations for Chl-*a* decreased from 7.1 mg/m^3 in 2014 to 4.7 mg/m^3 in 2023, with most individual concentrations below the nutrient criteria value (Fig. 3.1.5F).

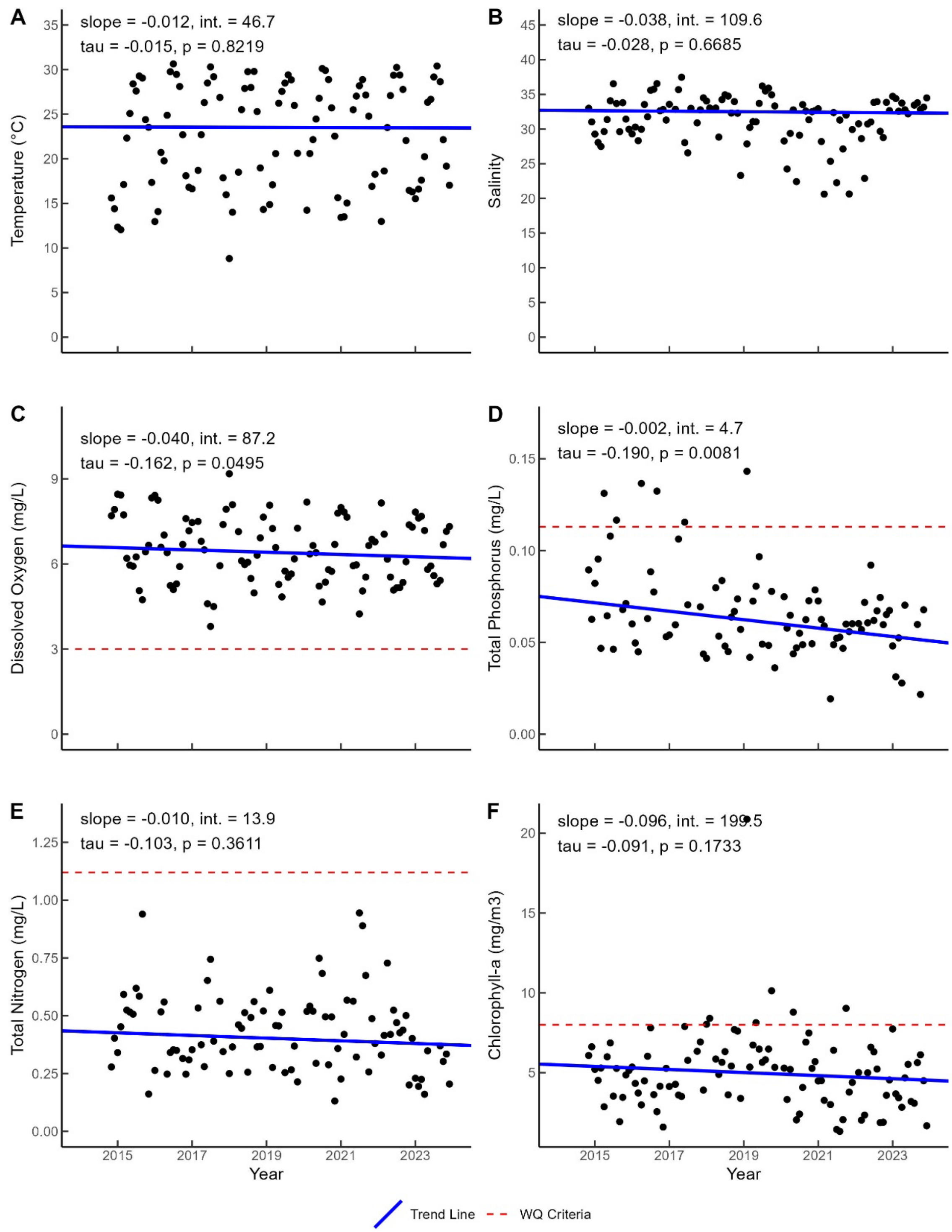

Figure 3.1.3. Trends for water temperature, salinity, DO, TP, TN, and Chl-*a* (A-F) at the Cumberland Sound station (NCB19010013). Seasonal Kendall = temperature, DO, TP, TN; Non-seasonal Kendall = salinity, Chl-*a*. The solid blue line depicts the monotonic trend for each parameter, with $p < 0.05$ denoting significant trends. The dashed red line indicates WQ criteria based on the 3.0 mg/L hypoxia stress threshold for DO and annual geometric mean values for TP, TN, and Chl-*a* outlined in F.A.C. Rule 62-302.532.

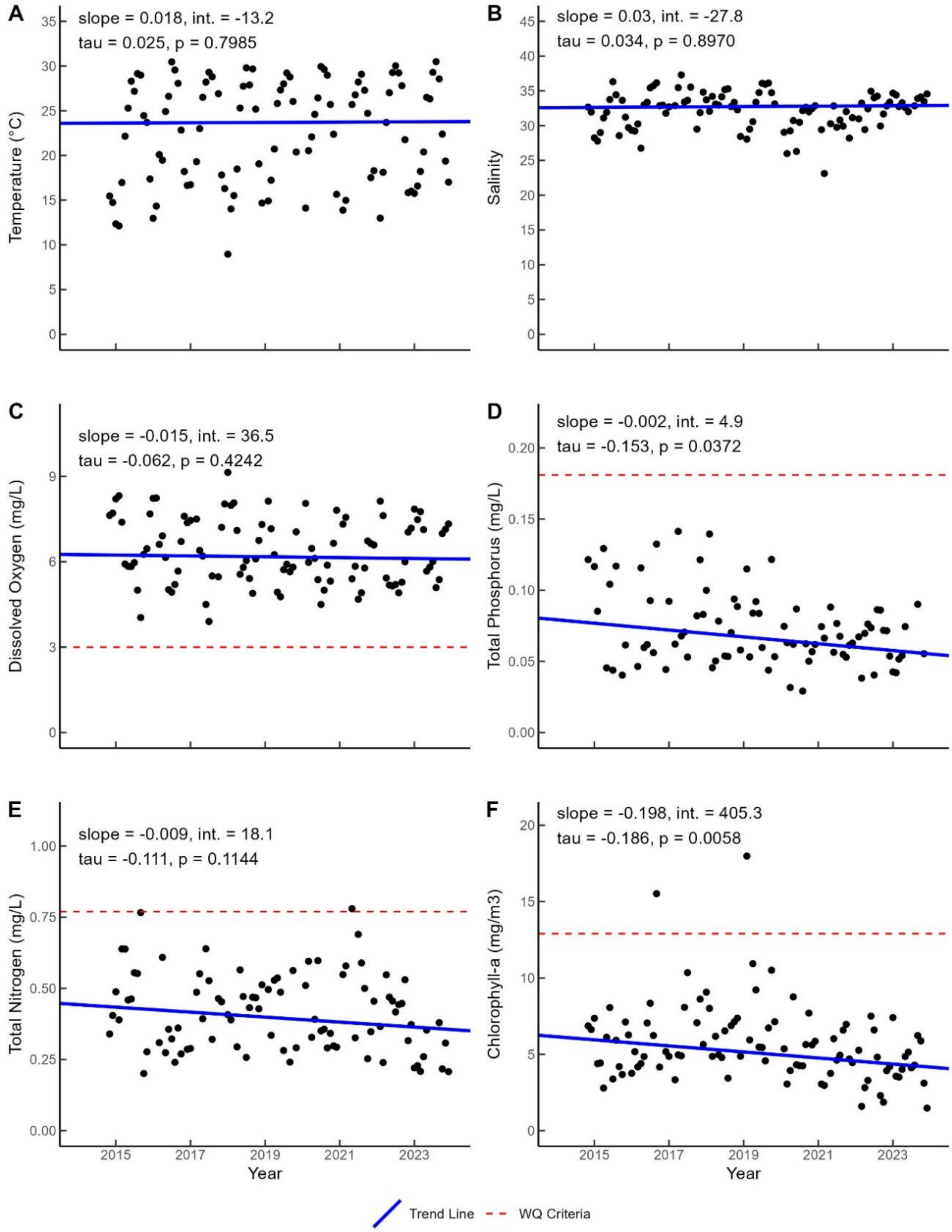

Figure 3.1.4. Trends for temperature, salinity, DO, TP, TN, and Chl-*a* (A-F) at the northern Amelia River station (NCB19020005). Seasonal Kendall = temperature, salinity, DO; Non-seasonal Kendall = TP, TN, Chl-*a*. The solid blue line depicts the monotonic trend for each parameter, with $p < 0.05$ denoting significant trends. The dashed red line indicates WQ criteria based on the 3.0 mg/L hypoxia stress threshold for DO and annual geometric mean values for TP, TN, and Chl-*a* outlined in F.A.C. Rule 62-302.532.

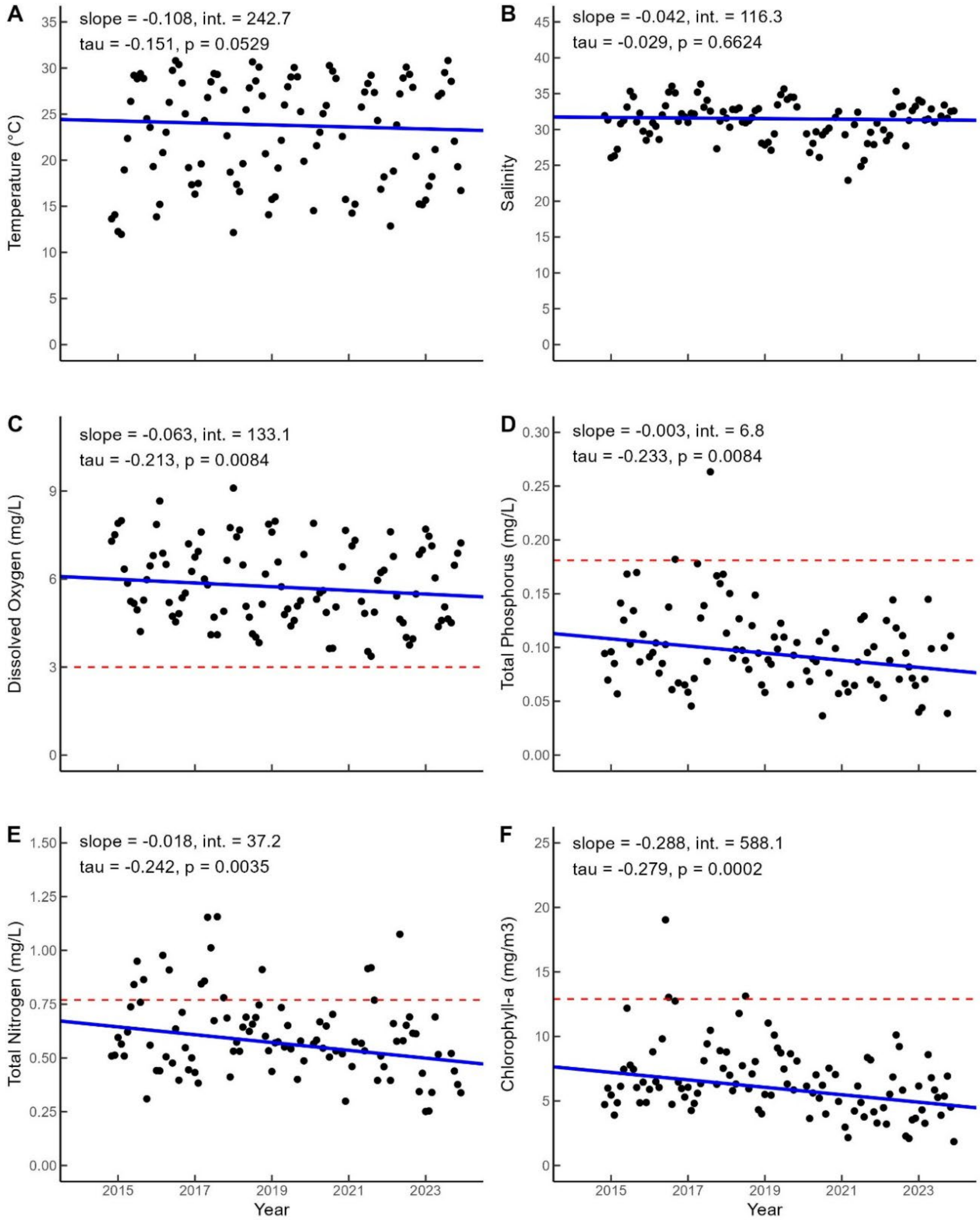

Figure 3.1.5. Trends for temperature, salinity, DO, TP, TN, and Chl-*a* (A-F) at the southern Amelia River station (NCB19020013). Seasonal Kendall = temperature, DO, TP, TN, Chl-*a*; Non-seasonal Kendall = salinity. The solid blue line depicts the monotonic trend for each parameter, with $p < 0.05$ denoting significant trends. The dashed red line indicates WQ criteria based on the 3.0 mg/L hypoxia stress threshold for DO and annual geometric mean values for TP, TN, and Chl-*a* outlined in F.A.C. Rule 62-302.532.

Discussion

Within the LSMPU, mean WQ parameters were similar among the three eastern stations located near Cumberland Sound, Amelia River north and Amelia River south. Mean salinity at these stations indicated marine (euhaline) conditions and pH were consistent with those typically reported for estuarine systems (Ohrel and Register 2006). Mean salinity at the westernmost St. Marys River station was oligohaline, indicating limited marine influence, consistent with previous studies (Blair et al. 2009, Fox et al. 2018). Low mean pH and high mean DOC and color align with blackwater system characteristics (Gallegos 2005, Blair et al. 2009, Flotemersch et al. 2024) and conditions observed in Florida's low salinity blackwater estuaries (Chen et al. 2015) including Nassau (Williams and Kimball, 2013) and GTM (Chaya et al. 2023). Parameters including conductivity, color, and TSS showed high variability, as expected, due to their sensitivity to rainfall and tidal mixing (FLW 2004, Phlips et al. 2004).

Trends in WQ parameters were observed at all four monitoring stations, with changes being weak (tau < 0.3) for all stations and parameters. In some cases, such as water temperature, these changes were expected to be gradual over time. Rising air and water temperatures have been reported across 346 global estuaries, with an average water warming rate of 0.070 ± 0.004°C per year, which has been correlated with warming air temperatures (Prum et al. 2024). The observed increase in water temperature in the upper estuary aligns with trends in the eastern United States (Oczkowski et al. 2015, Rice and Jastram 2015, Mallick and Dunn 2024) and South Florida (Shi and Hu 2022, Shi et al. 2024) estuaries and estuarine rivers.

Water temperature in estuarine systems regulates physical, chemical and biological processes including ecosystem metabolism, water-atmospheric exchange of gases (e.g., carbon dioxide, DO) and estuarine community dynamics (Mallick and Dunn 2024, Prum et al. 2024). Despite its importance in regulating various processes within estuaries, relatively little is known about how estuarine ecosystems will respond to long-term increases in average water temperature (Itsukushima et al 2024, Prum et al. 2024). Understanding the potential effects of warming water temperatures within Northeast Florida estuaries is important given the proximity of these systems to the tropical-temperate transition zone, which spans from Indian River Lagoon in the south to the St. Augustine inlet in the north (approximately 28° N–30° N; Kimball and Eash-Loucks 2021, Osland et al. 2021, Brewton and Lapointe 2023). Within tropical-temperate transition zones, winter air and water temperatures have a crucial ecological role and can serve as a barrier to the northward expansion of cold sensitive subtropical species (Boucek and Rehage 2014, Purtlebaugh et al. 2020, Mallick and Dunn 2024). Milder winters within these transition zones have led to the poleward expansion of subtropical species, such as western Atlantic sea bream (*Archosargus rhomboidalis*) in the Indian River Lagoon (Adams et al. 2024) and common snook (*Centropomus undecimalis*) in Cedar Key (29° N; Purtlebaugh et al. 2020) which has raised concerns over changes to biodiversity and species interactions (e.g., trophic webs). Warming water temperature is also anticipated to affect WQ through alteration to biogeochemical processes (e.g., carbon and nitrogen cycling), increased microbial activity and decomposition rates, and enhanced risk of hypoxia (Statham 2012, Nydahl et al. 2013, Wetz and Yoskowitz 2013, Prum et al. 2024). Considering current knowledge gaps, concerns over habitat functioning, and evidence of warming water temperatures in at least one region of the St. Marys estuary, further research on the mechanisms driving this change and monitoring ecosystem responses is crucial. This will provide important information for developing management plans to maintain healthy coastal habitats and good WQ throughout the NCB.

Estuarine salinity varies annually, influenced by freshwater inflow, evapotranspiration, precipitation amounts and corresponding meteorological phenomena (e.g., El Niño-Southern Oscillation), with increased freshwater inflow and precipitation typically lowering salinities (Schmidt and Luther 2002, Sumner and Belaineh 2005, Feher et al. 2023). Periods of high salinity (e.g., 16.8-20.6) in the upper estuary coincided with extreme droughts in 1998, 2006, and 2011. Increases in rainfall and freshwater inflow into the system since the last severe drought likely explain the decrease in salinity within this region. A 0.5 decrease in salinity over time in the upper estuary is not expected to have a biologically significant effect on estuarine aquatic plant and animal communities. Rather, abrupt or prolonged changes to salinity resulting from large-scale storms (e.g., hurricanes) or droughts is more concerning due to their negative impact on biological communities. For example, low salinity (< 5) from high freshwater inflow can hinder oyster recruitment, survival and growth, especially in warm water conditions (> 30 °C; La Peyre et al. 2013, Rybovich et al. 2016). Conversely, prolonged high salinity (e.g., 33) during drought can increase disease-related mortality, with infection rates being exacerbated by warm water conditions (Petes et al. 2012).

Decreasing DO was found in both the upper and lower regions of the estuary. This negative trend in the upper estuary can be explained by increasing water temperature and decreasing salinity. Warmer temperatures lower oxygen solubility and increase metabolic rates (Chapra et al. 2021, Prum et al. 2024) and freshwater in blackwater estuaries often have low DO due to high decaying vegetation (Blair et al. 2009, Flotemersch et al. 2024). Decaying vegetation, which is primarily broken down by bacteria, can directly lower DO within the water column and increase sediment oxygen demand (Todd et al. 2009). During the summer, warm water temperatures in blackwater systems can drive hypoxic events as the metabolic activity of bacteria accelerates, consuming DO faster than it can be replenished (Kerr et al. 2013). In the upper estuary, DO concentrations periodically dropped below 3.0 mg/L during the summer and early fall (June to October). Continued DO monitoring is important as concentrations are decreasing, and the river segment east of US HW 17 (overlapping WBID 2097N) is currently impaired for DO. The Georgia Department of Natural Resources Environmental Protection Division (GADNR EPD) attributes this impairment to natural conditions and both point and non-point pollution sources (GADNR EPD 2001, GADNR EPD 2002, GADNR EPD 2017). Point sources of pollution include effluent from two water treatment facilities, regulated and industrial stormwater discharge and concentrated animal feeding operations, and non-point sources include stormwater and land application system runoff (GADNR EPD 2001, GADNR EPD 2017).

Declining DO in the lower estuary may be related to other mechanisms (e.g., biological oxygen demand, chl-*a*) since no change in water temperature or salinity was observed. The northern Amelia River (WBID 2124A), from the St. Marys Inlet to State Road 200, is currently impaired for DO, with nutrients identified as the causative pollutant in FDEP's 2022-2024 WQ Assessment, TMDLs, and BMAPs. This region of the Amelia River has Site Specific Alternative Criteria which requires that DO concentrations from July 1 to September 30 be ≥ 3.2 mg/L at low tide, ≥ 4.0 mg/L during all other conditions, and have mean DO concentrations ≥ 5.0 mg/L per 24 hrs. While all individual DO concentrations at both the northern and southern Amelia River stations were above the 3.0 mg/L hypoxia stress threshold, current conditions do not meet the Site Specific Alternative Criteria for DO. From a management perspective, low DO concentrations are a concern for fisheries as chronic hypoxic conditions can lead to fish kills and mortality of oyster reefs (Landsberg et al. 2020, Coxe et al. 2023). Additionally, low DO concentrations can also impact biogeochemical processes influencing nitrogen cycling and the flux of phosphorus, manganese, iron, cobalt and DOC between sediments and water (Kemp and Dodds 2002, Skoog and Arias-Esquivel 2009). However, more information is needed to fully understand how surface DO concentrations relate to benthic DO and biogeochemical processes.

In addition to DO, GADNR EPD has developed TMDLs for fecal coliforms and bacteria across several sections of the St. Marys River in Georgia (GADNR EPD 2006, GADNR EPD 2022). Although fecal coliforms and bacteria were not assessed in this report, management efforts to reduce these pollutants within the St. Marys River may explain observed reductions in nutrients (TP/TN), TSS, and/or Chl-*a* throughout the estuary. Point and non-point pollution sources, including wastewater treatment facilities, stormwater discharge, animal feeding operations, agricultural runoff, and failing septic systems and sewer lines (GADNR EPD 2002, GADNR EPD 2006, GADNR EPD 2022) contribute to fecal coliforms and introduce suspended particles and excess nutrients, potentially leading to nutrient impairments and elevated chlorophyll concentrations. Similar impacts have been documented in Florida's Halifax and St. Lucie Rivers (Magley 2013, Lapointe et al. 2017). In addition to the management efforts of GADNR EPD and St. Marys River Management Committee, SJRWMD has partnered with the city of Fernandina Beach through its Cost-Share Program to improve stormwater management. The goal of the 2017 Flood Protection, Nutrient Reduction Project was to reduce flooding along 26 city blocks and provide stormwater treatment to reduce nutrient loading into the Amelia River. Additionally, the Fernandina Beach Area 1 Drainage Improvements Project (Fiscal Year 2024-2025) was recently funded (in part by SJRWMD's Cost-Share Program) with the goal of further reducing sediment and nutrients that drain into the Amelia River from stormwater effluent.

Despite decreasing nutrients and Chl-*a* in the lower estuary, the northern segment of the Amelia River (WBID 2124A) currently has TP and Chl-*a* impairments based on FDEP's

2022-2024 WQ Assessment, TMDLs, and BMAPs. Most TP and Chl-*a* concentrations were below nutrient criteria, but impairment is determined by the 7-year annual geometric mean across multiple stations within a WBID, not individual stations. This example highlights the importance of WQ monitoring, having an appropriate network to detect impairments, and use of comprehensive analyses which assess WQ trends across multiple stations at broad scales (e.g., WBIDS, regions, etc.) to evaluate overall ecosystem health, since individual station trends can differ from larger scale analyses. However, individual station analyses are still useful for understanding spatial differences in WQ parameters across a system and aiding ecosystem managers in determining locations that may be at risk of developing an impairment or should be prioritized for specific management goals.

Overall, analysis of long-term WQ trends in the LSMPU indicates that water temperature is increasing, and salinity is decreasing in the upper region of the estuary and that DO, TP, TN and Chl-*a* are decreasing across multiple stations throughout the estuary. Changes to temperature and DO concentrations may be a concern for biological communities as well as ecosystem function. Warming water temperatures may exacerbate naturally low DO concentrations during the summer months in the upper region of the estuary and milder winter water temperatures may allow for species range expansion or life history alteration, which can modify ecosystem dynamics. Continued monitoring of DO is especially important at the westernmost and southern Amelia River stations since DO is declining at both stations and they are located near or within regions of the estuary that are impaired for DO. Prolonged periods of low DO concentrations are a concern for fisheries since lower DO can increase fish and oyster stress and mortality (Landsberg, et al. 2020, Coxe et al. 2023). Although nutrient and Chl-*a* concentrations are declining, continued management aimed at reducing nutrient loading, such as the Fernandina Beach Area 1 Drainage Improvements Project, are needed to address current DO, TP, and Chl-*a* impairments in the northern segment of the Amelia River and prevent future impairments in other regions of the LSMPU from developing.

3.2. NASSAU RIVER PLANNING UNIT

Summary

- There are six SJRWMD WQ monitoring stations located within the Nassau River Planning Unit (NRPU) (1A) and watershed.

- Mean conditions in the upper estuary were oligohaline west of I-95 and mesohaline near US HW 17. Both locations had characteristics (low pH, high DOC and high color) typical of blackwater estuarine systems. The four stations in the lower estuary had similar conditions, with mean salinity being either polyhaline or euhaline.

- Trends for one or more WQ parameters were found at each station, with the strength of all changes being weak (tau < 0.3). In the upper estuary trends include increasing water temperature, pH, and Secchi and decreasing DO, TP, TN, NOx-D, DOC, and TSS. In the lower estuary trends include decreasing DO, Secchi, TP, TN, and Chl-*a* at one or more stations.

- Increasing water temperature can be explained by warmer weather conditions and decreasing DO by increasing water temperature and natural conditions. Continued monitoring is important since warmer conditions can affect physical, chemical and biological processes and exacerbate naturally low DO conditions in the upper estuary.

- More information is needed to determine the mechanisms driving nutrient declines. Despite decreasing nutrients and Chl-*a*, many individual concentrations exceeded their respective nutrient criteria values and further reductions in nutrient inputs could be beneficial for maintaining good WQ.

Introduction

The Nassau River is an 88 km (55 mi) long blackwater river system that forms a coastal plain estuary which drains into the Atlantic Ocean (Williams and Kimball, 2013). The headwaters of this system originate in Nassau County, and the watershed (1,100 km^2; 425 mi^2; Bacopoulos et al. 2012, Williams and Kimball 2013) is relatively small compared to the adjacent watersheds of the St. Marys (3,367 km^2; 1,300 mi^2; GADNR EPD 2002) and St. Johns Rivers (22,000 km^2; 8,494 mi^2; Bacopoulos et al. 2012). Major freshwater tributes into Nassau River headwaters include Thomas, Alligator, Mills, Little Boggy and Swamp Creeks (Coffin et al. 1992, Boning 2016). The basin can be divided into three zones; a freshwater upstream region, brackish middle section, and predominately saltwater area downstream (Coffin et al. 1992), with spring tides up to 2 m (6.5 ft; Bacopoulos et al. 2012, Williams and Kimball, 2013). The Nassau River estuary is the largest marsh-estuarine system found along the east coast of Florida (Dix et al. 2021), with expansive cordgrass (*Spartina alterniflora*) marshes that provides habitat for animals such as roseate spoonbill (*Platalea ajaja*), diamondback terrapins (*Malaclemys terrapin*) and bay anchovies (*Anchoa mitchilli*).

The Nassau River Planning Unit (NRPU) (1A) spans southern Nassau County and into the northern portion of Duval County, with the Nassau River forming the boundary between the two. The NRPU is bound in the north by the LSMPU and in the south by the ICWPU, with its eastern boundaries extending from the southern portion of Amelia Island through the Talbot Islands (Fig. 3.2.1). Similar to the Lower St. Marys, the western boundary for this PU is restricted to I-95 but the Nassau River watershed and brackish region extends past this boundary. There are several designated OFWs within the NRPU, including the Timucuan Ecological and Historic Preserve, Fort Caroline National Memorial, Nassau Valley State Reserve, Ft. George Island, and Nassau River-St. Johns River Marshes Aquatic Preserve.

The Nassau River has been partly buffered from development by nearby federal and state conservation areas including the Timucuan Ecological and Historic Preserve, Nassau River – St. Johns River Marshes Aquatic Preserve, Pumpkin Hill Creek Preserve State Park, and Amelia, Big Talbot, Little Talbot and Ft. George Island State Parks. Waterways throughout the NRPU are a mixture of Class II and Class III waters. Extensive oyster reefs can be found throughout the Amelia, Nassau, and Ft. George rivers and their tidal tributaries (Dix et al. 2019). Parts of the South Amelia River, Nassau River and Sound, ICW and Pumpkin Hill Creek are designated as Class II waters, with shellfish harvesting in all waters east of Black Hammock Island, and west of Little Talbot Island and Ft. George Island (Fig. 3.2.2). However, starting in the 1980s shellfish harvest within these areas has been periodically prohibited due to poor WQ (Fig. 3.2.2).

Methods

For a full description of WQ parameters and analyses performed refer to Chapter 2. At the Nassau River station west of I-95 (NRI), sampling was bimonthly from 1995-2014 and monthly from 2015-2023. Since most data were collected bimonthly, they were aggregated into bimonthly bins. While this station is outside of the NRPU boundary and is considered a stream based on F.A.C. regulations, it was included in the analysis since it overlaps with the brackish region of the Nassau River.

Additionally, the Nassau River station near US HW 17 (19020002) had bimonthly data from 1995-1997 and monthly data from 1998-2023. Since most data was collected monthly, data from 1995-1997 was removed. Additionally, Chl-*a* at this station had a large data gap between 1999-2006.

Figure 3.2.1. Map of the NRPU (1A) (blue). Red circles indicate SJRWMD WQ monitoring stations within the PU and contributing watershed.

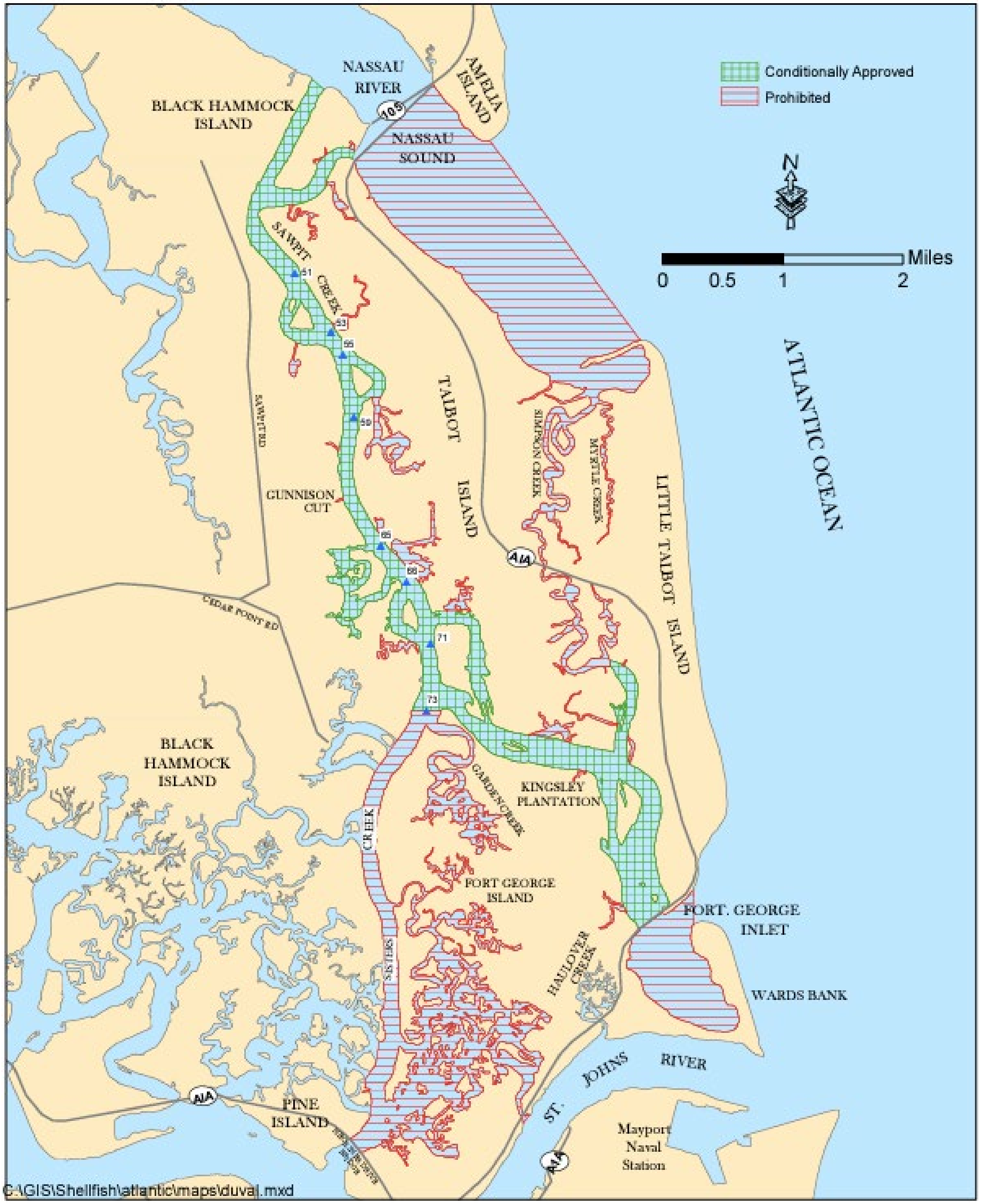

Figure 3.2.2. Map of open and prohibited shellfish harvesting areas within shellfish harvesting area 96 in Duval County as of March 30, 2020. Blue triangles on the map represent channel markers. Map provided by FDACS.

Results

Mean Water Quality Conditions

There are six SJRWMD monitoring stations located within the NRPU and watershed (Fig. 3.1.1). The two western stations located to the left of I-95 (NRI) and near US HW 17 (19020002) are the longest running stations with sampling beginning in 1995 and 1998, respectively. Sampling began near the end of 2014 for the other four stations (NCBGD, NCBCM24, NCBNAAM, NCB19020038). Comparison of mean physical WQ parameters showed similarities in water temperature and DO across all stations (Table 3.2.1). However, the two western stations had lower salinity, conductivity, pH, and Secchi depths compared to the other four stations, likely due to their upstream location. Mean chemical WQ parameters were also similar between all stations except for the two western stations, which had higher TN, DOC, and color (Table 3.2.2).

Monthly Variability

Differences across months were found for all WQ parameters at the Nassau River station west of I-95 and for water temperature, pH, DO, TP, TSS, and Chl-*a* at all other stations (Table 3.2.3). Results from the KW tests were also significant ($p < 0.05$) for TN at the Nassau River station east of US HW 17, northern Amelia River and Sisters Creek. Additionally, differences across months were found for salinity and conductivity at the Nassau River station near HW 17 and for Secchi and NOx-D at the northern Amelia River station (Table 3.2.3).

Station Water Quality Trends and Criteria

Investigation of WQ trends showed increases in water temperature and pH and decreases in TP, TN, DOC, and TSS at the **Nassau River west station (NRI)** (Table 3.2.3). Water temperature increased by 1°C (22.0°C to 23.0°C) from 1995 to 2023 (Fig. 3.2.3A). Salinity remained near 0.5 from 1999 to 2023, with higher salinities (mesohaline) primarily recorded during drought years (e.g., 2006, 2011) (Fig. 3.2.3B). While the DO trendline was above the 3.0 mg/L hypoxia threshold from 1995 to 2023, individual concentration periodically fell below this threshold (Fig. 3.2.3C). Across the 28-year (1995-2023) period of record, TP concentrations decreased from 0.15 mg/L to 0.09 mg/L and TN from 1.3 mg/L to 1.1 mg/L (Fig. 3.2.3D-E). Individual values for TP and TN periodically exceeded their respective numeric nutrient criteria values, with the trendline exceeding the nutrient threshold for TP until approximately 2010. While there is no nutrient criteria threshold for Chl-*a* at this location, concentrations were highly variable across years (range: 0.9 – 50.5 mg/m^3; Fig. 3.2.3F).

Table 3.2.1. Mean (±) standard deviation of physical WQ parameters for stations within the NRPU.

Station	Years	Temp. (°C)	Sal.	Cond. (µmhos/cm)	pH	DO (mg/L)	Secchi (m)
NRI	1995 - 2023	21.94 ± 6.10	1.76 ± 2.93	2872.91 ± 4750.55	6.51 ± 0.33	5.21 ± 2.04	0.48 ± 0.15
19020002	1998 - 2023	21.83 ± 6.00	12.01 ± 9.30	18976.33 ± 14063.38	6.92 ± 0.35	6.11 ± 1.86	0.50 ± 0.21
NCBGD	2014 - 2023	23.01 ± 5.88	25.53 ± 7.07	39867.83 ± 10204.12	7.43 ± 0.23	5.79 ± 1.40	0.70 ± 0.24
NCBCM24	2014 - 2023	22.93 ± 5.97	30.63 ± 2.85	47145.35 ± 3951.01	7.61 ± 0.11	5.71 ± 1.34	0.74 ± 0.2)
NCBNAAM	2014 - 2023	22.83 ± 5.84	31.68 ± 2.94	48240.31 ± 5197.17	7.81 ± 0.14	6.45 ± 1.18	0.83 ± 0.31
NCB19020038	2014 - 2023	23.50 ± 5.64	31.50 ± 3.41	48203.65 ± 4799.02	7.85 ± 0.13	6.59 ± 1.14	0.95 ± 0.33

Table 3.2.2. Mean (±) standard deviation of chemical WQ parameters for stations within the NRPU.

Station	Years	TP (mg/L)	TN (mg/L)	NOx-D (mg/L)	DOC (mg/L)	Color (PCU)	TSS (mg/L)	Chl-*a* (mg/m^3)
NRI	1995 - 2023	0.13 ± 0.06	1.26 ± 0.45	0.05 ± 0.04	33.08 ± 8.11	291.57 ± 163.88	11.56 ± 8.71	8.03 ± 11.17
19020002	1998 - 2023	0.15 ± 0.11	1.15 ± 0.35	0.05 ± 0.04	22.49 ± 9.65	191.96 ± 141.27	26.86 ± 22.87	9.06 ± 11.34
NCBGD	2014 - 2023	0.10 ± 0.04	0.71 ± 0.48	0.05 ± 0.04	10.41 ± 5.89	72.67 ± 67.46	26.21 ± 13.81	7.95 ± 4.71
NCBCM24	2014 - 2023	0.08 ± 0.03	0.55 ± 0.14	0.04 ± 0.03	6.17 ± 1.84	31.66 ± 18.74	23.76 ± 12.86	6.03 ± 3.00
NCBNAAM	2014 - 2023	0.08 ± 0.03	0.44 ± 0.15	0.03 ± 0.03	4.72 ± 2.10	22.40 ± 19.67	26.72 ± 13.67	8.22 ± 3.37
NCB19020038	2014 - 2023	0.08 ± 0.03	0.49 ± 0.17	0.05 ± 0.05	5.79 ± 2.50	28.83 ± 22.72	18.67 ± 12.43	7.21 ± 3.69

Table 3.2.3. Summary statistics for physical and chemical parameters at WQ monitoring stations in the NRPU.

	Temp. (°C)	**Sal.**	**Cond. (µmhos/cm)**	**pH**	**DO (mg/L)**	**Secchi (m)**	**TP (mg/L)**	**TN (mg/L)**	**NOx-D (mg/L)**	**DOC (mg/L)**	**Color (PCU)**	**TSS (mg/L)**	**Chl-*a* (mg/m³)**
	Station NRI (Nassau River west)												
KW (p)	**< 0.0001**	**0. 0073**	**0.0082**	**< 0.0001**	**< 0.0001**	**0.0181**	**0.0071**	**< 0.0001**	**0.0213**	**0. 0295**	**< 0.0001**	**< 0.0001**	**0.0001**
Kendall tau	0.131	-0.038	0.087	0.124	-0.022	0.047	-0.262	-0.191	-0.043	-0. 210	-0.057	-0.149	0.037
Kendall (p)	**0.0192**	0.5083	0.1147	**0.0223**	0.6843	0.4648	**< 0.0001**	**0.0009**	0.5873	**0.0250**	0.3431	**0.0143**	0.4334
	Station 19020002 (Nassau River)												
KW (p)	**< 0.0001**	**0.0156**	**0.0043**	**< 0.0001**	**< 0.0001**	0.7268	**0.0020**	0.0656	0.3105	0.9099	0.9510	**0.0044**	**< 0.0001**
Kendall tau	0.139	0.015	0.041	0.067	-0.104	0.130	-0.035	-0.204	-0.287	-0.284	-0.062	-0.238	-
Kendall (p)	**0.0016**	0.7908	0.3484	0.1542	**0.0132**	**0.0144**	0.4291	**< 0.0001**	**0.0046**	**0.0041**	0.3261	**< 0.0001**	-
	Station NCBGD (Nassau River east)												
KW (p)	**< 0.0001**	0.7317	0.7755	**< 0.0001**	**< 0.0001**	0.3963	**0.0079**	**0.0049**	0.1793	0.4359	0.2357	**0.0123**	**< 0.0001**
Kendall tau	-0.034	-0.008	-0.003	-0.005	-0.040	-0.027	0.010	-0.098	-0.094	-0.019	-0.066	0.105	-0.177
Kendall (p)	0.6896	0.9031	0.9617	0.8935	0.7166	0.6982	0.9706	0.2499	0.2582	0.7832	0.3446	0.2038	**0.0209**
	Station NCBCM24 (Amelia River north)												
KW (p)	**< 0.0001**	0.1186	0.0628	**< 0.0001**	**< 0.0001**	**0.0063**	**0.0001**	**0.0002**	**0.0204**	0.3686	0.1790	**0.0370**	**< 0.0001**
Kendall tau	-0.056	0.018	0.020	0.048	0.029	-0.120	-0.143	-0.075	0.037	-0.035	-0.074	0.086	-0.218
Kendall (p)	0.4829	0.7911	0.7611	0.5459	0.6349	0.1528	0.0813	0.3841	0.5891	0.6080	0.2809	0.2545	**0.0044**
	Station NCBNAAM (Amelia River south)												
KW (p)	**< 0.0001**	0.6122	0.5310	**0.0113**	**< 0.0001**	0.0567	**0.0042**	0.0974	0.2941	0.6326	0.4398	**0.0103**	**0.0002**
Kendall tau	0.027	0.003	0.039	-0.074	-0.099	0.065	-0.244	-0.109	0.113	0.061	0.015	-0.147	-0.236
Kendall (p)	0.8482	0.9675	0.5546	0.3562	0.2542	0.3497	**0.0076**	0.1116	0.2415	0.3702	0.8313	0.1081	**0.0029**
	Station NCB19020038 (Sisters Creek)												
KW (p)	**< 0.0001**	0.4450	0.4257	**0.0157**	**< 0.0001**	0.0727	**0.0230**	**0.0206**	0.4771	0.1555	0.2758	**0.0123**	**< 0.0001**
Kendall tau	-0.054	-0.113	-0.128	-0.093	-0.188	-0.221	-0.070	-0.233	-0.050	0.047	0.016	0.105	-0.165
Kendall (p)	0.5028	0.0900	0.0531	0.2017	**0.0195**	**0.0011**	0.4639	**0.0050**	0.5317	0.4867	0.8109	0.2038	**0.0433**

If $p < 0.05$ for the Kruskal–Wallis (KW) test, a Seasonal Kendall test was performed. If $p \geq 0.05$ for the KW test, a Mann-Kendall test was performed. Significant test results ($p < 0.05$) are shown in bold.

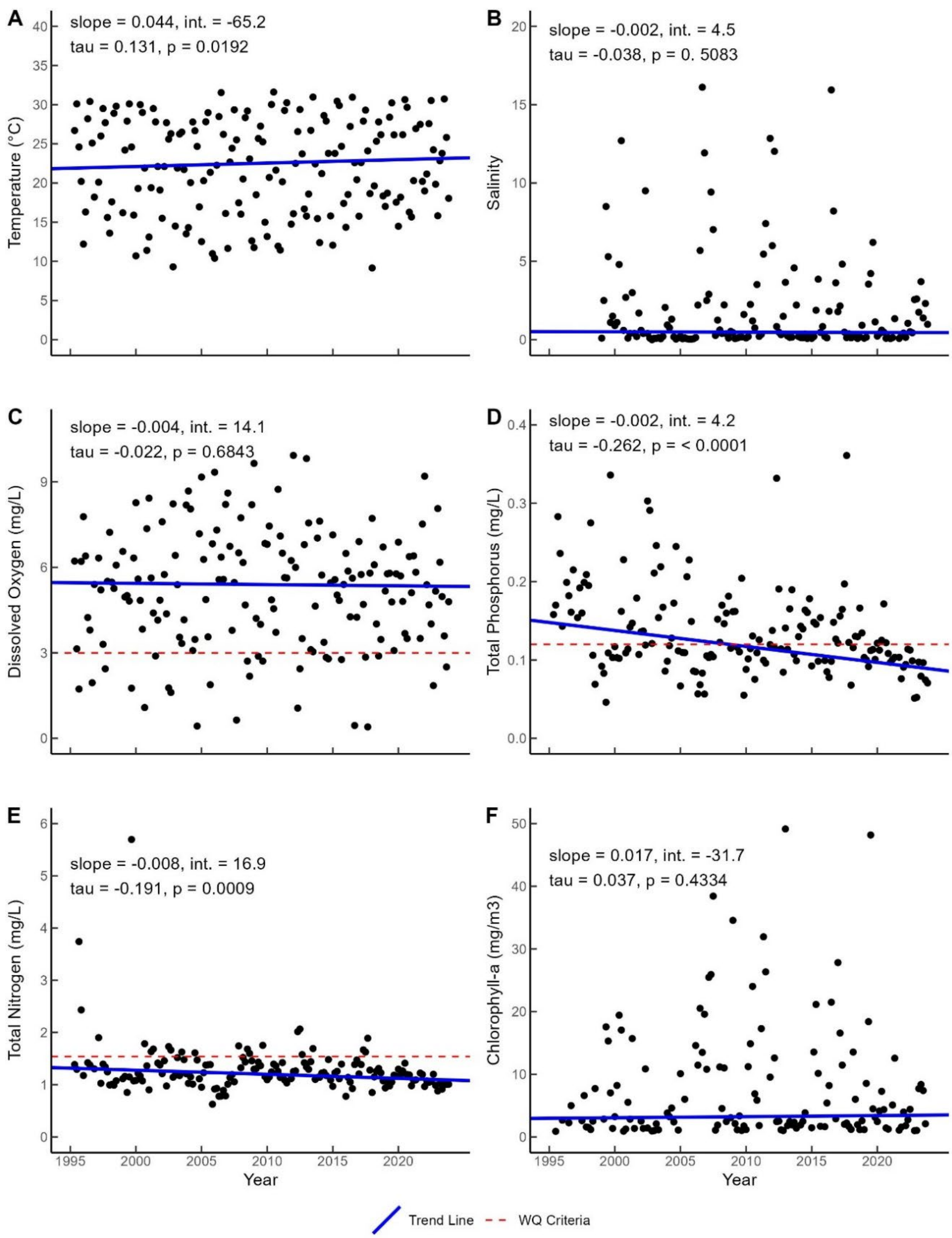

Figure 3.2.3. Seasonal Kendall trends for temperature, salinity, DO, TP, TN, and Chl-*a* (A-F) at the Nassau River station west (NRI). The solid blue line depicts the monotonic trend for each parameter, with $p < 0.05$ denoting significant trends. The dashed red line indicates WQ criteria based on the 3.0 mg/L hypoxia stress threshold for DO and annual geometric mean values for TP and TN outlined in F.A.C. 62-302.531(2)(c). Streams do not have Chl-*a* criteria values.

For the **Nassau River station (19020002)** increases in water temperature and Secchi depth and decreases in DO, TN, NOx-D, DOC, and TSS were found (Table 3.2.3). Across the 25-year period of record (1998 to 2023), water temperature increased by 1.1°C (21.9°C to 23.0°C), while salinity remained near 11.5 (Fig. 3.2.4A-B). For DO, concentrations decreased from 6.2 mg/L in 1998 to 5.7 mg/L in 2023, with both the trendline and most individual concentrations being above 3.0 mg/L (Fig. 3.2.4C). While no change in TP was detected, many individual concentrations exceeded the nutrient criteria value (Fig. 3.2.4D). From 1998 to 2023, TN decreased from 1.3 mg/L to 0.9 mg/L, with many individual data points also exceeding the nutrient criteria value (Fig. 3.2.4E). Significant heterogeneity was found for Chl-*a* (Chi-Square Heterogeneity Test, p = 0.0416). Results from individual Mann-Kendall tests performed on each month showed that Chl-*a* had a negative trend for all months except May, September and November where trends were positive, with January and March being the only two months with significant trends (Table 3.2.4). Many of the individual Chl-*a* concentrations and trendlines for individual months were above the nutrient criteria value (Fig. 3.2.4F).

Table 3.2.4. Results for individual monthly Mann-Kendall tests run for Chl-*a* (mg/m^3) at the Nassau River station (19020002).

Month	tau	p-value
January	-0.538	**0.0086**
February	-0.359	0.0995
March	-0.462	**0.0321**
April	-0.372	0.0870
May	0.289	0.2659
June	-0.231	0.2736
July	-0.218	0.3281
August	-0.236	0.3502
September	0.289	0.2831
October	-0.267	0.1810
November	0.256	0.2464
December	-0.319	0.1242

Significant test results ($p < 0.05$) are shown in bold.

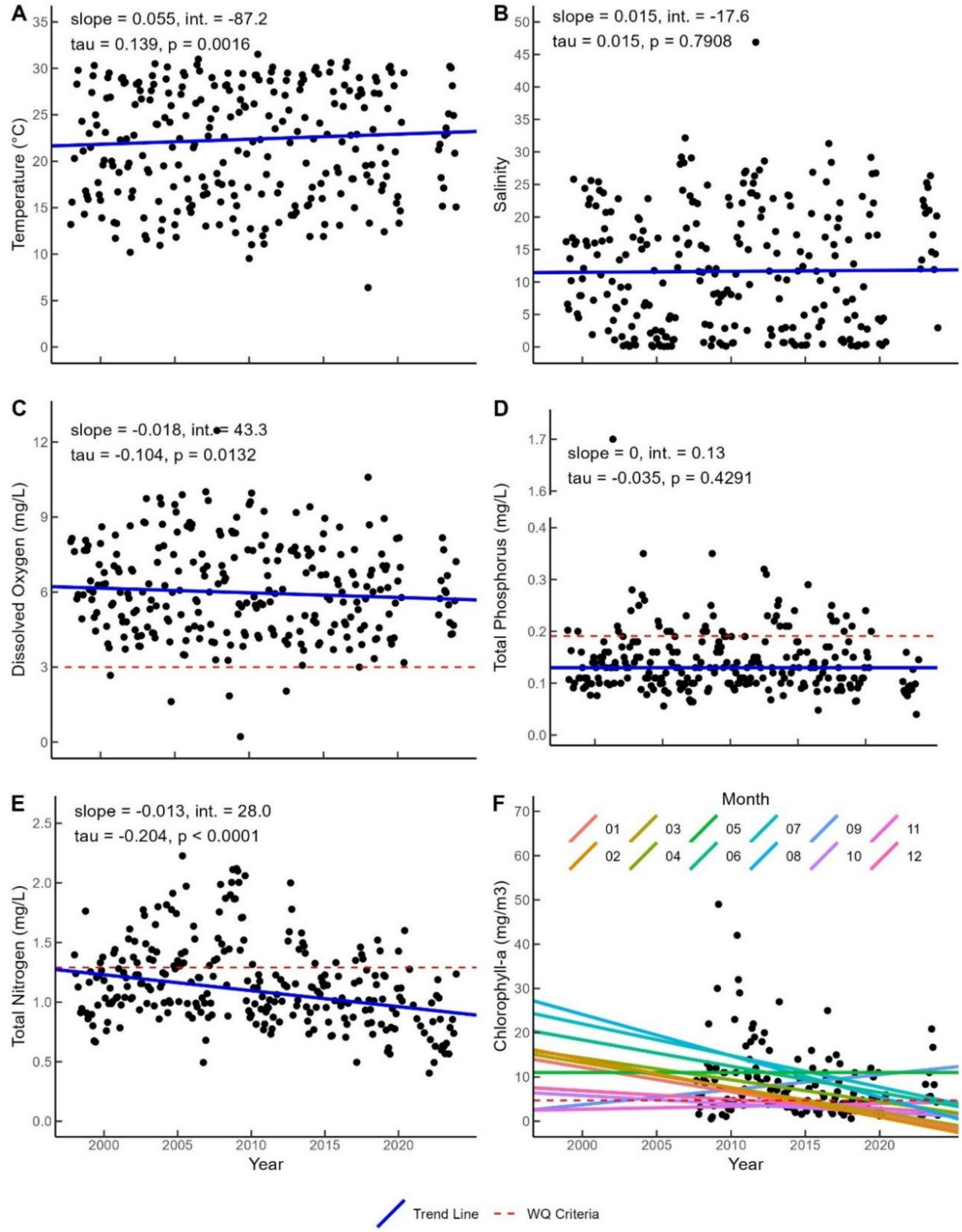

Figure 3.2.4. Trends for temperature, salinity, DO, TP, TN, and Chl-*a* (A-F) at the Nassau River station (19020002). Seasonal Kendall = temperature, salinity, DO, TP; Non-seasonal Kendall = TN, Chl-*a*. For months 01 = January, 02 = February, etc. The solid blue line depicts the monotonic trend for each parameter, with $p < 0.05$ denoting significant trends. P-values for Chl-*a* are listed in Table 3.2.4. The dashed red line indicates WQ criteria based on the 3.0 mg/L hypoxia stress threshold for DO and annual geometric mean values for TP, TN, and Chl-*a* outlined in F.A.C. Rule 62-302.532.

The only trend found at the **Nassau River east station (NCBGD)**, was a decrease in Chl-*a* (Table 3.2.3). Between 2014-2023, water temperature remained near 24.0°C and salinity near 32.5 (Fig. 3.2.5A-B). All individual values recorded for DO were at or above 3.0 mg/L, whereas individual values for TP and TN periodically exceeded their respective numeric nutrient criteria values (Fig. 3.2.5C-E). For Chl-*a*, concentrations decreased from 8.0 mg/m^3 in 2014 to 5.5 mg/m^3 in 2023, with most individual concentrations being below the nutrient criteria value for this location (Fig. 3.2.5F).

Similarly, only a decrease in Chl-*a* was found at the **Amelia River north station (NCBCM24)** (Table 3.2.3). From 2014 to 2023, water temperature and salinity remained around 24.0°C and 30.5, respectively (Fig. 3.2.6A-B). The trendline and all individual concentrations for DO were above 3.0 mg/L (Fig. 3.2.6C). While no change in TP or TN was detected, individual concentrations occasionally exceeded the nutrient criteria values (Fig. 3.2.6D-E). Over the 9 years, Chl-*a* concentrations decreased by 2 mg/m^3 (6.2 mg/m^3 to 4.2 mg/m^3), with only two individual concentrations exceeding the nutrient criteria value across all years (Fig. 3.2.6F).

At the **Amelia River south station (NCBNAAM)** decreases in TP and Chl-*a* were found (Table 3.2.3). Water temperature and salinity were stable from 2014 to 2023, with temperature near 24.0°C and salinity near 32 (Fig. 3.2.7A-B). Both the trendline and all individual concentrations for DO were above 3.0 mg/L (Fig. 3.2.7C). Across the 9-year period, TP concentrations decreased from 0.09 mg/L to 0.07 mg/L, with individual TP and TN concentrations occasionally exceeding their respective nutrient criteria values (Fig. 3.2.7D-E). Additionally, Chl-*a* decreased from 8.9 mg/m^3 in 2014 to 6.9 mg/m^3 in 2023, within individual concentrations periodically exceeding the nutrient criteria value (Fig. 3.2.7F).

Decreases in DO, Secchi, TN and Chl-*a* were found at **Sisters Creek (NCB19020038)** (Table 3.2.3). Water temperature remained near 24.9°C and salinity near 33 from 2014 to 2023 (Fig. 3.2.8A-B). Over the 9 years, DO decreased by 0.5 mg/L (6.7 mg/L to 6.2 mg/L), with all individual concentrations above 3.0 mg/L, and TN declined by 0.13 mg/L (0.50 mg/L to 0.37 mg/L) (Fig. 3.2.8C-D). Individual TP and TN concentrations periodically exceeded their respective nutrient criteria thresholds (Fig. 3.2.8D-E). Additionally, Chl-*a* concentrations decreased from 7.0 mg/m^3 in 2014 to 5.2 mg/m^3 in 2023, with individual concentrations occasionally exceeding the nutrient criteria value (Fig. 3.2.8F).

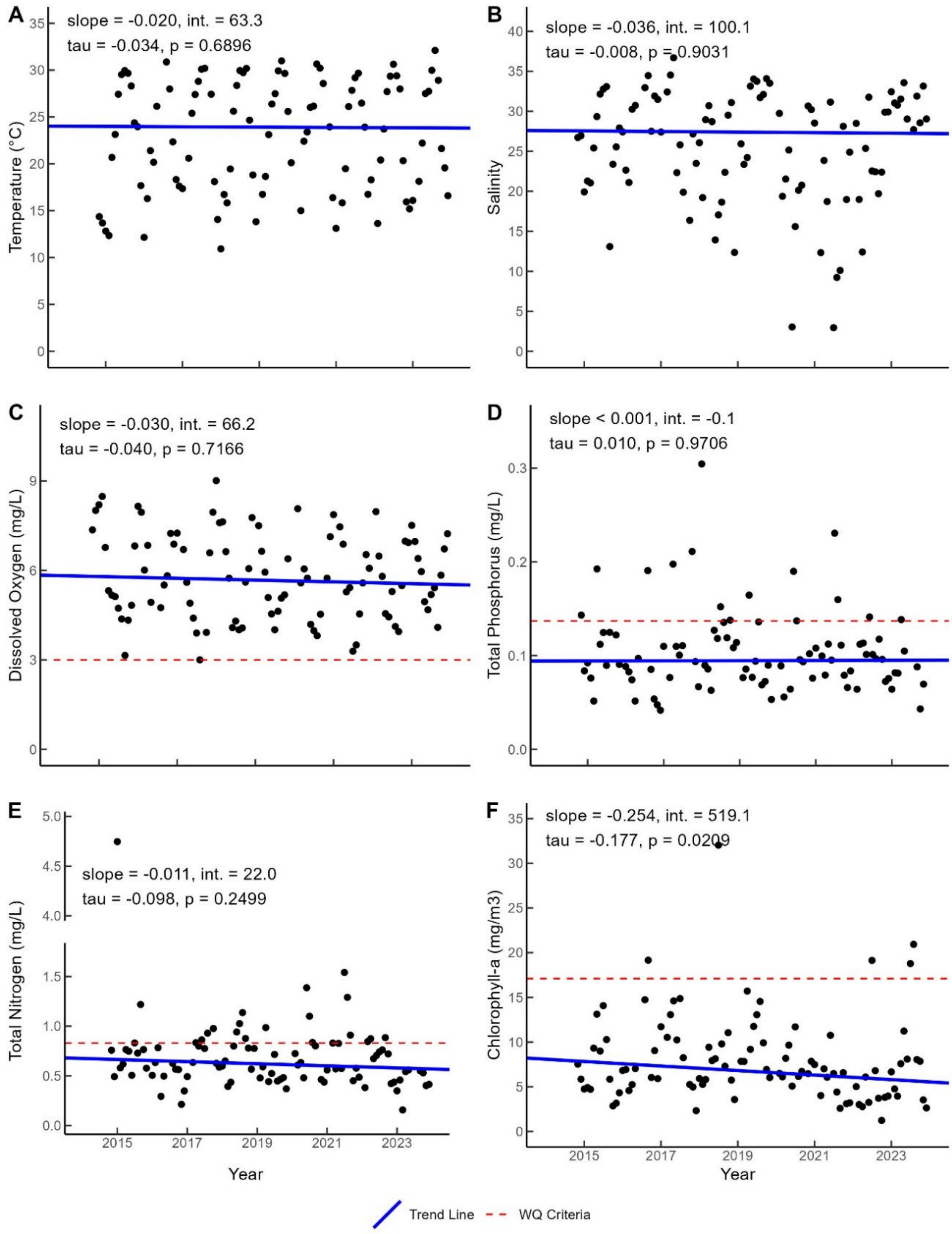

Figure 3.2.5. Trends for temperature, salinity, DO, TP, TN, and Chl-*a* (A-F) at the Nassau River station east (NCBGD). Seasonal Kendall = temperature, DO, TP, TN, Chl-*a*; Non-seasonal Kendall = salinity. The solid blue line depicts the monotonic trend for each parameter, with $p < 0.05$ denoting significant trends. The dashed red line indicates WQ criteria based on the 3.0 mg/L hypoxia stress threshold for DO and annual geometric mean values for TP, TN, and Chl-*a* outlined in F.A.C. Rule 62-302.532.

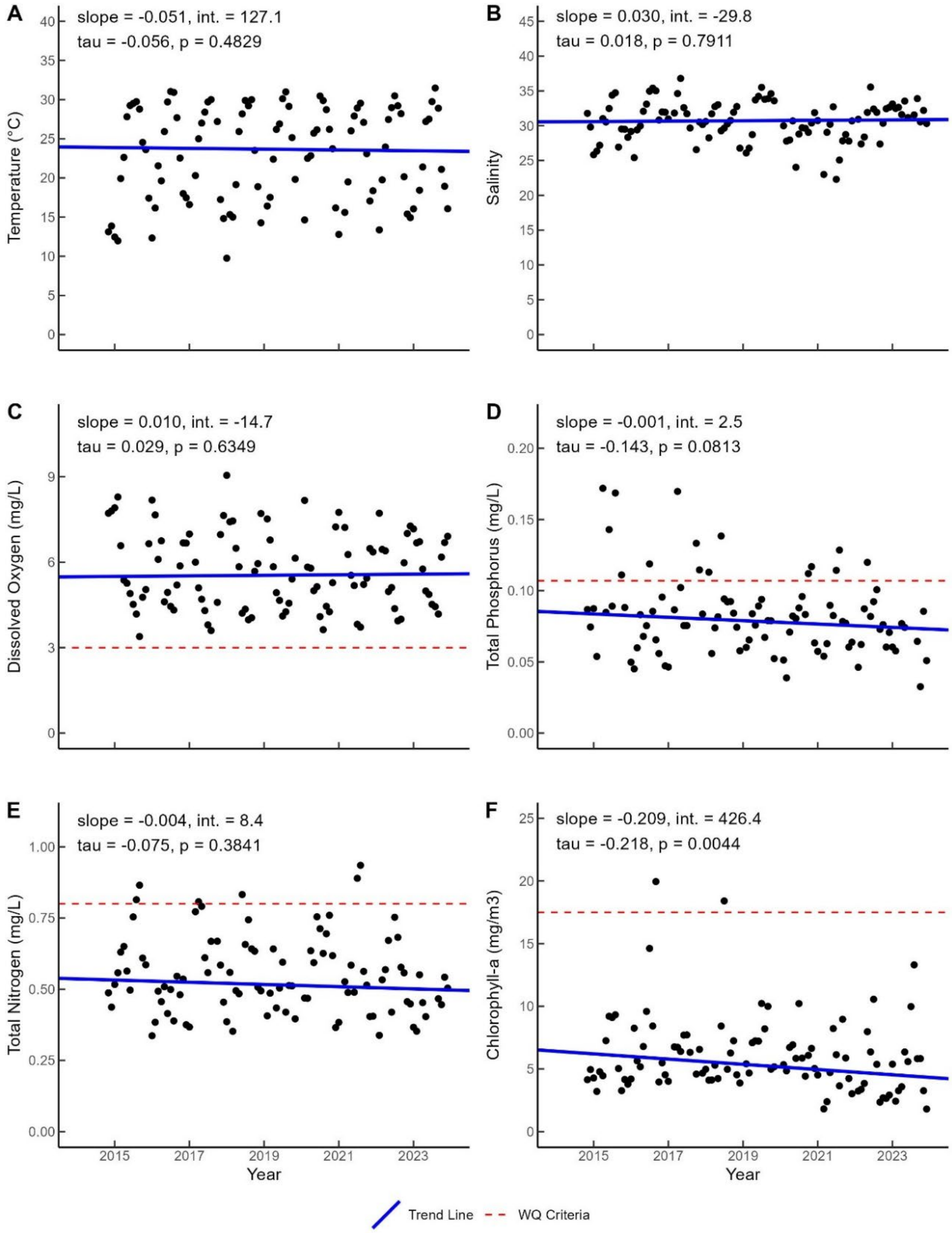

Figure 3.2.6. Trends for temperature, salinity, DO, TP, TN, and Chl-*a* (A-F) at the northern Amelia River station (NCBCM24). Seasonal Kendall = temperature, DO, TP, TN, Chl-*a*; Non-seasonal Kendall = salinity. The solid blue line depicts the monotonic trend for each parameter, with $p < 0.05$ denoting significant trends. The dashed red line indicates WQ criteria based on the 3.0 mg/L hypoxia stress threshold for DO and annual geometric mean values for TP, TN, and Chl-*a* outlined in F.A.C. Rule 62-302.532.

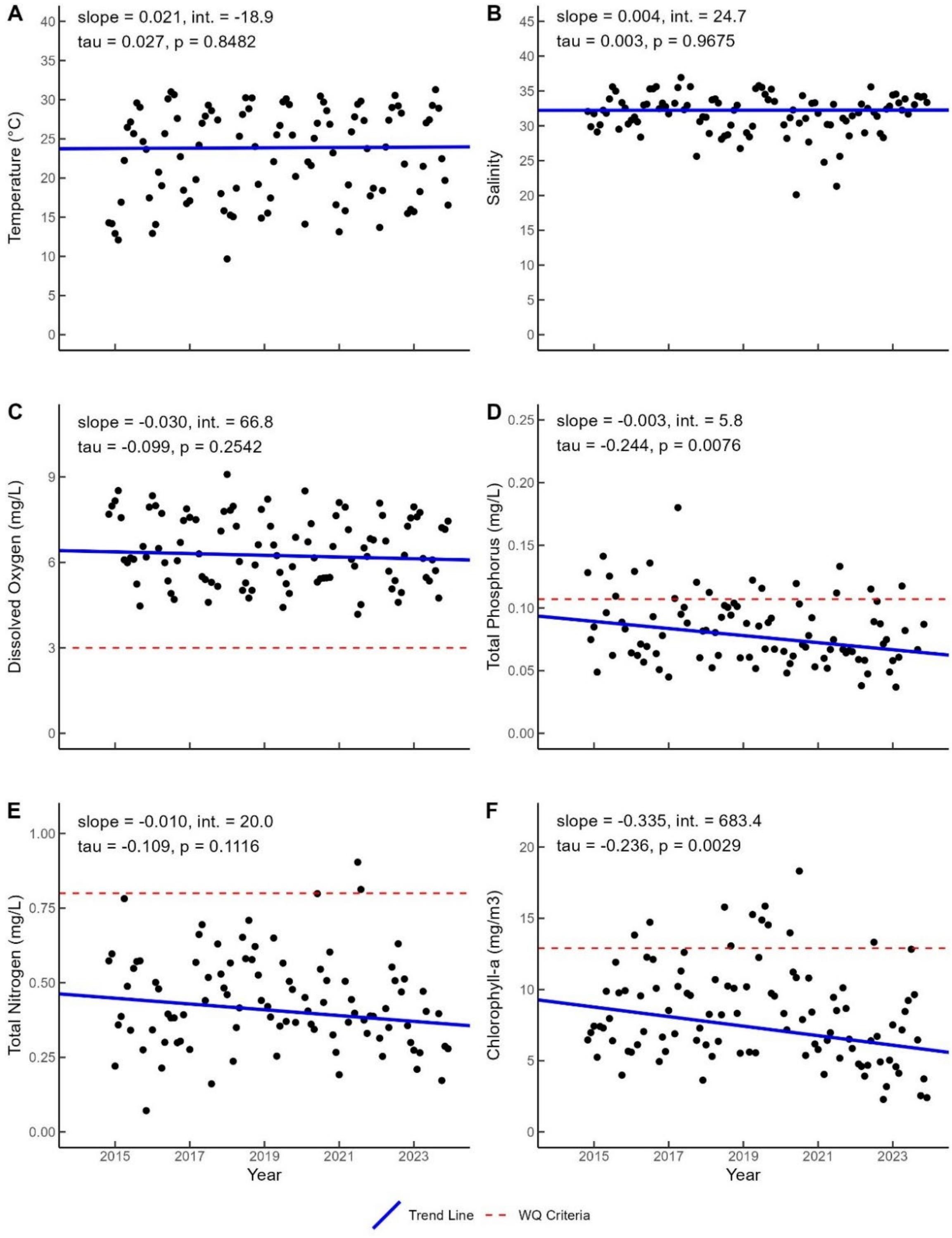

Figure 3.2.7. Trends for temperature, salinity, DO, TP, TN, and Chl-*a* (A-F) at the southern Amelia River station (NCBNAAM). Seasonal Kendall = temperature, DO, TP, Chl-*a*; Non-seasonal Kendall = salinity, TN. The solid blue line depicts the monotonic trend for each parameter, with $p < 0.05$ denoting significant trends. The dashed red line indicates WQ criteria based on the 3.0 mg/L hypoxia stress threshold for DO and annual geometric mean values for TP, TN, and Chl-*a* outlined in F.A.C. Rule 62-302.532.

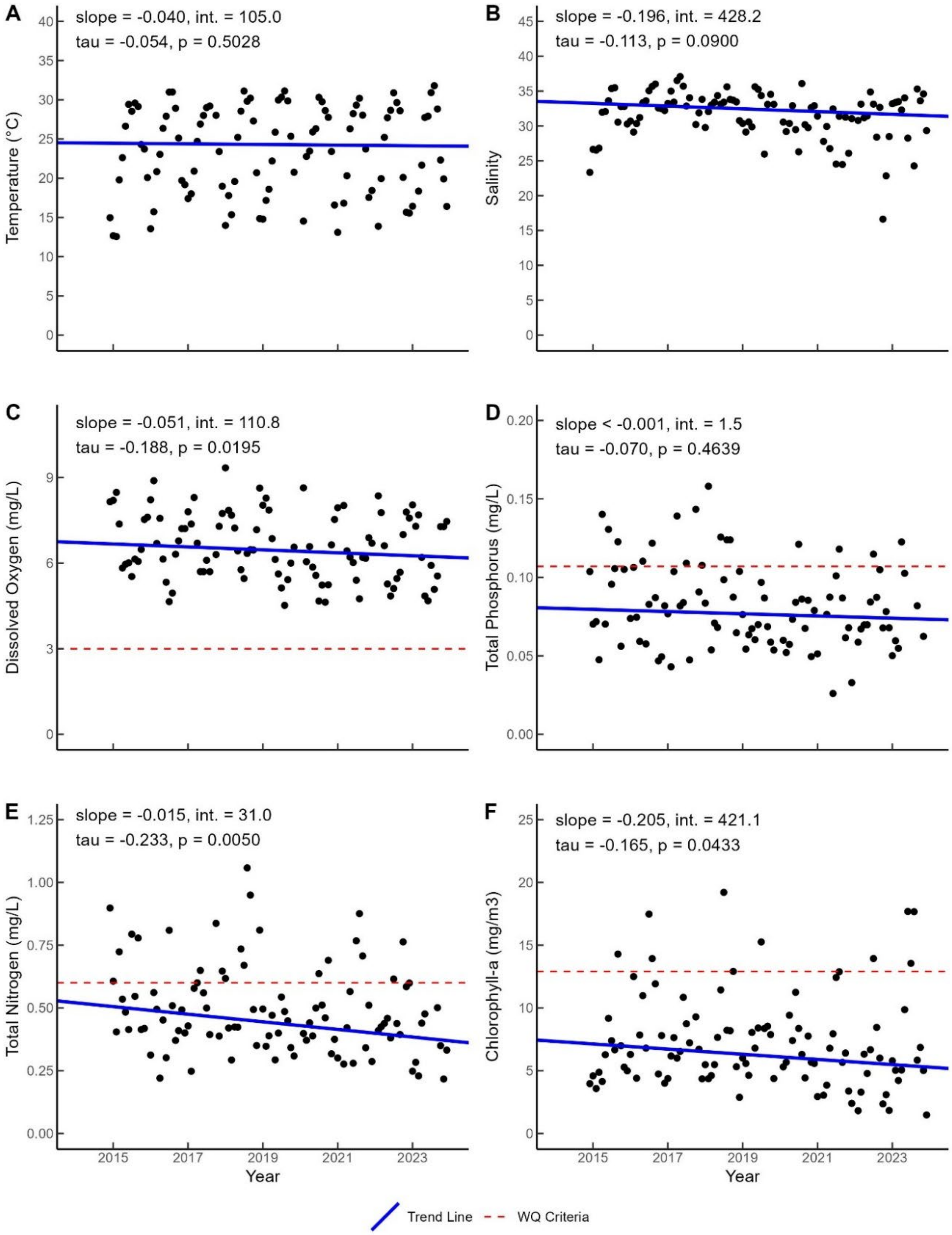

Figure 3.2.8. Trends for temperature, salinity, DO, TP, TN, and Chl-*a* (A-F) at the Sisters Creek station (NCB19020038). Seasonal Kendall = temperature, DO, TP, TN, Chl-*a*; Non-seasonal Kendall = salinity. The solid blue line depicts the monotonic trend for each parameter, with $p < 0.05$ denoting significant trends. The dashed red line indicates WQ criteria based on the 3.0 mg/L hypoxia stress threshold for DO and annual geometric mean values for TP, TN, and Chl-*a* outlined in F.A.C. Rule 62-302.532.

Discussion

Monitoring stations throughout the estuary captured a gradient of salinities with mean conditions ranging from oligohaline west of I-95 to euhaline in Amelia River and Sisters Creek. Lower pH and higher DOC and color in the upper estuary aligns with conditions reported for blackwater systems (Gallegos 2005, Blair et al. 2009, Flotemersch et al. 2024) and are consistent with findings from the Nassau River, which describe a broad freshwater-saltwater mixing zone between I-95 and US HW 17 (Coffin and Hampson 1992, Williams and Kimball 2013). The Nassau River is a small and shallow system, with salinities being greatly influenced by rainfall, freshwater discharge, evaporation and tidal regimes (Williams and Kimball 2013). This likely explains the high variability in salinity and other parameters (e.g., DOC and color) that are sensitive to freshwater inflow observed at the three Nassau River stations.

Weak trends (tau < 0.3) for one or more WQ parameters were found at each of the six NRPU stations. Some WQ parameters, such as water temperature, were expected to have small, gradual changes over time. For example, the average water warming rate reported for 346 global estuaries was 0.070 ± 0.004°C per year (Prum et al. 2024). Warming water temperatures in the upper Nassau estuary are further supported by studies which have reported rising temperatures within estuarine systems across the eastern United States (Oczkowski et al. 2015, Mallick and Dunn 2024) and South Florida (Shi and Hu 2022, Shi et al. 2024). Rising global and South Florida estuarine water temperatures have been related to increasing air temperatures (Prum et al. 2024, Shi et al. 2024). However, estuarine water temperatures and warming rates can also be influenced by factors such as landscape characteristics (e.g., vegetation cover) and land use, seawater temperature, and estuary characteristics (e.g., water depth, surface area) (Rice and Jastram 2015, Shi et al. 2022, Itsukushima et al 2024).

Increasing water temperatures may pose a management concern for NCB estuaries since temperature regulates a variety of biological and physiochemical processes such as estuarine community dynamics, ecosystem metabolism and productivity, water-atmospheric gas exchange (e.g., carbon dioxide, DO), and water chemistry (Mallick and Dunn 2024, Prum et al. 2024). In Florida, a tropical-temperate transition zone is located between 28° N–30° N, with more cold-tolerant temperate species to the north and more cold-sensitive subtropical species to the south (Kimball and Eash-Loucks 2021, Osland et al. 2021, Brewton and Lapointe 2023). Milder winter temperatures across this transition zone have been associated with the poleward expansion of subtropical species, including the western Atlantic sea bream (*Archosargus rhomboidalis*) in the Indian River Lagoon (Adams et al. 2024) and common snook (*Centropomus undecimalis*) in Cedar Key (29° N; Purtlebaugh et al. 2020). There is also evidence that warmer water temperatures can increase the vulnerability of riverine and marine environments to invasive species (Johnson et al. 2024, Wesselmann et al. 2024) and alter the migration patterns and spawning behavior of coastal fishes (Peer and Miller 2014, Slesinger et al. 2021). Differences in fish community composition resulting from subtropical species poleward expansion, biological invasions and changes to migratory and spawning behavior, raises concern over changes in community abundance, biodiversity, fisheries productivity, trophic interactions and potential negative socio-economic impacts to recreational and commercial fisheries.

Additionally, warming water temperatures can directly affect WQ by increasing metabolic rates which can increase microbial activity, decomposition rates, and biological oxygen demand, altering biogeochemical processes (e.g., carbon and nitrogen cycling) and enhancing risk of harmful algal blooms, hypoxia, and spread of pathogens (e.g., oyster *Vibrio vulnificus* infections) (Petes et al. 2012, Statham 2012, Nydahl et al. 2013, Wetz and Yoskowitz 2013, Prum et al. 2024). While warming water temperature was only found at the two stations in the upper estuary, these two stations also have 16-19 more years of data compared to the four stations in the lower estuary. It is possible other regions of the estuary are warming but require more data to be able to detect gradual temperature change. Time series analyses (Kendall trends) are more effective with longer data records (Meals et al. 2011) meaning shorter periods may reduce the likelihood of detecting significant changes, especially with WQ data which can have high interannual variability. This example highlights the importance of long-term WQ monitoring, as certain long-term trends may only become detectable after many years of data collection. It is unclear how warmer conditions may impact other aspects of the Nassau River and estuary, as relatively little is known about how estuarine ecosystems will respond to long-term spatial and temporal increases in atmospheric and water temperature (Itsukushima et al. 2024, Prum et al. 2024). This calls attention to a growing need for coordinated research efforts to investigate temperature-related changes within coastal estuarine environments (Kurylyk and Smith 2023). Monitoring ecosystem response and research on the effects of warming water temperatures on WQ is crucial for developing management plans that support heathy coastal habitats in the NCB.

Evidence of decreasing DO was found in both the upper and lower regions of the estuary. The DO decline in the upper estuary, near US HW 17, can be explained by several mechanisms including increasing water temperature and natural conditions of blackwater systems (e.g., high amounts of decomposing organic matter). Throughout the Nassau estuary DO was lowest during the summer to early fall (June-September) when water temperature was the warmest, as warmer conditions lower the solubility of DO (Chapra et al. 2021). Freshwater and low salinity regions of blackwater estuaries often exhibit low DO due to bacterial decomposition, a process intensified by warmer summer water temperatures (Todd et al. 2009, Kerr et al. 2013). More information is needed to understand why DO is declining in the lower region of the estuary since Sisters Creek did not have any coinciding changes in WQ parameters (e.g., temperature, salinity) that could explain the observed decrease. Despite DO declining, neither location currently appears to be at high risk of developing a DO impairment as most concentrations were above the 3.0 mg/L hypoxia stress threshold. However, continued monitoring of DO, especially in the upper region of the estuary, is important as continued warming trends may exacerbate naturally low DO conditions which can affect WQ and act as a stressor for biological communities. From a management perspective, prolonged periods of chronically low DO concentrations present a concern since hypoxic conditions can increase fish and oyster stress and mortality (Landsberg, et al. 2020, Coxe et al. 2023) and low DO concentrations can affect biogeochemical cycling and sediment-water fluxes of nutrients and metals (Kemp and Dodds 2002, Skoog and Arias-Esquivel 2009).

Decreasing nutrients and/or Chl-*a* was found for one or more stations in the upper and lower estuary. These results contrast with Williams and Kimball (2013) which reported increasing eutrophic conditions throughout the Nassau estuary from 1997 to 2011, with significant increases in TN, NH_4^+, NO_3^-, TKN and PO_4^{3-} but no change in TP over time. Differences between this study and Williams and Kimball (2013) can be explained by differences in monitoring station locations, period of record, statistical analyses and WQ parameters assessed. Observed decreases in Chl-*a* may be related to coinciding declines in TP, TN and NOx-D observed across stations. Chlorophyll concentrations are used as a proxy for phytoplankton biomass, which is sensitive to nutrient loading and considered an indicator of eutrophication within Florida estuarine systems (Doering et al. 2006, Boyer et al. 2009). Multiple studies have demonstrated correlations between nutrient reductions and Chl-*a* reductions (Lundsør et al. 2020, Codiga et al. 2022) highlighting the need to reduce nutrient eutrophication in these systems. While no TMDLs currently exist within the NRPU, the mechanism(s) behind recent nutrient declines remains unclear. Decreasing TSS may be a contributing factor, as suspended solids can act as a source or carrier for nutrients in estuarine systems (Paudel et al. 2019).

Long-term WQ trends in the NRPU indicate increasing water temperature in the upper estuary and declining DO, nutrients, and Chl-*a* at several locations. Continued long-term increases in water temperature and decreases in DO have the potential to alter physical, chemical and biological processes (Kemp and Dodds 2002, Skoog and Arias-Esquivel 2009, Mallick and Dunn 2024, Prum et al. 2024). Warming water temperatures raise concerns about changes to WQ (e.g., exacerbation of naturally low DO during the summer in the upper estuary) and biological communities which can affect habitat function and ecosystem services. In general, decreasing nutrient inputs can improve estuarine WQ by reducing the risk of eutrophication. While declines in nutrients and Chl-*a* were found throughout the estuary, most WQ monitoring stations also showed evidence of TP, TN or Chl-*a* concentrations periodically exceeding their respective nutrient criteria thresholds, especially in the upper estuary. Further reductions in nutrient input could be beneficial for maintaining good WQ and preventing future impairments. Further research is needed to understand the drivers of nutrient variability and decline, and analysis of additional nutrient compounds (e.g., NH_4^+) can aid in improving our understanding of nutrient trends throughout the NRPU.

3.3. Intracoastal Waterway Planning Units

Summary

- There are four SJRWMD WQ monitoring stations with the Intracoastal Waterway Planning Units (ICWPUs) 3K and 3I and contributing watershed. While the focus of the ICWPUs is estuarine areas to the north and south of the SJR (e.g., Sisters Creek, ICW) data from the contributing watershed, which overlaps with the SJR, was included to provide additional information on WQ throughout this region.

- Mean WQ parameters were consistent with findings typically reported for blackwater estuaries, with upstream stations having lower salinity and higher nutrients, DOC, color, and Chl-*a* compared to stations closer to the inlet and within the ICW.

- Each station showed trends for at least four WQ parameters. The only parameters to not have a significant trend at any station were water temperature and color.

- Decreases in salinity (and conductivity) may be explained by changes in sampling methodology, rainfall, water levels (water height/ depth) and the frequency of episodic storm events. Reductions in nutrients, suspended solids, and Chl-*a* can be explained by ongoing water management practices throughout the system.

- Changes to salinity, pH and DO concentrations may pose risks for oyster populations, but additional WQ data are needed for key areas within ICWPU 3K, particularly Clapboard and Sisters Creeks and other smaller tidal waterways, that support high oyster densities.

- Although nutrients and Chl-*a* are declining, ongoing impairments for nutrients, Chl-*a* and fecal coliforms in ICWPU 3I highlight the need for continued efforts to reduce nutrient and fecal bacteria inputs to improve WQ and address current impairments.

Introduction

The Florida ICW is part of the Atlantic ICW which extends approximately 2,253 km (1,400 mi) along the eastern coast of the United States from Key West, FL, to Boston, MA, with Florida's portion of the waterway being over 804 km (500 mi) long (Abecassis 2005, Crawford 2006). From 1882-1912 the Florida Coast Line Canal and Transportation Company dredged a series of inland canals collectively known as the "Florida East Coast Canal" to form a continuous waterway from Jacksonville to Miami (Abecassis 2005). In 1927, the Florida Inland Navigation District (FIND) was created by Florida Legislature which authorized the purchase of the East Coast Canal and established FIND as the local sponsor of the federally authorized Atlantic ICW project. Under the River and Harbor Act (1927) the ICW in Florida was widened and deepened to better fit transportation needs of larger commercial vessels (Abecassis 2005). Today, the ICW in northeast Florida has depths between 3 to 3.7 m and is a relatively narrow, sheltered waterway that is protected from strong winds (Kinnaird 1983).

The ICW Planning Units (ICWPUs) 3I and 3K span southeastern Duval County and into northern St. Johns County (Fig 3.3.1), bordered by the NRPU to the north and TRPU to the south. Their contributing watershed, the Lower St. Johns River (SJR), extends westward beyond the ICWPUs boundaries. Within these PUs, the ICW canal intersects with the SJR through Sisters Creek in the north and Pablo Creek in the south (Fig 3.3.1). The SJR is Florida's longest river spanning 500 km (310 mi.) from Blue Cypress Lake in Indian River County to the Atlantic Ocean east of Jacksonville and is one of the few rivers in the US that flows north (Bacopoulos et al. 2012). This blackwater river system discharges 235 m^3/ sec (8,300 ft^3/ sec) into the Atlantic Ocean (Miller 1998) and has a maximum tidal range of 1.5 m (4.9 ft; Bacopoulos et al. 2012, Dix et al. 2021). Tributaries and waterways to the northeast and southeast of the SJR include Clapboard Creek, Pablo Creek, Sisters Creek, and the ICW itself. Additionally, the Timucuan Ecological and Historic Preserve and Pumpkin Hill Creek State Buffer Preserve in the northeast are designated OFWs. Salt marshes, which are primarily found northeast of the SJR and bordering the ICW in the southeast, include a mix of black needlerush (*Juncus roemerianus*) and smooth cordgrass (*Spartina alterniflora*). Mangroves have also recently migrated into the area. Additionally, oyster reefs can be found interspersed throughout tidal creeks in the ICW, however, seawalls limit their spatial extent (Dix, 2019). The intended focus of the ICWPUs are the estuarine creeks and the ICW east of the SJR rather than the river itself. While the ICWPUs overlap with the SJR, the river mainstem is managed separately and has a Basin Management Action Plan (BMAP) to reduce nutrient loading and restore WQ throughout the system.

Methods

The analyses used within the two ICWPUs and contributing watershed follow those outlined in Chapter 2. Although the ICWPUs focus on estuarine areas east of the SJR (e.g., Sisters Creek, ICW), data from stations near Charter Point (JAXSJR17) and San Jose (JAXSJR40) were included to provide additional context, as they fall within the contributing watershed and overlap with mesohaline and oligohaline regions of the Lower SJR.

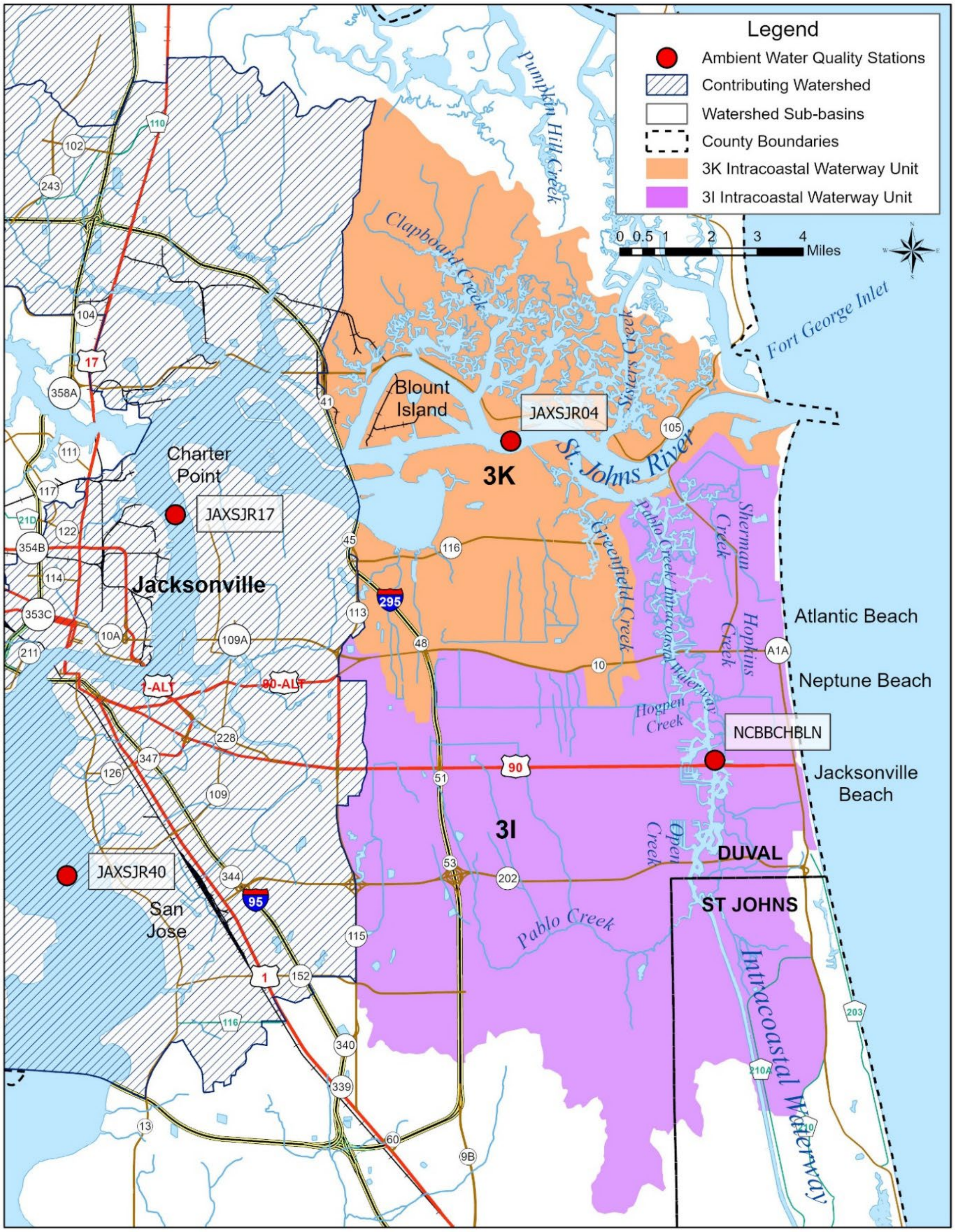

Figure 3.3.1. Map of ICWPUs 3K (orange) and 3I (purple). Red circles indicate SJRWMD WQ monitoring stations within the PUs and contributing watershed.

Results

Mean Water Quality Conditions

There are four SJRWMD WQ monitoring stations located within ICWPUs 3K/3I and contributing watershed (Fig 3.3.1). Blount Island (JAXSJR04) is the longest running station with sampling beginning in 1994, followed by the Charter Point (JAXSJR17) and San Jose (JAXSJR40) stations which began in 1995 (Table 3.3.1, Table 3.3.2). The ICW station (NCBBCHBLN, formerly NCB27010116) is the most recently established, with sampling beginning in November 2014. Comparison of mean WQ parameters across stations suggested similarities in water temperature, pH, Secchi and TP (Table 3.3.1, Table 3.3.2). Of the four stations, DO was highest at the San Jose and lowest at the ICW. At the Charter Point and San Jose stations, mean salinity and conductivity were lower and mean color was higher compared to the Blount Island and ICW station. Additionally, mean TN, NOx-D, Chl-*a*, and TSS were higher at the Charter Point and San Jose stations compared to Blount Island and ICW.

Monthly Variability

KW tests showed that most parameters varied across months at each station (Table 3.3.3). However, no monthly differences were found for TSS at the San Jose and Charter Point stations; salinity, conductivity and Secchi at the Blount Island station; or DOC and color at the ICW station.

Station Water Quality Trends and Criteria

At the **San Jose station (JAXSJR40)** decreases in pH, DO, Secchi, TN, TSS, and a marginally significant decline in Chl-*a* were found (Table 3.3.3). The trendline for water temperature remained near 24.5°C from 1995 to 2023 and for salinity near 5 (oligohaline) from 1998 to 2023 (Fig. 3.1.2A-B). All individual DO concentrations and the trend were above 3.0 mg/L (Fig. 3.1.2C). The Seasonal Kendall test performed on TP had significant heterogeneity (Chi-Square Heterogeneity Test, $p = 0.0417$), with all months having negative trends except January, February, March and December where trends were positive and trends being significant for June, July, September, October and November (Table 3.3.4., Fig. 3.1.2D). Over the 28-year period of record (1995-2023) TN decreased from 1.30 mg/L to 1.05 mg/L and Chl-*a* from 8.8 mg/m^3 to 7.0 mg/m^3 (Fig. 3.1.2E-F), with the trend for Chl-*a* and many individual concentrations exceeding the nutrient criteria value.

Table 3.3.1. Mean (±) standard deviation of physical WQ parameters for stations within ICWPU 3K and 3I.

Station	Years	Temp. (°C)	Sal.	Cond. (µmhos/cm)	pH	DO (mg/L)	Secchi (m)
JAXSJR40	1995 - 2023	23.49 ± 5.88	6.23 ± 5.75	10380.86 ± 9223.79	7.84 ± 0.20	8.10 ± 1.34	0.78 ± 0.30
JAXSJR17	1995 - 2023	23.22 ± 5.68	16.88 ± 7.39	26947.47 ± 11275.58	7.84 ± 0.22	7.35 ± 1.58	0.98 ± 0.46
JAXSJR04	1994 - 2023	22.79 ± 5.52	28.64 ± 5.16	44123.81 ± 7683.17	7.97 ± 0.16	7.13 ± 1.38	1.17 ± 0.43
NCBBCHBLN	2014 - 2023	23.19 ± 5.77	21.51 ± 5.32	34169.79 ± 7736.14	7.53 ± 0.20	6.01 ± 1.37	0.87 ± 0.29

Table 3.3.2. Mean (±) standard deviation of physical WQ parameters for stations within ICWPU 3K and 3I.

Station	Years	TP (mg/L)	TN (mg/L)	NOx-D (mg/L)	DOC (mg/L)	Color (PCU)	TSS (mg/L)	Chl-*a* (mg/m^3)
JAXSJR40	1995 - 2023	0.12 ± 0.03	1.23 ± 0.33	0.20 ± 0.14	15.87 ± 4.12	118.37 ± 85.01	17.11 ± 10.51	10.49 ± 8.77
JAXSJR17	1995 - 2023	0.15 ± 0.05	1.12 ± 0.41	0.22 ± 0.12	12.77 ± 4.67	88.50 ± 74.03	31.34 ± 34.50	7.08 ± 6.20
JAXSJR04	1994 - 2023	0.12 ± 0.05	0.81 ± 0.38	0.15 ± 0.10	8.90 ± 4.30	58.07 ± 57.10	27.35 ± 26.51	5.83 ± 3.99
NCBBCHBLN	2014 - 2023	0.10 ± 0.03	0.66 ± 0.15	0.08 ± 0.05	8.82 ± 2.15	50.39 ± 22.96	13.43 ± 8.91	4.90 ± 3.11

Table 3.3.3. Summary statistics for physical and chemical parameters at WQ monitoring stations in ICWPU 3K and 3I.

	Temp. (°C)	Sal.	Cond. (μmhos/cm)	pH	DO (mg/L)	Secchi (m)	TP (mg/L)	TN (mg/L)	NOx-D (mg/L)	DOC (mg/L)	Color (PCU)	TSS (mg/L)	Chl-a (mg/m^3)
	Station JAXSJR40 (San Jose)												
KW (p)	**< 0.0001**	**< 0.0001**	**< 0.0001**	**0.0242**	**< 0.0001**	**< 0.0001**	**< 0.0001**	**< 0.0001**	**< 0.0001**	**< 0.0001**	**< 0.0001**	**0.0412**	**< 0.0001**
Kendall tau	0.007	-0.066	-0.028	-0.125	-0.146	-0.196	-	-0.213	-0.062	0.031	-0.019	-0.164	-0.077
Kendall (p)	0.8919	0.1275	0.5191	**0.0024**	**0.0004**	**< 0.0001**	-	**< 0.0001**	0.1322	0.4537	0.6625	**< 0.0001**	**0.0454**
	Station JAXSJR17 (Charter Point)												
KW (p)	**< 0.0001**	**0.0004**	**0.0019**	**< 0.0001**	**< 0.0001**	**< 0.0001**	**0.0002**	**< 0.0001**	**< 0.0001**	**< 0.0001**	**< 0.0001**	0.5492	**< 0.0001**
Kendall tau	-0.009	-0.121	-0.096	-0.229	-0.173	-0.117	-0.314	-0.271	-0.135	0.036	0.016	-0.398	-0.297
Kendall (p)	0.8216	**0.0039**	**0.0196**	**< 0.0001**	**< 0.0001**	**0.0064**	**< 0.0001**	**< 0.0001**	**0.0016**	0.3586	0.6641	**< 0.0001**	**< 0.0001**
	Station JAXSJR04 (Blount Island)												
KW (p)	**< 0.0001**	0.3448	0.2970	**< 0.0001**	**< 0.0001**	0.0698	**0.0003**	**< 0.0001**	**< 0.0001**	**< 0.0001**	**< 0.0001**	0.9919	**0.0009**
Kendall tau	0.066	-0.195	-0.181	-0.334	-0.224	-0.216	-0.195	-0.179	-0.038	0.081	0.028	-0.483	-0.443
Kendall (p)	0.1154	**< 0.0001**	**< 0.0001**	**< 0.0001**	**< 0.0001**	**< 0.0001**	**< 0.0001**	**< 0.0001**	0.4098	**0.0430**	0.4761	**< 0.0001**	**< 0.0001**
	Station NCBBCHBLN (ICW)												
KW (p)	**< 0.0001**	**0.0223**	**0.0184**	**0.0006**	**< 0.0001**	**0.0298**	**0.0021**	**0.0346**	**< 0.0001**	0.3166	0.3657	**0.0004**	**< 0.0001**
Kendall tau	0.020	0.010	0.010	0.293	0.164	0.012	-0.189	-0.222	-0.025	-0.056	0.081	0.010	0.062
Kendall (p)	0.7964	1.0000	1.0000	**0.0004**	**0.0421**	0.8229	**0.0316**	**0.0082**	0.9405	0.4244	0.2413	0.8957	0.4258

If $p < 0.05$ for the Kruskal–Wallis (KW) test, a Seasonal Kendall test was performed. If $p \geq 0.05$ for the KW test, a Mann-Kendall test was performed. Significant test results ($p < 0.05$) are shown in bold.

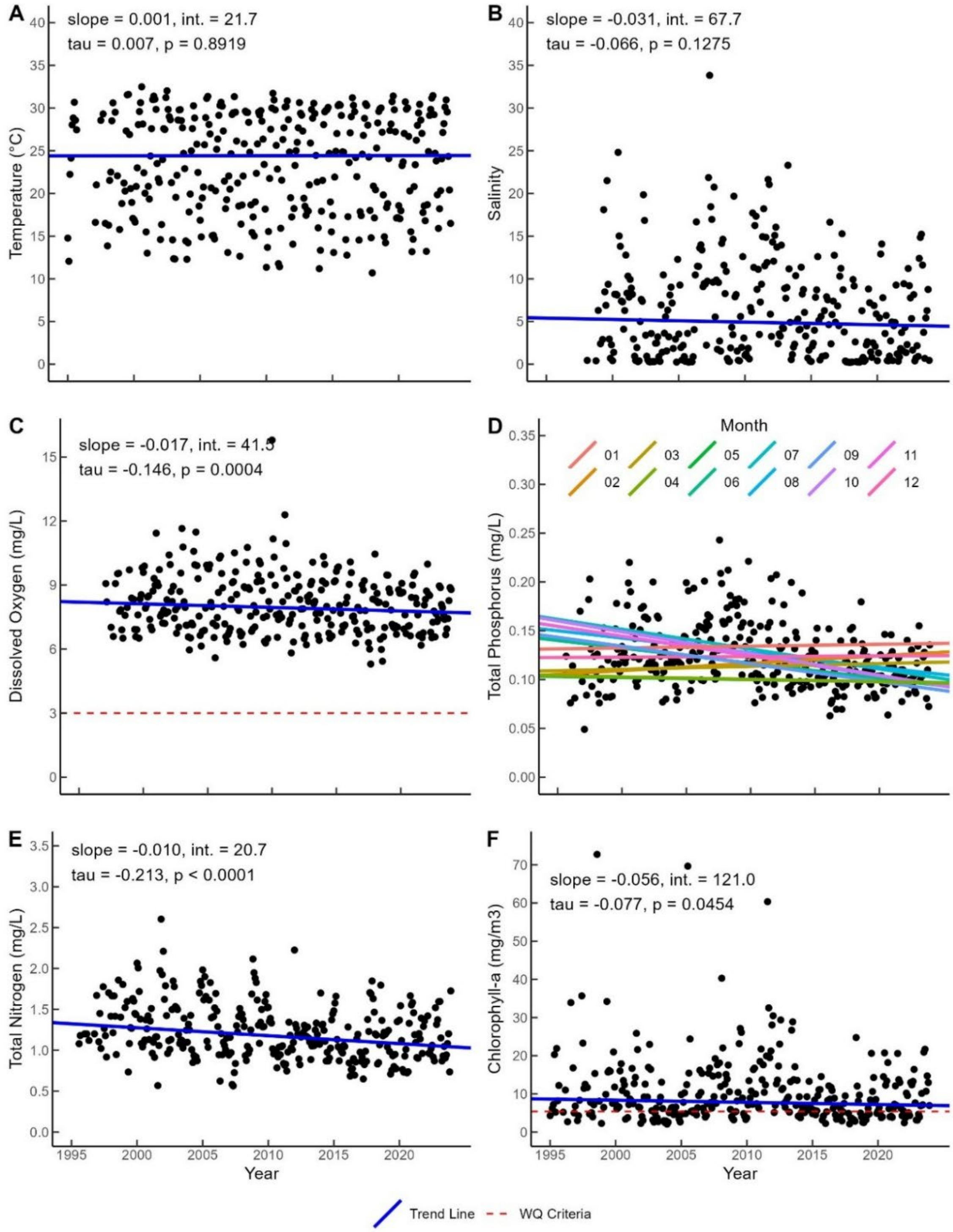

Figure 3.3.2. Trends for temperature, salinity, DO, TP, TN*,* and Chl-*a* (A-F) at the San Jose station (JAXSJR40). Seasonal Kendall = temperature, salinity, DO, TN, Chl-*a*; Non-seasonal Kendall = TP. For months 01 = January, 02 = February, etc. The solid blue line depicts the monotonic trend for each parameter, with p < 0.05 denoting significant trends. P-values for TP are listed in Table 3.3.4. The dashed red line indicates WQ criteria based on the 3.0 mg/L hypoxia stress threshold for DO and annual geometric mean value for Chl-*a* outlined in F.A.C. Rule 62-302.532.

Table 3.3.4. Results for individual monthly Mann-Kendall tests run for TP (mg/L) at the San Jose station (JAXSJR40).

Month	tau	p-value
January	0.036	0.8233
February	0.089	0.5371
March	0.048	0.7387
April	-0.071	0.6168
May	-0.083	0.5592
June	-0.282	**0.0411**
July	-0.397	**0.0048**
August	-0.237	0.0937
September	-0.385	**0.0063**
October	-0.390	**0.0046**
November	-0.317	**0.0246**
December	0.007	0.9802

Significant test results ($p < 0.05$) are shown in bold.

Trend analyses at the **Charter Point station (JAXSJR17)** showed decreasing trends for salinity, conductivity, pH, DO, Secchi, TP, TN, NOx-D, TSS, and Chl-*a* (Table 3.3.3). Water temperature was around 23.5 °C across the period of record (1995-2023), whereas salinity decreased from 17.5 to 14.5 between 1998-2023 (Fig. 3.3.3A-B). Both the trendline and all individual DO concentrations were above 3.0 mg/L (Fig. 3.3.3C). Concentrations decreased for TP from 0.17 mg/L to 0.12 mg/L and for TN from 1.25 mg/L to 0.85 mg/L between 1996-2023 (Fig. 3.3.3D-E). Chl-*a* concentrations decreased from 7.0 mg/m^3 in 1995 to 3.5 mg/m^3 in 2023 (Fig. 3.3.3F). Many individual Chl-*a* values exceeded the nutrient criteria, with the trendline only dropping below the threshold around 2010.

Evidence of decreasing salinity, conductivity, pH, DO, Secchi, TP, TN, TSS, and Chl-*a* and marginally decreasing DOC was found at the **Blount Island station (JAXSJR04)** (Table 3.3.3). Water temperature remained near 23.0°C from 1994 to 2023 whereas salinity decreased from 32 in 1998 to 27.7 in 2023 (Fig. 3.3.4A-B). All individual DO measurements and trendline were above 3.0 mg/L (Fig. 3.3.4C). Over the 29-year period of record (1994-2023), TP decreased from 1.3 mg/L to 0.95 mg/L, TN from 0.90 mg/L to 0.65 mg/L and Chl-*a* from 7.2 mg/m^3 in 1994 to 3.0 mg/m^3 (Fig. 3.3.4D-F). Individual Chl-*a* concentrations periodically exceeded the nutrient criteria value with the trendline falling below this value around 2010.

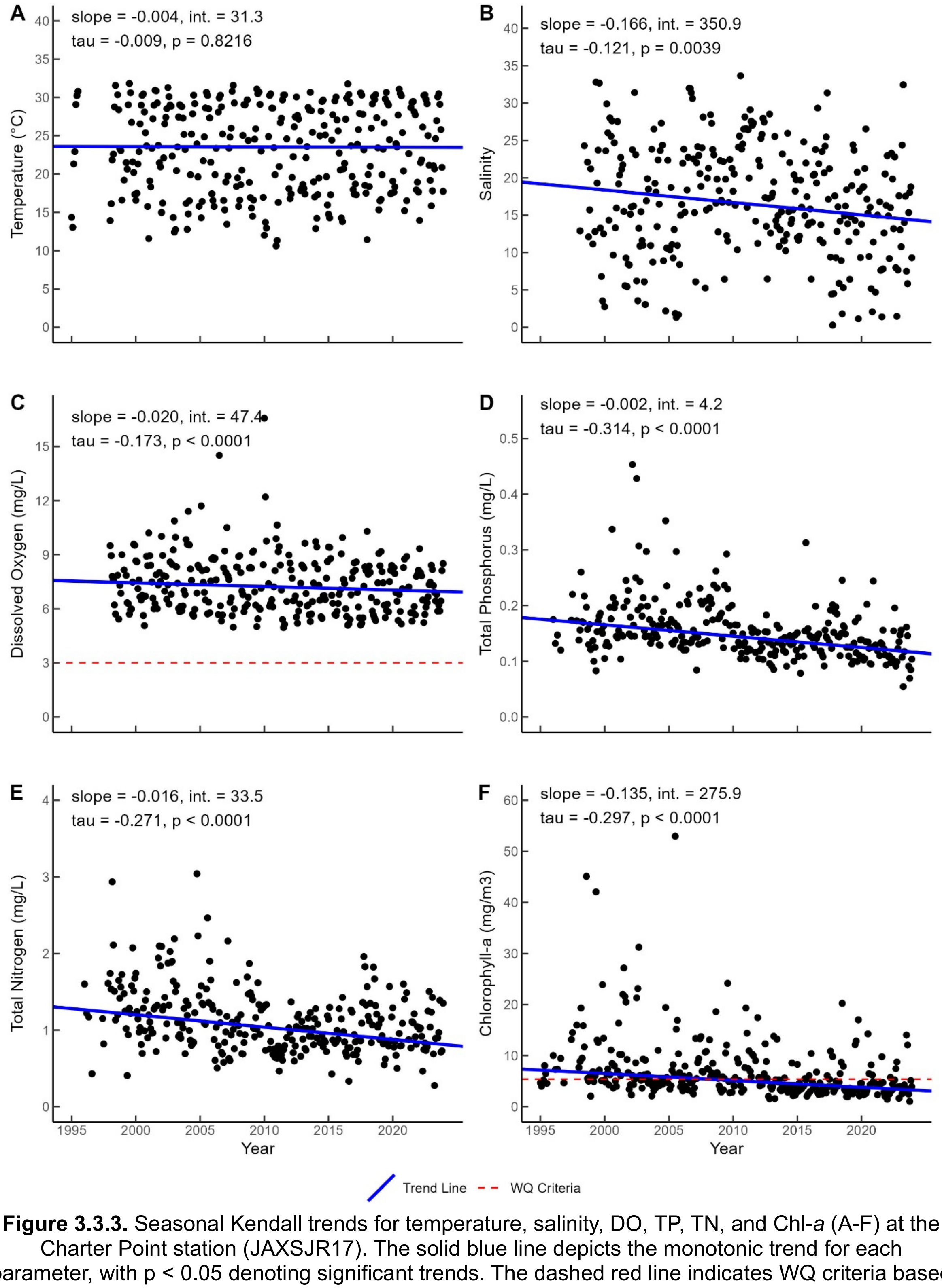

Figure 3.3.3. Seasonal Kendall trends for temperature, salinity, DO, TP, TN, and Chl-*a* (A-F) at the Charter Point station (JAXSJR17). The solid blue line depicts the monotonic trend for each parameter, with $p < 0.05$ denoting significant trends. The dashed red line indicates WQ criteria based on the 3.0 mg/L hypoxia stress threshold for DO and annual geometric mean value for Chl-*a* outlined in F.A.C. Rule 62-302.532.

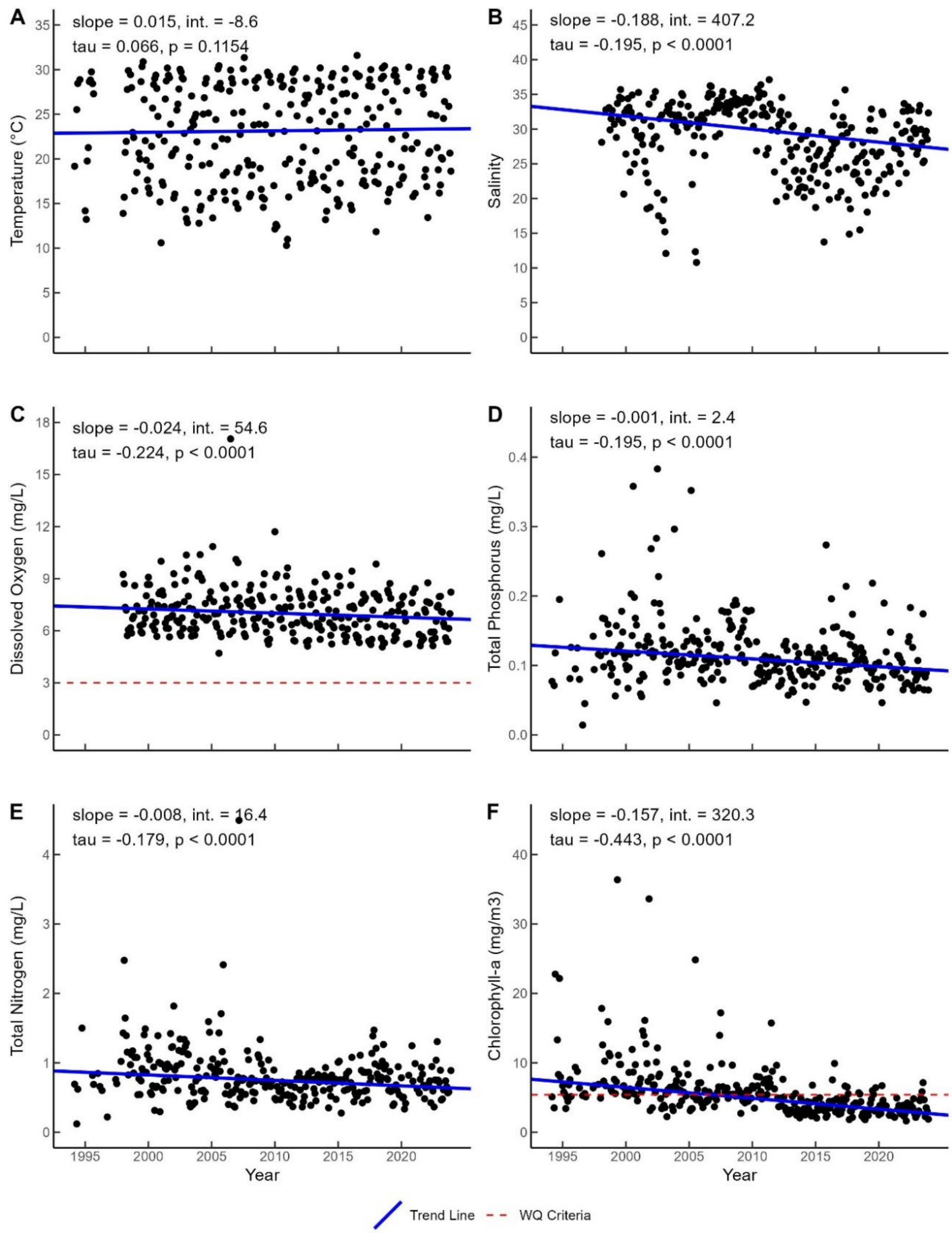

Figure 3.3.4. Trends for temperature, salinity, DO, TP, TN, and Chl-*a* (A-F) at the Blount Island station (JAXSJR04). Seasonal Kendall = temperature, DO, TP, TN, Chl-*a*; Non-seasonal Kendall = salinity. The solid blue line depicts the monotonic trend for each parameter, with $p < 0.05$ denoting significant trends. The dashed red line indicates WQ criteria based on the 3.0 mg/L hypoxia stress threshold for DO and annual geometric mean value for Chl-*a* outlined in F.A.C. Rule 62-302.532.

At the **ICW station (NCBBCHBLN)**, pH increased, DO showed a marginally significant increase, and both TP and TN concentrations declined (Table 3.3.3). Water temperature and salinity trends were stable and remained near 24.0°C and 22 (polyhaline), respectively across the 9-year period (2014-2023) (Fig. 3.1.5A-B). All individually measured DO concentrations were above 3.0 mg/L, with the trendline increasing from 5.7 mg/L in 2014 to 6.1 mg/L in 2023 (Fig. 3.1.5C). TP concentrations declined from 0.10 in 2014 to 0.08 in 2023, and TN from 0.72 in 2014 to 0.59 over the same period, with most TP and all TN values falling below their respective nutrient criteria (Fig. 3.1.5D-E). Additionally, Chl-*a* was stable across years, with most values being below the nutrient criteria for this location (Fig. 3.1.5F).

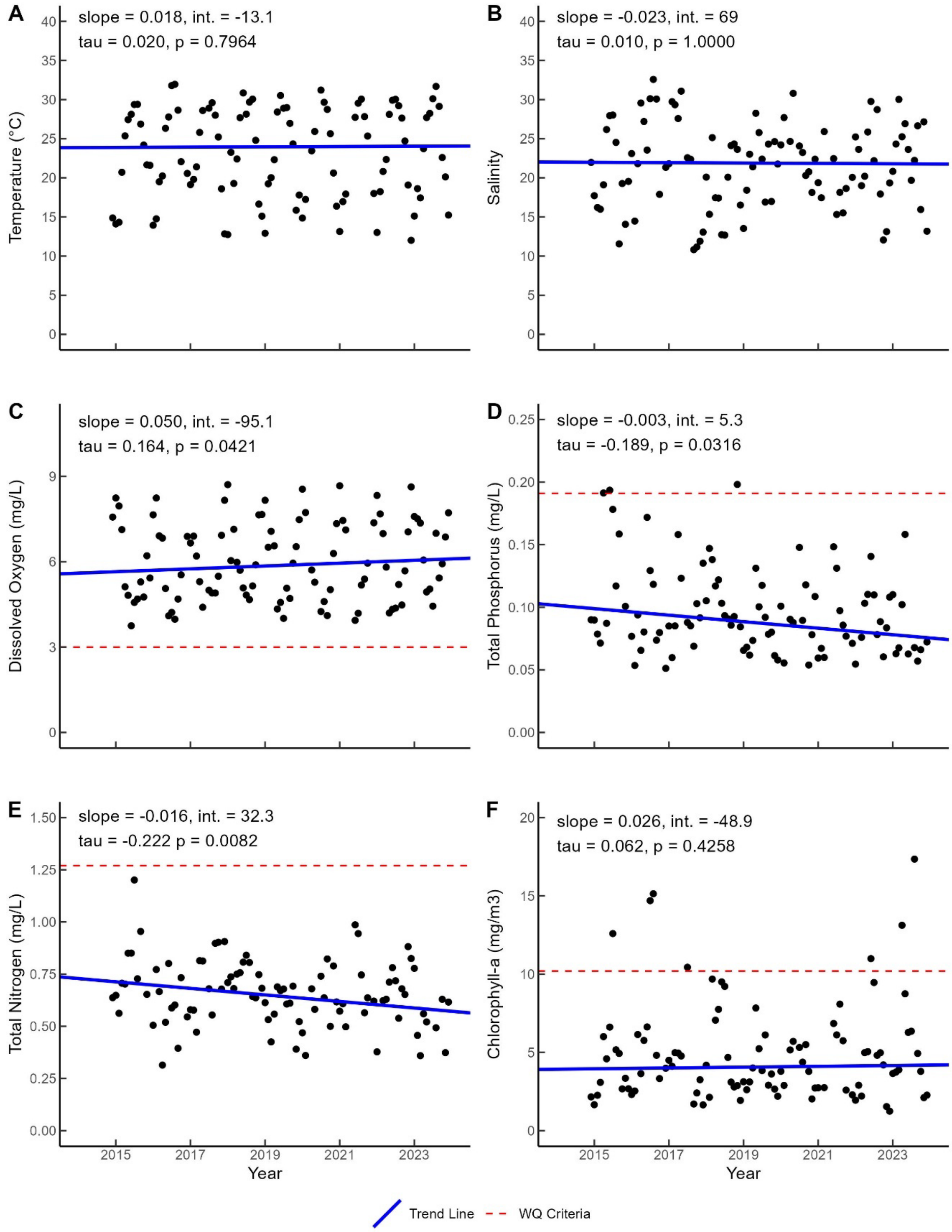

Figure 3.3.5. Respective Mann-Kendall or Seasonal Kendall trends for temperature, salinity, DO, TP, TN, and Chl-*a* (A-F) at the ICW station (NCBBCHBLN). The solid blue line depicts the monotonic trend for each parameter, with $p < 0.05$ denoting significant trends. The dashed red line indicates WQ criteria based on the 3.0 mg/L hypoxia stress threshold for DO and annual geometric mean values for TP, TN, and Chl-*a* outlined in F.A.C. Rule 62-302.532.

Discussion

Across the two ICWPUs, there were differences in mean salinity among stations. At the Blount Island and ICW stations, mean salinities were polyhaline whereas salinities at San Jose and Charter Point stations were mesohaline. These findings are supported by other studies reporting average salinity conditions ranging from oligohaline to mesohaline along a 35 km stretch from Doctors Lake (south of ICWPUs) to southern Jacksonville and transitioning from mesohaline to polyhaline over the next 40 km from Jacksonville to the river mouth (Mason 1998, Cooksey and Hyland 2007, Pinto et al. 2023). Additionally, observations of higher mean DOC and color at the two lower salinity stations align with conditions reported for blackwater estuarine systems (Gallegos 2005, Blair et al. 2009, Chen et al. 2015). The high variability observed for WQ parameters, such as color and TSS, was expected since these parameters are sensitive to changes in freshwater inflow and tidal mixing (FLW 2004, Pinto et al. 2024).

Significant differences in WQ trends were found for several parameters at each station, with the strength of observed changes ranging from low (tau < 0.3) to moderate (0.3 ≤ tau < 0.5). Decreases in salinity (and conductivity) at the Charter Point and Blount Island stations may be explained by changes in sampling methodology (e.g., tide-based sampling protocols; personal communication with R. Timbs and SJRWMDs Bureau of Water Resource Information), rainfall, water levels (water height/ depth) and the frequency of episodic storm events. Precipitation associated with the El Niño-Southern Oscillation and hurricanes are known to affect salinity regimes within estuarine systems (Schmidt and Luther 2002, Sumner and Belaineh 2005, Feher et al. 2023). In the SJR, periods of high precipitation raise water levels and increase freshwater outflow, whereas low precipitation lower water levels, allowing for greater ocean water intrusion (Pinto et al. 2024). For example, strong La Niña events from 1998 to 2001, and 2006 to 2008, brought extreme drought conditions throughout much of northeast Florida and contributed to increased salinity within the SJR and estuary (Verdi et al. 2006, Sagan 2007). Additionally, large scale storms such as Hurricanes Matthew (2016) and Irma (2017) can also influence water levels though tidal storm surge and high rainfall, which can alter salinity in the system (Bacopoulos 2017, Bielmyer-Fraser 2020).

Notably, there was no change in salinity across years at the ICW station. Differences in WQ trends between Blount Island in ICWPU 3K and the ICW in ICWPU 3I, may be due to varying periods of record (29 vs 9 years), as Kendall tests are more effective at detecting trends in longer dataset (Meals et al. 2011), or site location. Differences resulting from location are an important consideration since the only station in the ICWPU 3K is located within the SJR, which is not the primary focus of this PU. It is possible estuarine waters to the northeast (e.g., Clapboard Creek, Sisters Creek) have different WQ trends and/or responses to changing environmental conditions (e.g., precipitation) and management practices (e.g., nutrient load reductions) compared to the SJR. Further supporting this idea is that there was no change in salinity at the NRPU Sisters Creek station (see Chapter 3.2, Figure 3.2.8), which is located near the northern border of the ICWPU 3K, but this station was also limited to 9 years of data. Currently more information, either through additional analysis or sampling, is needed to determine if WQ is different between Blount Island and the tidal creeks northeast of the SJR, which are the intended focus of the ICWPU 3K.

Abrupt or prolonged changes to salinity present a management concern due to the adverse effects it can have on aquatic organisms (Guenther and MacDonald 2012, Bielmyer-Fraser 2020). For example, extended periods of low salinity (≤ 5) can negatively impact oyster recruitment, survival and growth (La Peyre et al. 2013, Parker et al. 2013), while extended periods of high salinity (e.g., 33), especially when water temperature is warm, can increase oyster disease-related mortality (Petes et al. 2012). However, higher salinities have also been associated with great oyster faunal community abundance, biomass, and diversity (Tolley et al. 2005, Marshall et al. 2019). Across the ICWPUs, extensive oyster reefs can be found in tidal tributaries such as Clapboard and Sisters Creeks and other smaller tidal waterways connected to the ICW (Dix et al. 2019). While salinity appears to be stable at the ICW WQ monitoring station, potential shifts in freshwater inflow and salinity regimes are an important consideration for management of oyster habitat within this system, as well as throughout the NCB.

Blackwater estuarine systems with lower salinity naturally exhibit low pH and DO due to the breakdown of decaying vegetation, which releases tannins (e.g., humic and fulvic acids) that lowers pH, while microbial decomposition consumes oxygen (Flotemersch et al. 2024). Declines in pH and DO at the Chater Point and Blount Island stations may be partially due to decreases in salinity, consistent with findings from Guenther and MacDonald (2012), who reported lower pH and DO at lower salinities in the SJR. Phytoplankton biomass can influence estuarine pH and DO, as CO_2 uptake during photosynthesis raises pH (Carstensen and Duarte 2019, Raven et al. 2020, Shen et al. 2020, Hall et al. 2023), while photosynthetic oxygen production increases DO (Wang et al. 2024). Thus, decreasing Chl-*a*, which is a proxy for phytoplankton biomass, can also explain decreases in pH and DO at the San Jose, Chater Point and Blount Island stations. Changes in pH and DO may also be related to other mechanisms, such as gas exchange, alkalinity, and microbial activity. For example, ocean acidification resulting from increased atmospheric CO_2 and absorption into marine surface waters is thought to be why pH is decreasing across eight Florida estuaries (Robbins and Lisle, 2018).

In contrast to the other three, the ICW station had increasing pH and DO trends, which could be a result of differences in spatial location or sampling period. Both decreases in pH and DO can have negative effects on estuarine biological communities. For example, lower pH can reduce the availability of carbonate ions in the water column, resulting in decreased biocalcification rates and increased shell dissolution of shell forming organisms (Waldbusser et al. 2011a, Waldbusser et al. 2011b). Low DO concentrations can stress aquatic organisms, resulting in fish kills and oyster reef mortality (Landsberg, et al. 2020, Coxe et al. 2023) and can also disrupt biogeochemical processes such as nitrogen cycling and sediment-water fluxes of nutrients and heavy metals (Kemp and Dodds 2002, Skoog and Arias-Esquivel

2009). Regardless of the observed trends mean pH fell within typical estuarine conditions (Ohrel and Register 2006) and all DO concentrations were above the 3.0 mg/L hypoxia stress threshold across the ICWPUs stations. However, there are waterways that flow into the ICW, including Sherman (WBID 2227A) and Hogpen (WBID 2270) Creeks, that are currently impaired for DO and have been placed on FDEP's study list to identify a causative pollutant. Additionally, while this report focuses on DO concentrations that present general concerns for most marine organisms (e.g., oysters), the Lower SJR and its tributaries have Site Specific Alternative Criteria for DO (F.A.C. Rule 62-302.800) to protect more sensitive species such as the Black Creek crayfish (*Procambarus pictus*). This criteria specifies a minimum DO concentration of 4.0 mg/L and uses a total fractional exposure (TFE) metric to assess the duration and extent in which DO concentrations fall between 4.0-5.0 mg/L on an annual basis, with the requirement that the TFE be at or below 1.0.

Nutrient, TSS and Chl-*a* declines observed across the ICWPUs can be explained by ongoing water management practices throughout the system. The SJR is a highly eutrophic system, with nutrient reductions underway through BMAPs targeting TP and TN impairments in the freshwater segment and TN impairment in the marine segment, aimed at reducing harmful algal blooms and meeting the Site Specific Alternative Criteria for DO (F.A.C. Rule 62-302.800). Within the ICWPU 3I there are TMDLs for bacteria (fecal coliforms) within waterways located directly east and west of the ICW; Sherman (WBID 2227), Hopkins (WBID 2266), Greenfield (WBID 2240) and Open Creeks (WBID 2299). Sources of fecal coliform pollution in these waterways include effluent from wastewater treatment facilities, municipal storm sewers, urban runoff, pet waste, and leaking septic tanks, sewer lines and wastewater collection systems. Efforts to reduce fecal coliform pollution from these sources can result in reductions in nutrients and suspended solids entering the system (Jeng et al. 2005, Ohrel and Register 2006). Declines in Chl-*a* are also an expected outcome of nutrient reductions (Lundsør et al. 2020, Codiga et al. 2022) due to their sensitivity to nutrient loading (Doering et al. 2006, Boyer et al. 2009). Compared to the stations further upstream in the SJR, the Blount Island and ICW stations had less Chl-*a* exceedances and very few nutrient exceedances were observed at the ICW station. However, there are impaired WBIDs within Sherman and Greenfield Creeks for TP and Chl-*a*, respectively, suggesting further nutrient reductions should be made to alleviate current impairments.

Overall, analysis of long-term WQ trends showed evidence of decreasing salinity, pH, DO, TSS, nutrients and/or Chl-*a* across the ICWPUs. While salinity, pH, and DO fell within typical ranges reported for estuaries, abrupt decreases in salinity and/or continued declines in pH and DO are important management considerations for oyster populations as well as other marine organisms. Currently more information on WQ is needed in areas such as Clapboard and Sisters Creeks and other smaller tidal waterways, which are the focus of ICWPU 3K, to determine if these areas are experiencing similar changes in WQ to the Blount Island station in the SJR and if these changes are a concern for resident oyster populations (Dix et al. 2019). When analyzing WQ data, differing and/or shorter periods of record (e.g., less than 5 years), low sampling frequency and poor spatial coverage present challenges for statistical analyses (Meals et al. 2011) and resulting interpretation. Currently, the ICWPUs have the fewest WQ monitoring stations out of the eight NCB PUs. Additionally, of the four stations within the ICWPUs, only two are within the NCB boundaries, and only one is located within the focal waterways of the ICWPUs. Expanding WQ monitoring within tidal tributaries across the two PUs can improve statistical power, enhance understanding of spatial and temporal trends, and support more comprehensive management strategies of the ICWPUs waterways. Differences in observed trends among stations may also be due in part to variations in period of record, with some sites having nearly three decades of data and others only about a decade. While the purpose of this report is to analyze WQ trends at individual stations based on all available data, ideally to compare trends across a region, WQ collection years should be standardized across all stations. Additional analyses which standardize the time period across stations can provide information on whether a uniform trend is seen throughout the ICWPUs (and other NCB PUs) or if there are hotspots for specific trends. Finally, given ongoing impairments for fecal coliforms, nutrients, and Chl-*a* in ICWPU 3I, continued efforts to reduce anthropogenic nutrient and bacterial inputs will benefit the coastal system by lowering the risk of eutrophication, harmful algal blooms, and shellfish contamination.

3.4. TOLOMATO RIVER AND MATANZAS RIVER PLANNING UNITS

Summary

- There are six SJRWMD WQ monitoring stations, with two in the Tolomato River Planning Unit (TRPU) (9D) and four in the Matanzas River Planning Unit (MRPU) (9C). While the focus of the NCB is estuarine waters, two freshwater tributary stations in the MRPU, which were established for hydrodynamic modeling, were included to provide additional information on WQ trends throughout this region.

- The tributary stations had the lowest mean salinity, conductivity, pH, and Chl-*a* and highest mean TN, DOC and color. Fresh and polyhaline regions also had lower DO compared to euhaline regions of the estuary. High color and low pH and DO in lower salinity regions are characteristics typical of blackwater estuarine systems.

- Increasing water temperature, driven by warmer weather, is a management concern due to its effects on WQ (e.g., increase Chl-*a*, decrease DO) and the poleward expansion of subtropical aquatic species. Currently there is a need to better understand the effects of long-term warming on estuarine ecosystem function.

- Positive salinity trends may be explained by changes in rainfall and tidal flushing. Increases in pH and DO can be explained by coinciding increases in salinity and Chl-*a*, whereas a decrease in pH may be related to a coinciding increase in water color.

- Decreases in nutrients may be explained by ongoing water management practices or alterations to rainfall and tidal flushing. Although nutrients appear to be decreasing throughout the system, many estuarine regions currently have nutrient and Chl-*a* impairments.

- Increasing Chl-*a* at several locations may be related to coinciding increases in water temperature.

- Continued reductions in nutrient inputs are needed to improve WQ throughout this system. It is unclear how changes in floral and faunal communities, resulting from increased water temperature and eutrophication, may affect ecosystem dynamics. Implementation of species monitoring can aid in tracking changes to biological communities and WQ.

Introduction

The Guana, Tolomato, and Matanzas (GTM) estuary is a bar-bound system which extends from northern St. Johns County past Marineland in Flagler County (Frazel 2009). The natural hydrology of these systems has been altered over the years through the introduction of various water control structures including dikes, drainage ditches, inland wells, and a dam. The Guana River runs parallel to the Tolomato River, with the two joining 9.7 km (6 mi) north of the St. Augustine Inlet and the Matanzas River extends nearly 33.8 km (21 mi) southward from the St. Augustine Inlet (Fig. 3.4.1). Both the Tolomato and Matanzas Rivers are maintained as part of the ICW. River dredging, to maintain them as navigable waterways, has straightened portions of the waterways and created additional spoil islands throughout the system (Frazel 2009). Tributaries into the GTM estuary include the Guana River itself, the San Sebastion River, Moses and Moultrie Creeks, and Pellicer Creek (Fig. 3.4.1, Fig. 3.5.1; Dix et al. 2008, Frazel 2009).

The Tolomato River Planning Unit (TRPU) (9D) has a small portion located in the southeast section of Duval County, with the remainder of the PU extending through St. Johns County (Fig. 3.4.1). The TRPU is bound in the north by the ICWPU 3I, and by the Matanzas River Planning Unit (MRPU) (9C) in the south. The MRPU is located solely within St. Johns County, ending near Crescent Beach, FL (Fig. 3.4.1), and is bound in the south by the PCPU. The Guana Tolomato Matanzas National Estuarine Research Reserve (GTMNERR) was established in 1999 and encompasses approximately 303 km^2 (117 mi^2) of coastal estuarine habitat across the TRPU, MRPU and PCPU. Additionally, the Guana River Marsh Aquatic Preserve, which is the northern section of the GTMNERR, was designated as an OFW. The reserve includes a mix of salt marsh, pinelands, hardwood hammocks, shrub and brushlands. Salt marsh occupies over 20% of the total land cover across the GTMNERR. Dominant species include cordgrass (*Spartina alterniflora*), black needlerush (*Juncus roemerianus*), glasswort (*Salicornia* spp.), saltwort (*Batis maritima*) and mangroves (*Avicennia germinans, Rhizophora mangle, Laguncularia racemosa*). The forests, brushlands, and salt marsh throughout the preserve provides essential habitat for 48 protected animal species including Anastasia Island beach mouse (*Peromyscus polionotus phasma*), gopher tortoise (*Gopherus polyphemus*), wood stork (*Mycteria americana*) and striped newt (*Notopthalmus perstriatus*).

St. Johns County is among Florida's fastest growing counties, with a 43% population increase from 2010 to 2020 (190,039 to 273,425 residents) (United States Census Bureau 2020). Waterways throughout this system include a mixture of Class II and Class III waters, with extensive intertidal oyster reefs through the Tolomato and Matanzas Rivers. There is one conditionally approved shellfish harvest area (92) and five individual shellfish lease areas (SJ-1043, SJ-1154, SJ-240, SJ-1058, SJ-201) throughout the GTM estuary (FDACS 2024) (Fig. 3.4.2).

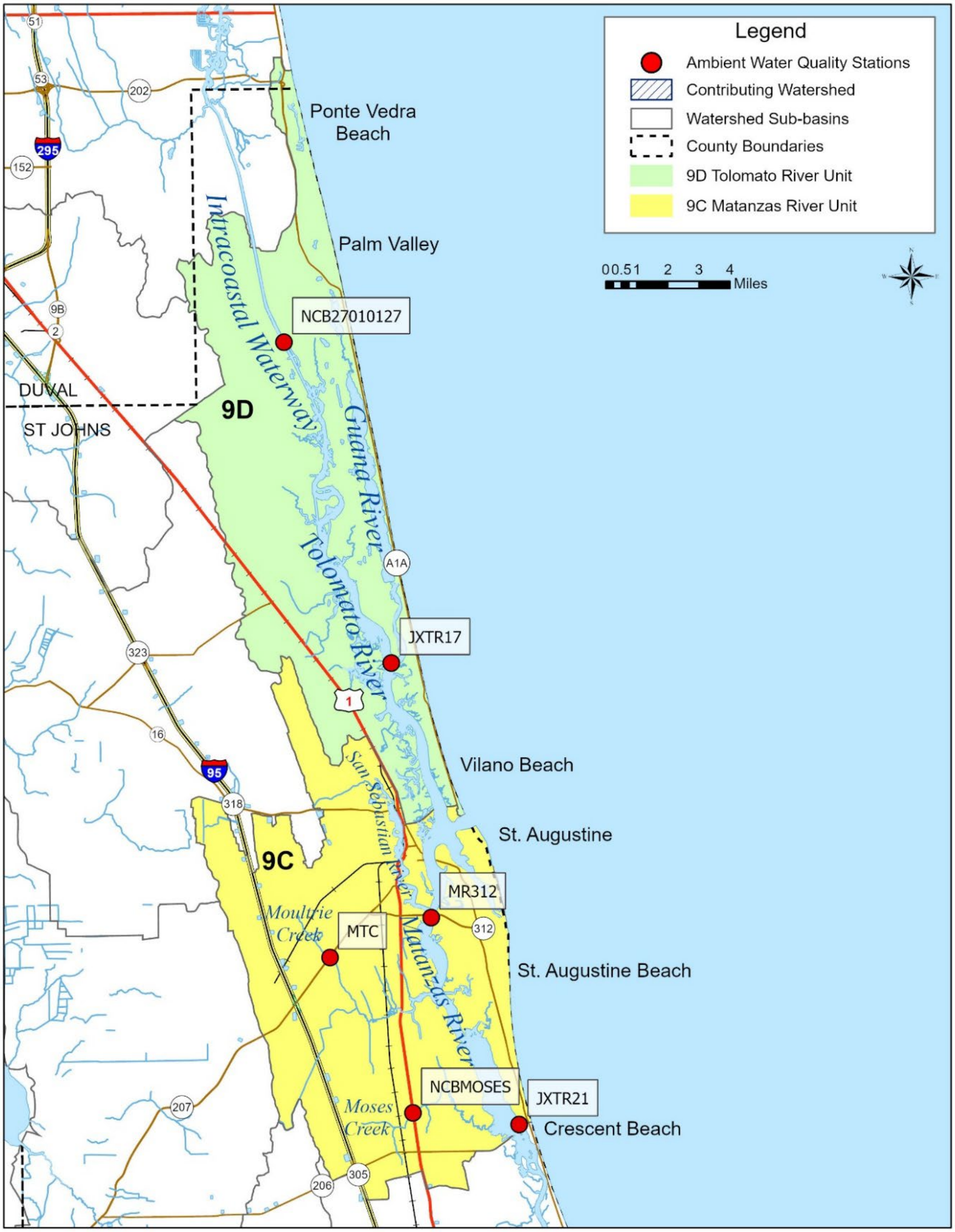

Figure 3.4.1. Map of TRPU 9D (green) and MRPU (yellow). Red circles indicate SJRWMD WQ monitoring stations.

Figure 3.4.2. Map of open and prohibited shellfish harvesting areas within shellfish harvesting area 92 in St. Johns County as of May 1, 2018. Blue squares represent channel markers. Map provided by FDACS.

Methods

For a full description of WQ parameters and analyses performed refer to Chapter 2. While the focus of the TRPU and MRPUs are estuarine waters, freshwater tributary stations in Moultrie (MTC) and Moses Creeks (NCBMOSES) were incorporated into these PUs for hydrodynamic modeling (e.g., salinity, nutrient loading). Tributary stations are therefore included in this report to provide additional context about WQ trends across these PUs.

Both the Matanzas River County Road (CR) 312 station (MR312) and Moultrie Creek station (MTC) had systematic differences in sampling frequency. At the Matanzas River station, data was collected bimonthly from 1991-1996 and monthly from 1997-2023. Since the majority of the data at this station was collected monthly, bimonthly data was removed from the analyses.

At Moultrie Creek, data were collected bimonthly from 1984-2004 and monthly from 2005-2023. Because the majority of the dataset is bimonthly, values were aggregated into bimonthly bins for most parameters. Secchi, NOx-D and DOC were not aggregated into bimonthly bins, as only monthly data were available for these parameters. Due to limited sampling frequency and large data gaps, salinity was analyzed from 2002-2023 and Secchi and NOx-D from 2005-2023. Additionally, sampling for DOC did not begin until 2018 for this location.

Results

Mean Water Quality Conditions

There are two SJRWMD WQ monitoring stations in the TRPU and four in the MRPU (Fig. 3.4.1). Of the six stations, Moultrie Creek (MTC) in the MRPU is the oldest with monitoring beginning in 1984, while the ICW station (NCB27010127) in the TRPU is the most recent, with monitoring starting in 2014 (Table 3.4.1). Data collection began in the 1990s for the Tolomato River (JXTR17), Matanzas River CR 315 (MR312) and Crescent Beach (JXTR21) stations and in 2008 for Moses Creek (NCBMOSES) (Table 3.4.1).

Mean WQ parameters were similar across all stations for water temperature, TP, and NOx-D (Table 3.4.1, Table 3.4.2). Moultrie and Moses Creeks had lower salinity (freshwater), pH, DO and Chl-*a* and higher TN, DOC and color compared to the other four stations (Table 3.4.1, Table 3.4.2). By contrast, salinities at the ICW, Tolomato River, and two Matanzas River stations were polyhaline or euhaline and Secchi and TSS were highest at the Tolomato River and two Matanzas River stations (Table 3.4.1, Table 3.4.2).

Table 3.4.1. Mean (±) standard deviation of physical WQ parameters for stations within the TRPU and MRPU.

Station	Years	Temp. (°C)	Sal.	Cond. (μmhos/cm)	pH	DO (mg/L)	Secchi (m)
NCB27010127	2014 - 2023	23.98 ± 5.91	20.85 ± 7.71	33105.96 ± 11189.70	7.35 ± 0.18	5.75 ± 1.43	0.77 ± 0.27
JXTR17	1997 - 2023	22.60 ± 5.60	31.57 ± 4.41	48352.77 ± 6143.90	7.72 ± 0.33	6.37 ± 1.28	0.98 ± 0.44
MTC	1984 - 2023	20.29 ± 4.75	0.16 ± 0.06	313.95 ± 116.24	6.81 ± 0.54	5.47 ± 1.84	0.51 ± 0.18
NCBMOSES	2008 - 2023	20.20 ± 5.09	0.18 ± 0.08	382.37 ± 161.97	6.83 ± 0.38	5.36 ± 1.57	0.36 ± 0.14
MR312	1997 - 2023	22.53 ± 5.32	33.38 ± 2.59	50830.81 ± 3546.77	7.89 ± 0.21	6.56 ± 1.21	1.06 ± 0.39
JXTR21	1997 - 2023	22.73 ± 5.39	33.08 ± 2.92	50472.67 ± 3886.51	7.81 ± 0.21	6.30 ± 1.25	1.05 ± 0.43

Table 3.4.2. Mean (±) standard deviation of chemical WQ parameters for stations within the TRPU and MRPU.

Station	Years	TP (mg/L)	TN (mg/L)	NOx-D (mg/L)	DOC (mg/L)	Color (PCU)	TSS (mg/L)	Chl-*a* (mg/m^3)
NCB27010127	2014 - 2023	0.12 ± 0.04	0.79 ± 0.19	0.08 ± 0.05	11.81 ± 4.44	77.85 ± 56.05	12.30 ± 8.90	9.20 ± 6.17
JXTR17	1997 - 2023	0.09 ± 0.07	0.49 ± 0.20	0.03 ± 0.03	5.08 ± 3.24	30.16 ± 31.65	25.51 ± 17.33	6.56 ± 6.02
MTC	1984 - 2023	0.09 ± 0.06	1.18 ± 0.46	0.07 ± 0.04	36.28 ± 12.30	349.93 ± 251.88	8.12 ± 18.26	2.43 ± 5.21
NCBMOSES	2008 - 2023	0.12 ± 0.09	1.02 ± 0.41	0.04 ± 0.02	39.62 ± 17.39	286.98 ± 224.97	5.87 ± 6.96	2.32 ± 1.98
MR312	1997 - 2023	0.09 ± 0.08	0.43 ± 0.18	0.04 ± 0.02	3.37 ± 1.57	22.09 ± 29.52	27.10 ± 17.84	5.26 ± 2.47
JXTR21	1997 - 2023	0.09 ± 0.08	0.42 ± 0.18	0.03 ± 0.02	3.81 ± 1.84	25.53 ± 27.17	23.77 ± 17.36	4.03 ± 2.25

Table 3.4.3. Summary statistics for physical and chemical parameters at WQ monitoring stations in the TRPU and MRPU.

	Temp. (°C)	**Sal.**	**Cond. (µmhos/cm)**	**pH**	**DO (mg/L)**	**Secchi (m)**	TP (mg/L)	TN (mg/L)	NOx-D (mg/L)	DOC (mg/L)	Color (PCU)	TSS (mg/L)	Chl-*a* (mg/m^3)
	Station NCB27010127 (ICW)												
KW (p)	**< 0.0001**	**0.0006**	**0.0006**	**< 0.0001**	**< 0.0001**	0.1005	**0.0060**	**0.0132**	**0.0006**	0.0833	0.0917	**0.0197**	**< 0.0001**
Kendall tau	0.025	-0.132	-0.132	0.083	0.173	0.024	-0.190	-0.073	-0.115	0.060	0.020	0.075	0.102
Kendall (p)	0.7740	0.0909	0.0909	0.3695	**0.0326**	0.7243	**0.0230**	0.4639	0.2884	0.3909	0.7728	0.3518	0.2074
	Station JXTR17 (Tolomato River)												
KW (p)	**< 0.0001**	**0.0016**	**< 0.0001**	**< 0.0001**	**< 0.0001**	**< 0.0001**	0.0821	**< 0.0001**	**0.0196**	0.4752	0.2755	**0.0120**	**< 0.0001**
Kendall tau	0.115	0.029	0.032	0.183	0.005	0.044	-0.104	-0.161	-0.058	0.146	-0.279	-0.233	-0.054
Kendall (p)	**0.0040**	0.4745	0.4304	**< 0.0001**	0.9233	0.2875	**0.0078**	**0.0001**	0.4052	**0.0153**	**< 0.0001**	**< 0.0001**	0.1861
	Station MTC (Moultrie Creek)												
KW (p)	**< 0.0001**	**0.0319**	**0.0023**	0.1024	**< 0.0001**	**<0.0001**	**< 0.0001**	**0.0012**	0.5126	0.7510	**0.0002**	0.5005	**< 0.0001**
Kendall tau	0.138	0.323	0.286	0.415	0.295	0.233	-0.182	-0.069	-0.075	0.102	-0.077	0.264	0.271
Kendall (p)	**0.0040**	**< 0.0001**	**< 0.0001**	**< 0.0001**	**< 0.0001**	**0.0018**	**0.0002**	0.1276	0.4557	0.3090	0.1510	**< 0.0001**	**< 0.0001**
	Station NCBMOSES (Moses Creek)												
KW (p)	**< 0.0001**	0.3001	0.3137	0.6348	**0.0009**	0.6472	**0.0409**	0.5219	**0.0042**	0.2066	0.5649	**0.0327**	0.6482
Kendall tau	0.158	-0.056	-0.055	-0.154	0.276	-0.102	-0.117	0.239	0.172	0.006	0.204	-0.008	0.008
Kendall (p)	**0.0227**	0.3756	0.3858	**0.0154**	**0.0003**	0.2385	0.1234	**0.0002**	0.0631	0.9459	**0.0018**	1.0000	0.9392
	Station MR312 (Matanzas River CR312)												
KW (p)	**< 0.0001**	**< 0.0001**	**< 0.0001**	**< 0.0001**	**< 0.0001**	**0.0003**	**0.0295**	**0.0005**	0.1753	0.1278	0.5099	0.4702	**< 0.0001**
Kendall tau	0.123	0.153	0.163	0.152	0.023	-0.070	-0.110	-0.240	-0.040	0.112	-0.345	-0.295	-0.033
Kendall (p)	**0.0021**	**0.0001**	**< 0.0001**	**0.0001**	0.5966	0.0892	**0.0104**	**< 0.0001**	0.4469	0.0605	**< 0.0001**	**< 0.0001**	0.4041
	Station JXTR21 (Matanzas River Crescent Beach)												
KW (p)	**< 0.0001**	**0.0005**	**0.0002**	**< 0.0001**	**< 0.0001**	**< 0.0001**	**0.0029**	**< 0.0001**	0.3650	0.1829	0.1207	**0.0013**	**< 0.0001**
Kendall tau	0.096	0.140	0.146	0.124	-0.015	0.034	-	-0.263	-0.017	0.158	-0.310	-0.324	-
Kendall (p)	**0.0147**	**0.0005**	**0.0003**	**0.0021**	0.6881	0.3794	-	**< 0.0001**	0.7582	**0.0079**	**< 0.0001**	**< 0.0001**	-

If $p < 0.05$ for the Kruskal–Wallis (KW) test, a Seasonal Kendall test was performed. If $p \geq 0.05$ for the KW test, a Mann-Kendall test was performed. Significant test results ($p < 0.05$) are shown in bold.

Monthly Variability

Results from the KW tests showed monthly variability for five or more WQ parameters at each station across the two PUs, with all stations having monthly differences for water temperature and DO (Table 3.4.3). The only station to not have a p-value < 0.05 for TP was the Tolomato River and the only station to not have significant monthly differences for salinity, conductivity, TN and Chl-*a* was Moses Creek. Additionally, differences in color across months was only found for Moultrie Creek.

Station Water Quality Trends and Criteria

Investigation of physical and chemical WQ trends in the TRPU revealed a positive trend in DO and a negative trend in TP at the **ICW station (NCB27010127)** (Table 3.4.3). From 2014 to 2023, water temperature remained around 25°C and salinity decreased from 22.5 to 19, although this change was not significant ($p > 0.05$) (Fig. 3.4.3A-B). The trendline and individual DO concentrations were above the 3.0 mg/L hypoxia threshold (Fig. 3.4.3C). Concentrations for TP decreased from 0.13 mg/L in 2014 to 0.10 mg/L in 2023, with the trendline recently (2021) falling below the nutrient criteria value (Fig. 3.4.3D). While results for TN and Chl-*a* were not significant, trends exceeded the nutrient criteria value across all years for TN and from 2018 to 2023 for Chl-*a* (Fig. 3.4.3E-F).

For the **Tolomato River station (JXTR17)**, positive trends in water temperature, pH, and DOC and negative trends in TP, TN, color, and TSS were found (Table 3.4.3). Water temperature increased from 22.5°C in 1997 to 23.2°C in 2023, while salinity remained near 32.5 across all years (Fig. 3.4.4A-B). No change in DO was found, with most concentrations and the trendline being above 3.0 mg/L (Fig. 3.4.4C). Over the 26-year period of record, TP decreased from 0.09 mg/L to 0.07 mg/L and TN decreased from 0.52 mg/L to 0.43 mg/L, with individual TP and TN concentrations occasionally exceeding their respective numeric nutrient criteria values (Fig. 3.4.4D-E). Additionally, many Chl-*a* concentrations exceeded the nutrient criteria value of 6.6 mg/m^3 (Fig. 3.4.4F).

In the MRPU, increasing trends were found for all six physical WQ parameters at the **Moultrie Creek station (MTC)** as well as increases in TSS and Chl-*a* and a decrease in TP (Table 3.4.3). Water temperature increased from 20.6°C in 1984 to 22.0°C in 2023, and salinity increased from 0.12 in 2002 to 0.20 in 2023 (Fig. 3.4.5A-B). Most DO concentrations and the trendline were above 3.0 mg/L (Fig. 3.4.5C). For TP, concentrations decreased across 39 years (1984-2023) from 0.09 mg/L to 0.06 mg/L, with individual values periodically exceeding the nutrient criteria (Fig. 3.4.5D). While TN was stable across all years, concentrations also periodically exceeded the nutrient criteria value (Fig. 3.4.5E). For Chl-*a,* concentrations increased by 1 mg/m^3 (0.5 mg/m^3 to 1.5 mg/m^3) from 1997 to 2023 (Fig. 3.4.5F).

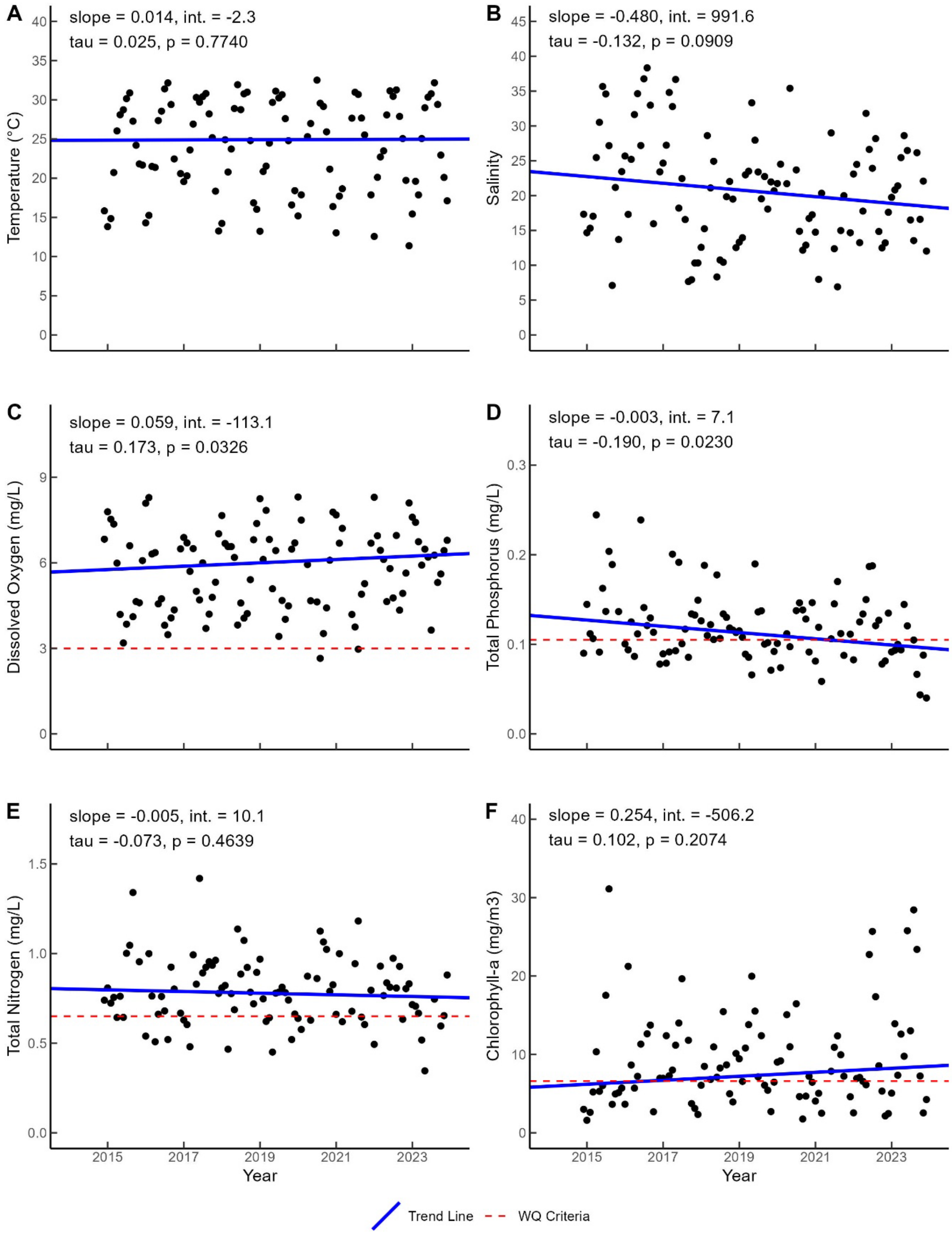

Figure 3.4.3. Seasonal Kendall trends for temperature, salinity, DO, TP, TN, and Chl-*a* (A-F) at the ICW station (NCB27010127). The solid blue line depicts the monotonic trend for each parameter, with $p < 0.05$ denoting significant trends. The dashed red line indicates WQ criteria based on the 3.0 mg/L hypoxia stress threshold for DO and annual geometric mean values for TP, TN, and Chl-*a* outlined in F.A.C. Rule 62-302.532.

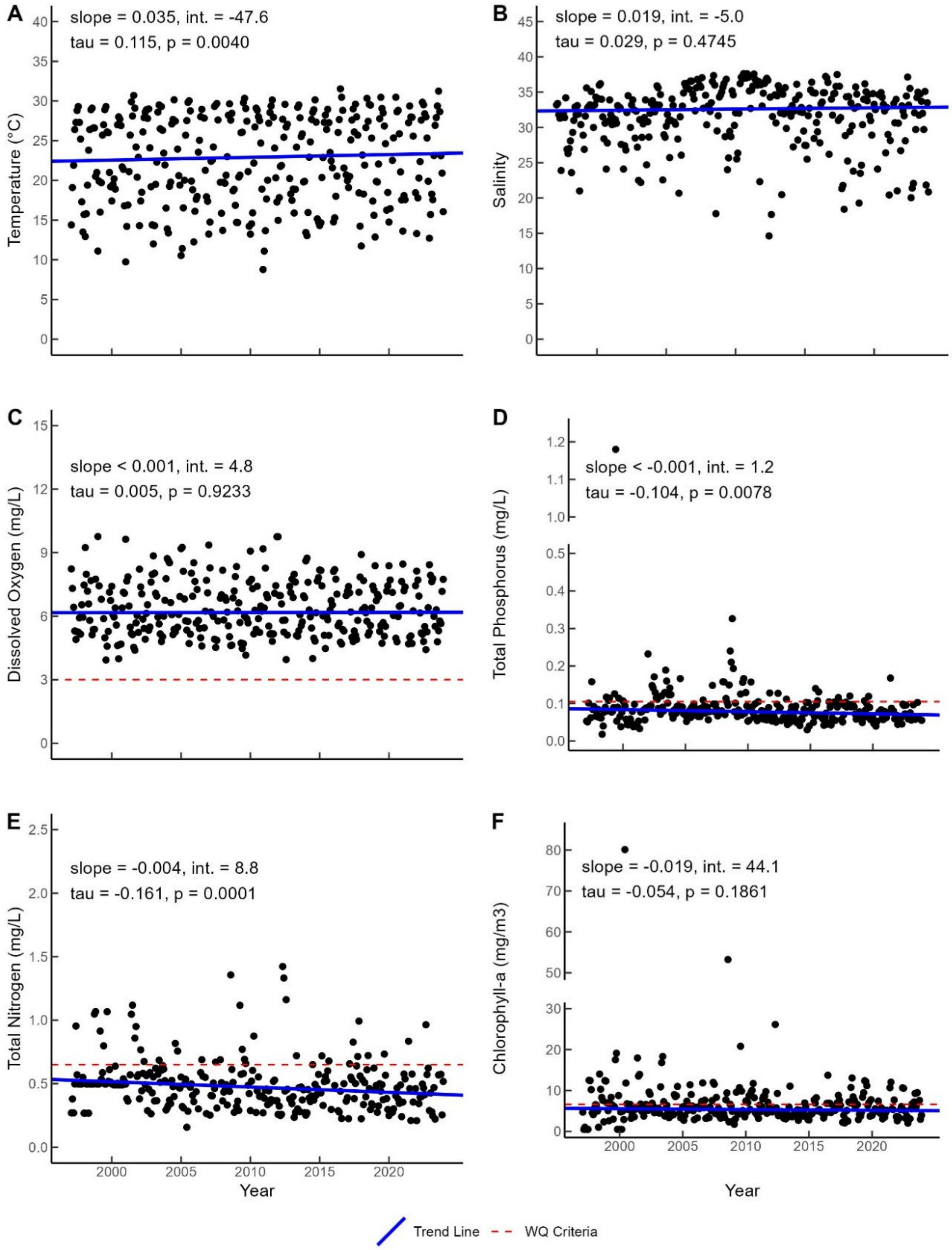

Figure 3.4.4. Trends for temperature, salinity, DO, TP, TN, and Chl-*a* (A-F) at the Tolomato River station (JXTR17). Seasonal Kendall = temperature, salinity, DO, TN, and Chl-*a*; Non-seasonal Kendall = TP. The solid blue line depicts the monotonic trend for each parameter, with $p < 0.05$ denoting significant trends. The dashed red line indicates WQ criteria based on the 3.0 mg/L hypoxia stress threshold for DO and annual geometric mean values for TP, TN, and Chl-*a* outlined in F.A.C. Rule 62-302.532.

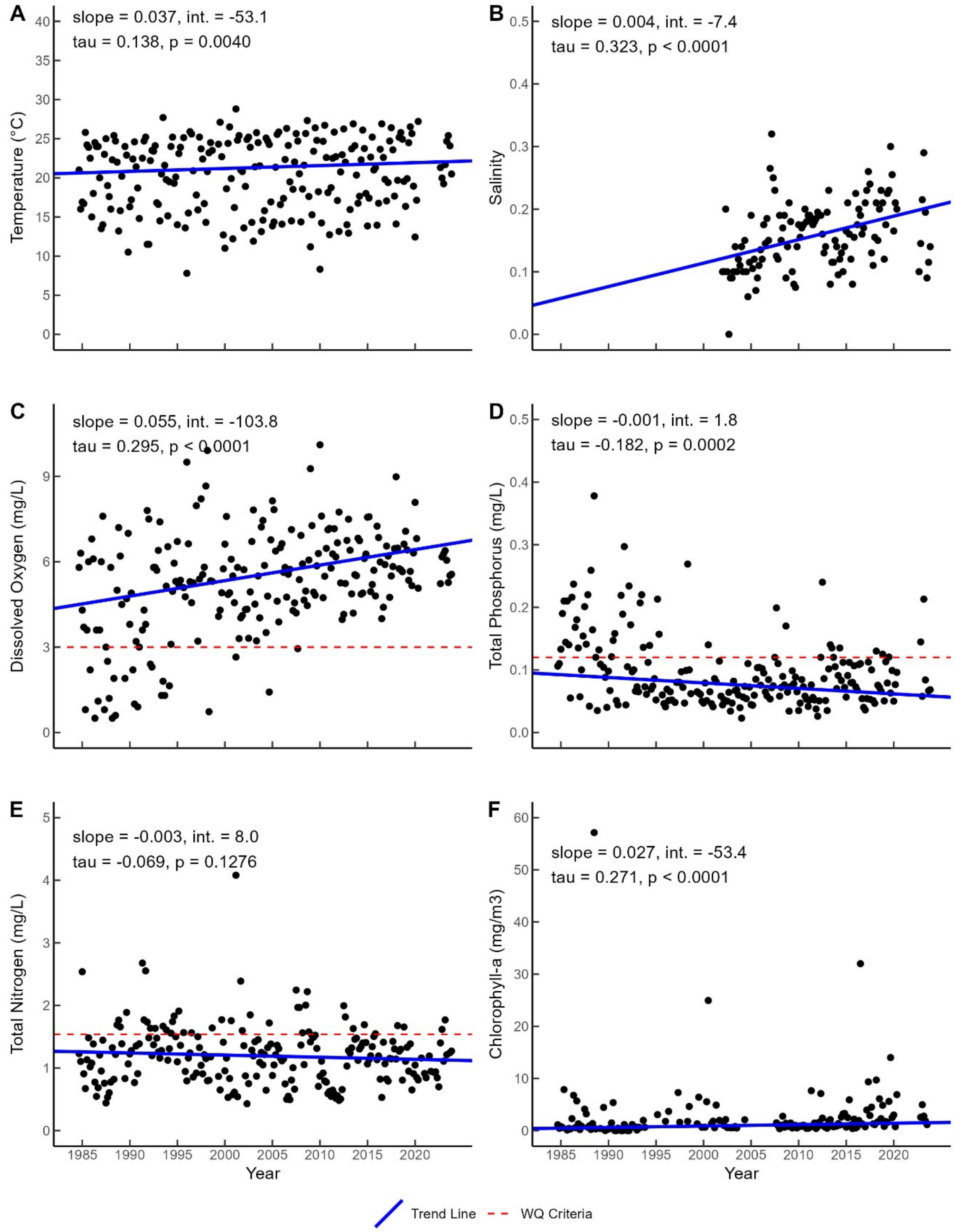

Figure 3.4.5. Seasonal Kendall trends for temperature, salinity, DO, TP, TN, and Chl-*a* (A-F) at the Moultrie Creek station (MTC). The solid blue line depicts the monotonic trend for each parameter, with $p < 0.05$ denoting significant trends. The dashed red line indicates WQ criteria based on the 3.0 mg/L hypoxia stress threshold for DO and annual geometric mean values for TP and TN outlined in F.A.C. 62-302.531(2)(c). Streams do not have Chl-*a* criteria values.

For the **Moses Creek station (NCBMOSES)**, increases in water temperature, DO, TN, and color and a decrease in pH were found (Table 3.4.3). From 2008 to 2023, water temperature increased by 1°C (20.0°C to 21.0°C) while salinity remained around 0.17 (Fig. 3.4.6A-B). For DO, the trendline and most values were above 3.0 mg/L (Fig. 3.4.6C). Over the 15 years no change in TP was found, whereas TN increased from 0.74 mg/L to 1.20 mg/L, with individual TP and TN concentrations periodically exceeding their respective nutrient criteria values (Fig. 3.4.6D-E). No change in Chl-*a* was found across years, with most concentrations being below 5 mg/m^3 (Fig. 3.4.6F).

At the **Matanzas River CR 312 station (MR312)**, there were increases in water temperature, salinity, conductivity and pH and decreases in TP, TN, color, and TSS (Table 3.4.3). Over 26 years (1997-2023), water temperature increased by 1°C (22.1°C to 23.1°C) and by 1.5 (33.0 to 34.5) for salinity (Fig. 3.4.7A-B). The trendline and all individual DO concentrations were above 3.0 mg/L (Fig. 3.4.7C). From 1997 to 2023, TP decreased from 0.08 mg/L to 0.06 mg/L, and TN decreased from 0.56 mg/L to 0.32 mg/L, with individual TP and TN concentrations occasionally exceeding their respective numeric nutrient criteria values (Fig. 3.4.7D-E). No change in Chl-*a* was found but the trendline was above the nutrient criteria value (4.0 mg/m^3) across all years (Fig. 3.4.7F).

Positive trends for water temperature, salinity, conductivity, pH, and DOC and negative trends for TN, color, and TSS were found at the **Matanzas River Crescent Beach station (JXTR21)** (Table 3.4.3). From 1997 to 2023 water temperature increased by 0.8°C (22.8°C to 23.6°C) and salinity increased by 1.4 (32.8 to 34.2) (Fig. 3.4.8A-B). Almost all individual concentrations and the trendline were above 3.0 mg/L (Fig. 3.4.8C). Significant heterogeneity was found for TP (Chi-Square Heterogeneity Test, $p = 0.0482$) with negative trends for all months except May (Table 3.4.4). All monthly TP trendlines were below the nutrient criteria value, but individual concentrations periodically exceeded this value (Fig. 3.4.8D). Over the 26-year period, TN decreased from 0.45 mg/L to 0.30 mg/L, with individual concentrations occasionally exceeding the 0.55 mg/L nutrient criteria value (Fig. 3.4.7E). Significant heterogeneity was also found for Chl-*a* ($p = 0.0238$), with trends being negative for all months except May, June, July and December (Table 3.4.4). Many Chl-*a* concentrations exceeded the nutrient criteria value (4.0 mg/m^3) at this location, with trendlines for warmer months (e.g., summer to early fall) being above the criteria value (Fig. 3.4.8D).

Table 3.4.4. Results for individual monthly Mann-Kendall tests run for TP (mg/L) and Chl-*a* (mg/m^3) at the Matanzas River station (JXTR21) near Crescent Beach.

	TP (mg/L)		Chl-*a* (mg/m^3)	
Month	**tau**	**p-value**	**tau**	**p-value**
January	-0.464	**0.0016**	-0.117	0.4635
February	-0.186	0.2244	-0.080	0.5912
March	-0.273	0.0585	-0.034	0.8255
April	-0.093	0.5283	-0.179	0.1962
May	0.098	0.5189	0.243	0.0856
June	-0.145	0.3106	0.120	0.4137
July	-0.075	0.6345	0.048	0.7387
August	-0.486	**0.0005**	-0.231	0.1029
September	-0.022	0.9013	-0.240	0.0896
October	-0.030	0.8656	-0.248	0.1235
November	-0.271	0.0551	-0.477	**0.0007**
December	-0.505	**0.0021**	0.107	0.4923

Significant p-values ($p < 0.05$) are shown in bold.

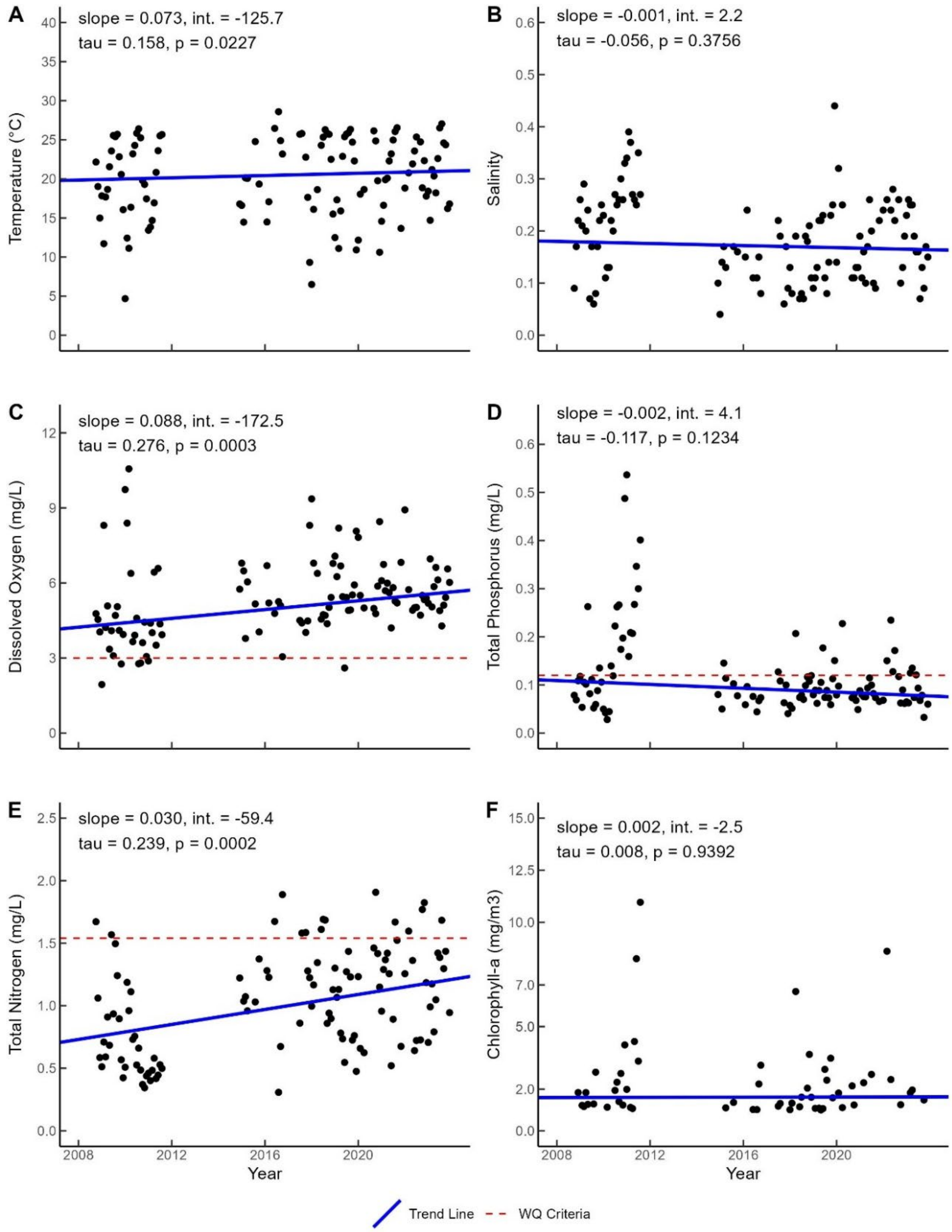

Figure 3.4.6. Trends for temperature, salinity, DO, TP, TN, and Chl-*a* (A-F) at the Moses Creek station (NCBMOSES). Seasonal Kendall = temperature, DO, and TP; Non-seasonal Kendall = salinity, TP, and Chl-*a*. The solid blue line depicts the monotonic trend for each parameter, with $p < 0.05$ denoting significant trends. The dashed red line indicates WQ criteria based on the 3.0 mg/L hypoxia stress threshold for DO and annual geometric mean values for TP, TN, and Chl-*a* outlined in F.A.C. Rule 62-302.532.

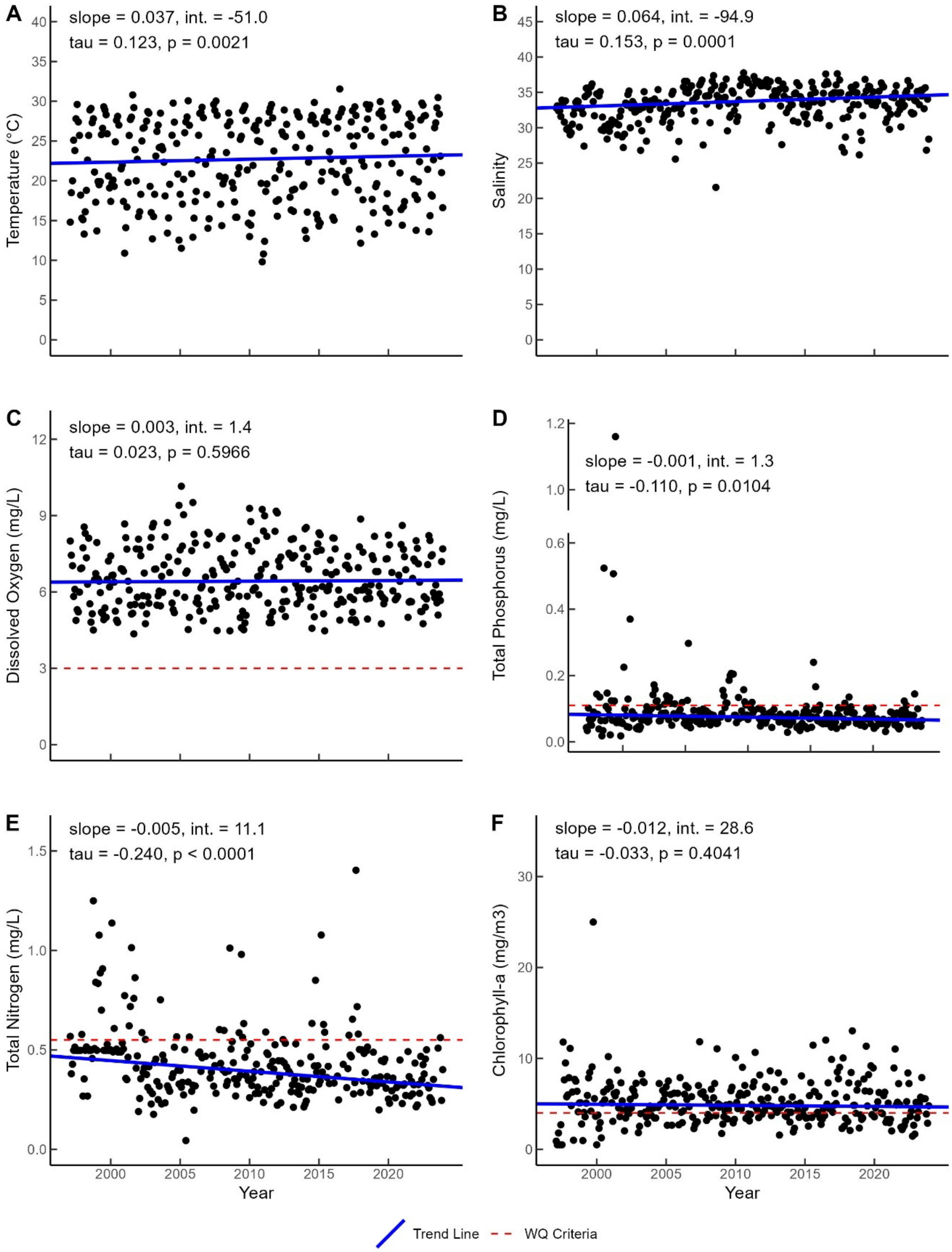

Figure 3.4.7. Seasonal Kendall trends for temperature, salinity, DO, TP, TN, and Chl-*a* (A-F) at the Matanzas River station (MR312). The solid blue line depicts the monotonic trend for each parameter, with $p < 0.05$ denoting significant trends. The dashed red line indicates WQ criteria based on the 3.0 mg/L hypoxia stress threshold for DO and annual geometric mean values for TP and TN outlined in F.A.C. 62-302.531(2)(c). Streams do not have Chl-*a* criteria values.

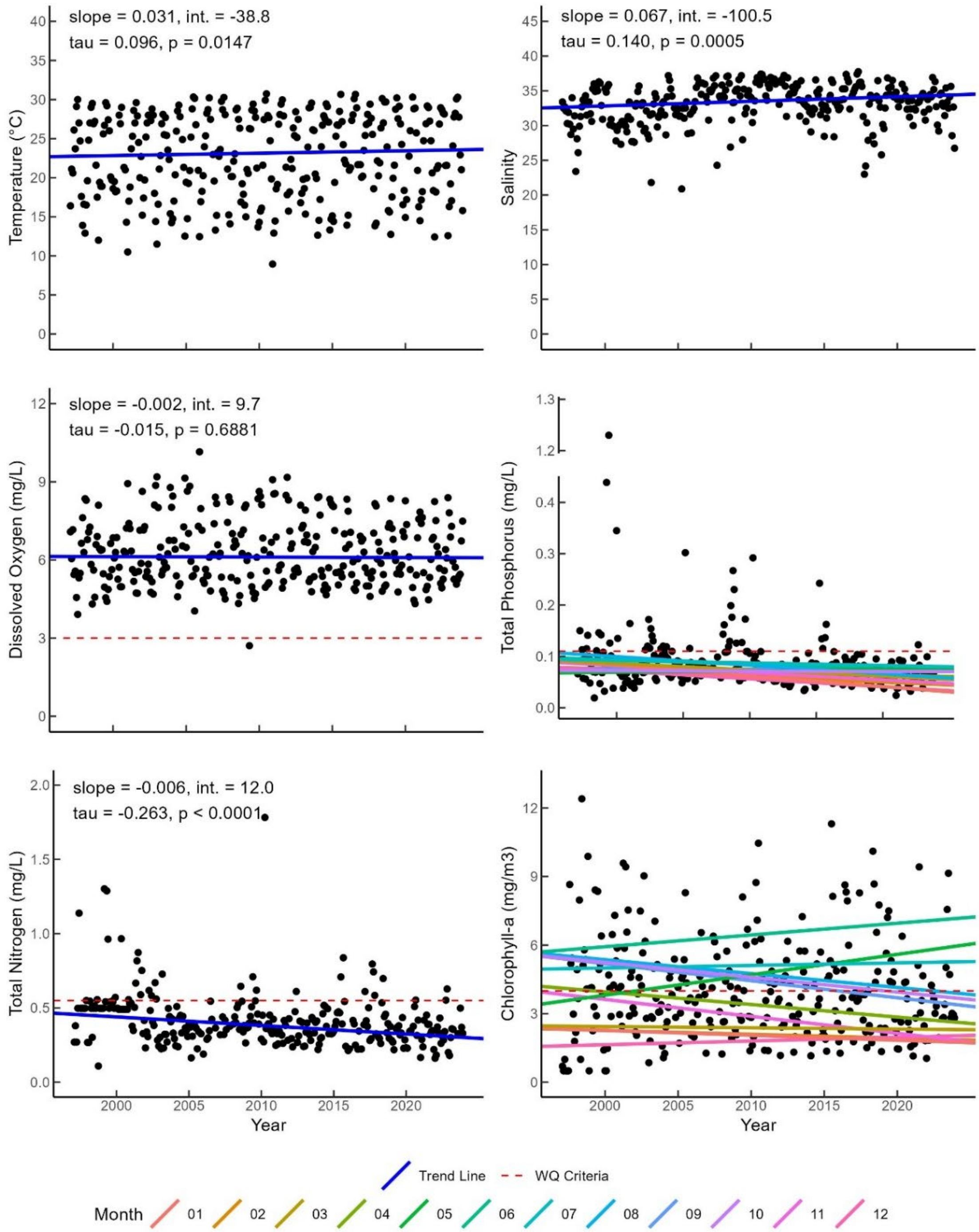

Figure 3.4.8. Trends for temperature, salinity, DO, TP, TN, and Chl-*a* (A-F) at the Matanzas River station (JXTR21) near Crescent Beach. Seasonal Kendall = temperature, salinity, DO, and TN; Non-seasonal Kendall = TP and Chl-*a*. For months 01 = January, 02 = February, etc. The solid blue line depicts the monotonic trend for each parameter, with $p < 0.05$ denoting significant trends. P-values for TP and Chl-*a* are listed in Table 3.4.4. The dashed red line indicates WQ criteria based on the 3.0 mg/L hypoxia stress threshold for DO and annual geometric mean values for TP, TN, and Chl-*a* outlined in F.A.C. Rule 62-302.532.

Discussion

Previous studies have linked differences in physiochemical WQ properties across the GTM estuary and tributaries to proximity to freshwater or inlet (seawater) sources (Chaya et al. 2023), a pattern further supported by this report. Of the six monitoring stations, the two tributary stations located in Moultrie and Moses Creeks had the lowest mean salinity, pH, and Chl-*a* and highest mean TN, DOC, and color. Stations with lower salinity also had lower DO compared to locations with euhaline (marine) salinities. Observations of low pH and DO, and high DOC and color, within the low salinity areas of the GTM estuary align with conditions reported for blackwater estuarine systems (Gallegos 2005, Blair et al. 2009, Chen et al. 2015). High variability of color and DOC at the Moultrie and Moses Creek stations can be explained by fluctuations in rainfall and freshwater inflow. Increased freshwater inflow and precipitation runoff can increase turbidity, dissolved organic matter (DOM), color, nutrient inputs, and trace elements (e.g., metals) within coastal estuaries (Durako et al. 2010, Sankar et al. 2019). Higher TN at freshwater stations aligns with findings from Hart el at. (2015), which reported a negative correlation between nutrients and salinity in the GTM estuary. Lower nutrient concentrations in higher salinity regions of estuaries, compared to freshwater regions, are partly attributed to seawater dilution and the masking effects of tidal cycles and flushing (Coffin and Hampson 1992, Chaya et al. 2023).

Weak (tau < 0.3) to moderate ($0.3 \leq$ tau < 0.5) trends were found for two or more WQ parameters at each station. Parameters such as water temperature were expected to have gradual (weak) changes over time. Globally, estuarine waters are warming at an average rate of 0.070 ± 0.004°C per year, closely linked to rising air temperatures (Prum et al. 2024). Increasing water temperatures in the GTM estuary were supported by other studies conducted across Florida (Shi et al. 2022, Powell 2024, Shi et al. 2024), which report rises in air and estuarine water temperatures in recent decades.

Notably, 2022 and 2023 were among the state's warmest years, with south Florida estuaries showing clear warming responses (Shi et al. 2024). Between 2003 and 2023, water temperatures in Florida Bay, Caloosahatchee, and Tampa Bay rose at an accelerated warming rate of 0.70°C/decade – up from 0.55°C/decade reported for 2000 to 2021 (Shi et al. 2022). Changes to water temperature pose a management concern because temperature has a key role in regulating physiochemical and biological processes within estuarine systems (Mallick and Dunn 2024, Prum et al. 2024). Warm water temperatures increase metabolic rates which can elevate biological oxygen demand, microbial activity, decomposition rates, and primary productivity, and enhance the risk of hypoxia and the occurrence of harmful algal blooms (Nydahl et al. 2013, Wetz and Yoskowitz 2013). Furthermore, oxygen solubility is lower, and the spread of pathogens is higher under warmer water conditions (Petes et al. 2012, Chapra et al. 2021).

In addition to its effects on WQ, increasing water temperature, especially during the winter, can promote the poleward expansion of subtropical marine taxa. Shifts in the latitudinal range of plankton, seagrass, invertebrates, fish and marine mammals have been reported around the globe (Hastings et al. 2020). The TRPU and MRPUs overlap with the tropical-temperate transition zone, which is located between the Indian River Lagoon and the St. Augustine Inlet (28° N–30° N) (Kimball and Eash-Loucks 2021, Osland et al. 2021, Brewton and Lapointe 2023). Changes in biological community composition are likely to occur within these PUs with continued increases in water temperature. Milder winter water temperatures across the tropical-temperate transition zone have already been associated with the poleward expansion of the western Atlantic sea bream (*Archosargus rhomboidalis*) in the Indian River Lagoon (Adams et al. 2024) and common snook (*Centropomus undecimalis*) in Cedar Key (Purtlebaugh et al. 2020). The northward expansion of subtropical species raises concerns over changes in fish community abundance, biodiversity, and trophic interactions within these systems. Currently, there is a need for research that investigates and monitors long-term changes to estuarine ecosystems in response to warming water temperatures (Kurylyk and Smith 2023, Itsukushima et al 2024, Prum et al. 2024).

Salinity within the GTM estuary is sensitive to precipitation and tidal water exchange (Dix et al. 2008, Hart et al. 2015), with MRPU positive salinity trends likely related to decreases in rainfall or increases in tidal inflow. Gradual salinity increases, such as the 1.4-1.5 in the Matanzas River and 0.08 in Moultrie Creek over the past 26 to 39 years, are unlikely to have direct negative effects on aquatic species. Rather, abrupt or prolonged changes to salinity over time are more of a management concern for this system. The GTM estuary has previously had prolonged decreases in salinity resulting from high rainfall associated with Hurricanes Charley, Frances, Ivan, and Jeanne in 2004 (Sheng et al. 2008) and Hurricane Irma in 2017 (Kimmel et al. 2024). Extended periods of low salinity can negatively impact oyster recruitment, survival and growth (La Peyre et al. 2013, Parker et al. 2013), decrease oyster reef faunal community abundance, biomass and diversity (Tolley et al. 2005, Marshall et al. 2019), and have adverse effects on the survival and development of fish (Bachman and Rand, 2008). Under normal conditions, the GTM is considered a high salinity, ocean dominated estuary (Kimball and Eash-Loucks 2021). While polyhaline to euhaline salinities are expected throughout the Tolomato and Matanzas Rivers, continued long-term increases in salinity could be a potential concern when combined with increasing water temperature. A previous study found that high salinity (e.g., 33) combined with warm water temperatures (e.g., 25°C) increased the infection intensity of *Perkinsus marinus* in oysters, leading to higher disease related mortality (Petes et al. 2012).

Changes to pH and DO were found at several stations across the two PUs. At some stations, increases in pH and DO may be linked to increasing salinity and/or Chl-*a*. This is supported by studies reporting positive correlations between salinity and both pH and DO in the GTM estuary (Chaya et al. 2023), as well as between phytoplankton biomass and elevated pH (Carstensen and Duarte 2019, Raven et al. 2020, Shen et al. 2020, Hall et al. 2023) and surface DO (Wang et al. 2024). The observed decrease in pH at one location may be explained by the coinciding increase in water color. Water color reflects the presence of DOM, and compounds such as humic and fulvic acids from DOM can lower pH (Cai et al. 1998, Bowers et al. 2004). Although the mechanism(s) driving changes in pH and DO were not clear for all stations, they could be related to other processes such as stratification and mixing, residence time, nutrient loading, CO_2 concentrations, biological respiration, and decomposition (McCabe et al. 2021, Hall et al. 2023, Testa et al. 2024). In general, increases in pH and DO are beneficial. Higher pH increases carbonate ion availability for shell formation for oysters and other organisms (Waldbusser et al. 2011a, Waldbusser et al. 2011b), while elevated DO supports the overall aquatic animal health.

Declines in nutrients were found throughout the TRPU and MRPU which can be explained by ongoing water management practices and/or changes in rainfall or tidal flushing. Pollution sources that introduce nutrients into the GTM estuary include wastewater facilities, agriculture (e.g. animal waste, fertilizer), urban and stormwater runoff, leaking septic tanks, and wildlife (Frazel 2009). Improvements to water management practices such as converting septic systems to municipal sewer, repairing and/or upgrading conventional septic tanks to advanced treatment systems, upgrading wastewater treatment facilities to advanced treatment practices, and retrofitting existing stormwater systems to modern requirements may lower nutrient inputs to waterways. For example, both St. Johns County and the City of St. Augustine have implemented Florida-Friendly Fertilizer Use ordinances to reduce nutrient inputs into waterways and St. Johns County has the goal of reducing 100,000 pounds of nutrients discharged into the Matanzas River through effluent by 2032 (Viti 2024). Additionally, decreases in rainfall and increases in tidal flushing can reduce nutrient concentrations within the system (Hart et al. 2015, Chaya et al. 2023). However, changes in data distribution and MDLs may have also contributed to nutrient declines at some stations. At Moses Creek, TP concentrations were notably higher from 2010-2011 compared to following years, although the underlying reason for elevated concentrations remains unclear. Also, the two Matanzas River stations had higher MDLs for TN in the late 1990s to early 2000s relative to more recent years. While the Kendall trend test is robust to outliers, notable changes in data variability caused by exogenous factors such as evolving MDLs can reduce the statistical power of the test to detect true trends in WQ parameters (Meals et al. 2011, Helsel et al. 2020). The only location across the two PU's that had a nutrient (TN) increase was Moses Creek, which flows into the Matanzas River near Crescent Beach. While the exact mechanism behind this increase is unknown, higher tidal flushing and shorter water residence times closer to the inlets can dilute nutrient concentrations (Chaya et al. 2023) and may explain why increases in TN were not seen elsewhere in the Matanzas River.

Although nutrient levels have declined at most WQ monitoring stations, many individual TP, TN, and Chl-*a* concentrations still exceeded nutrient criteria values in recent years (e.g., 2018-2023), and WQ impairments for these parameters persist throughout the TRPU and MRPU. Impaired water segments include the Guana River (WBID 2320B2, 2320C, 2320), Tolomato River (2363I1), Matanzas River (2363H), and tidal creeks and streams (2406C, 2435, 2499, 2502C) which flow into the Tolomato and Matanzas. Although this study did not assess bacteria, many WBIDS are impaired for fecal coliforms, an indicator of pathogenic microorganisms (Ohrel and Register 2006), which can limit oyster harvesting due to increased risk of oyster-borne diseases. The Chl-*a* increases in the MRPU are likely linked to increasing water temperature, which can increase primary productivity and phytoplankton biomass (Harris et al. 2006). This is further supported by observed water temperature and Chl-*a* increases during the warmest months in two South Atlantic estuarine systems (Mallick and Dunn, 2024). Considering there are current nutrient impairments across the estuary, further reductions are needed to improve and maintain good WQ. Eutrophication within coastal estuaries is a management concern because it contributes to the degradation of WQ directly through increases in phytoplankton production and indirectly through hypoxia, algal toxins, decreases in light attenuation, increases in aquatic animal mortality, and habitat change (Medina et al. 2025). For example, eutrophication contributes to salt marsh loss (Deggan et al. 2012), can increase mangrove freeze resistance (Feller et al. 2023), and accelerate mangrove range-expansion within the GTM estuary (Dangremond et al. 2020).

Overall, analysis of long-term WQ trends suggests that water temperature is increasing at most stations across the TRPU and MRPU. Increasing water temperature throughout the two PU's is a management concern as it can directly affect WQ (e.g., increase phytoplankton, decrease DO) and facilitate the range expansion of aquatic species (Harris et al. 2006, Purtlebaugh et al. 2020, Chapra et al. 2021, Adams et al. 2024). Increases in salinity, pH and DO were observed across the system as well as a decrease in pH at one location. In general, increases in pH are beneficial for oysters (Waldbusser et al. 2011a, Waldbusser et al. 2011b) and other shell forming organisms and increases in DO can enhance overall WQ. While small increases in salinity are unlikely to have a large impact on biological communities there is evidence that euhaline salinities paired with warm water conditions can elevate oyster disease infection and mortality (Petes et al. 2012). Despite overall nutrient declines throughout the system, continued exceedances, rising Chl-*a,* and current FDEP-identified impairments indicate that further nutrient reductions are needed to improve WQ and prevent future impairments. Additionally, it is currently unknown how changes in floral and faunal species, resulting from increased water temperature and eutrophication may affect ecosystem dynamics. Implementation of species monitoring may help ecosystem managers track potential changes to biological communities and WQ.

3.5. PELLICER CREEK PLANNING UNIT

Summary

- There are three SJRWMD WQ monitoring stations across the Pellicer Creek Planning Unit (PCPU) (9B).

- Mean WQ parameters were relatively uniform across the three stations, with salinities ranging from polyhaline to euhaline.

- Increases in salinity and pH, and decreases in DO were minimal. The salinity trendline remained within the typical range for this region, while the DO trendline consistently exceeded the 3.0 mg/L hypoxia stress threshold by twofold.

- Trends in salinity may reflect decreases in rainfall or increases in tidal flushing; pH trends may be associated with coinciding increases in salinity and/or Ch-*a*; and DO trends may be influenced by increasing DOC levels.

- Increasing TP and Chl-*a* are management concerns for the PCPU. All stations showed evidence of elevated Chl-*a*, with trendlines near or above nutrient criteria values.

- WQ trends and current impairments for nutrients, Chl-*a*, and fecal coliforms across the region indicate that further improvements to WQ and additional reductions in nutrient loading are necessary within the PCPU.

Introduction

Pellicer Creek, located approximately 6 km (4 mi) south of the Matanzas Inlet, is the largest tributary of the Matanzas River and has connections to the ICW (Fig. 3.5.1; Dix 2019, Schafer et al. 2022). The Pellicer Creek watershed has a drainage basin of approximately 411 km^2 (159 mi^2) with salinities typically ranging from 0-35, which is influenced by freshwater inflow from creeks and brackish water from the Matanzas River (Frazel 2009, Schafer et al. 2022). The surrounding coastal watershed consists of a mix of mangrove, salt marsh, and oyster reef habitats. Dominant vegetation species include black mangroves (*Avicennia germinans*), cordgrass (*Spartina alterniflora*) and black needlerush (*Juncus roemerianus*).

The Pellicer Creek Planning Unit (PCPU) (9B) is located south of the State Road 206 bridge near Crescent Beach in St. Johns County and extends through northern Flagler County, ending near Flagler Beach (Fig. 3.5.1). The PCPU is bound in the north by the MRPU and in the south by the Halifax River Planning Unit (HRPU). Federal and state conservation areas surrounding Pellicer Creek, including the Guana Tolomato Matanzas National Estuarine Research Reserve (GTMNERR) make it one of the few relatively undisturbed tidal marsh

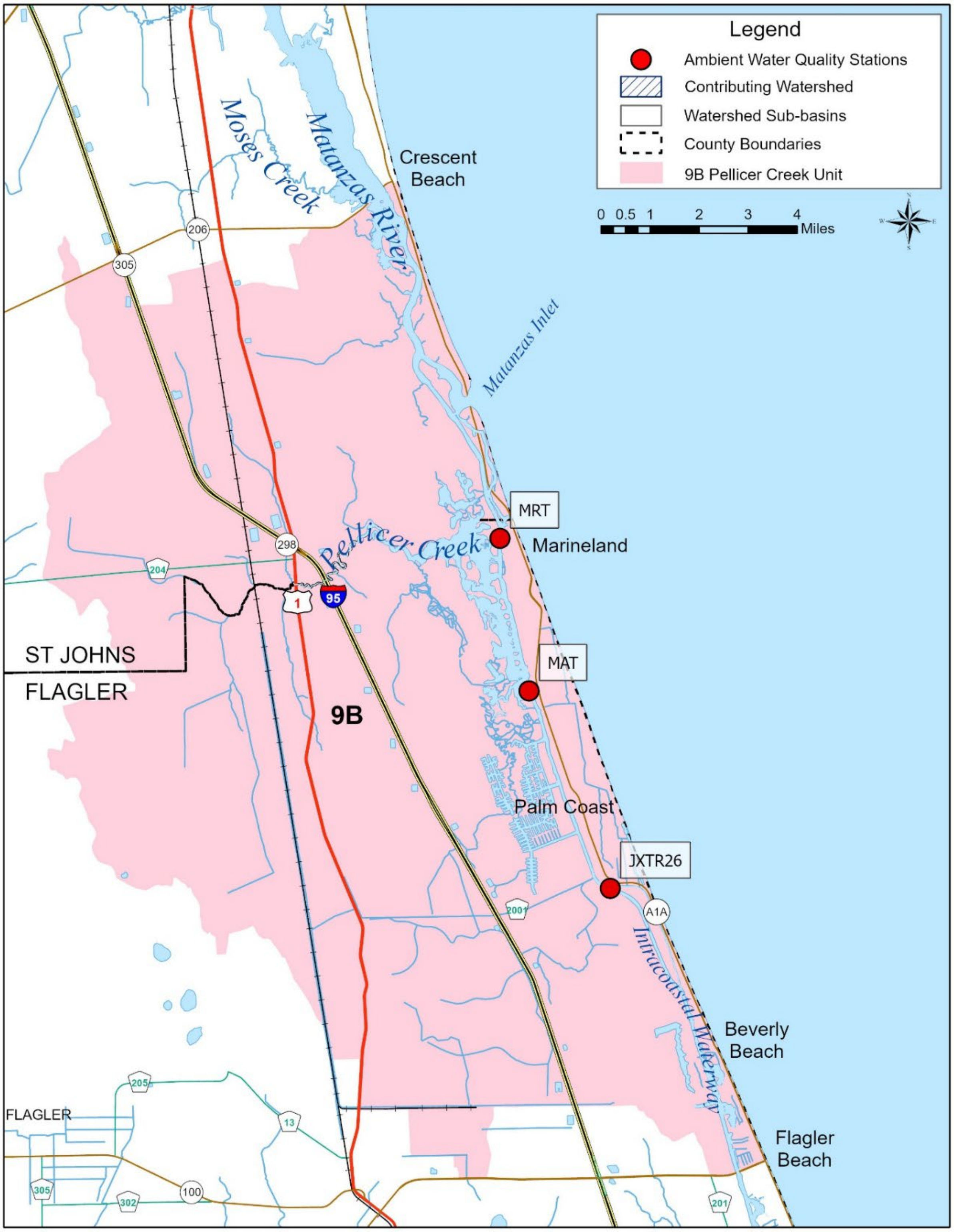

Figure 3.5.1. Map of the PCPU (9B). Red circles indicate SJRWMD WQ monitoring stations.

creek systems along the east coast of Florida (Frazel 2009). Pellicer Creek is an OFW, maintained as an aquatic preserve and designated as a State Canoe Trail. Waterways throughout this system include a mixture of Class II and Class III waters (Dix et al. 2019). Shellfish harvest area 88 includes the Matanzas River and ICW north of marker 87, and south of the State Road 206 bridge. A total of 6.2 km^2 (2.4 mi^2) out of 8.5 km^2 (3.3 mi^2) of shellfish harvest area 88 is conditionally approved, with harvest closing if cumulative rainfall exceeds 12.29 cm (4.84 in) in a single day (Fig.3.5.2).

Methods

For a full description of WQ parameters and analyses performed refer to Chapter 2. Sampling frequency differed between Pellicer Creek (MRT) and Matanzas River (MAT) stations. At Pellicer Creek, samples were collected bimonthly from 1986-1989 and monthly from 1997-2023. Due to the limited duration and subsequent data gap, the 1986-1989 data were excluded. At the Matanzas River station, data were collected bimonthly from 1986-1996 and monthly from 1997-2023. Because roughly one-third of the data were collected bimonthly, most WQ parameters were aggregated into bimonthly bins. Salinity, NOx-D and DOC were retained at monthly resolution due to data limitations; salinity had large data gaps from 1986-1996, NOx-D only had 1 data point in 1991 and 1992 followed by missing data from 1993-2004, and DOC sampling did not begin until 2011.

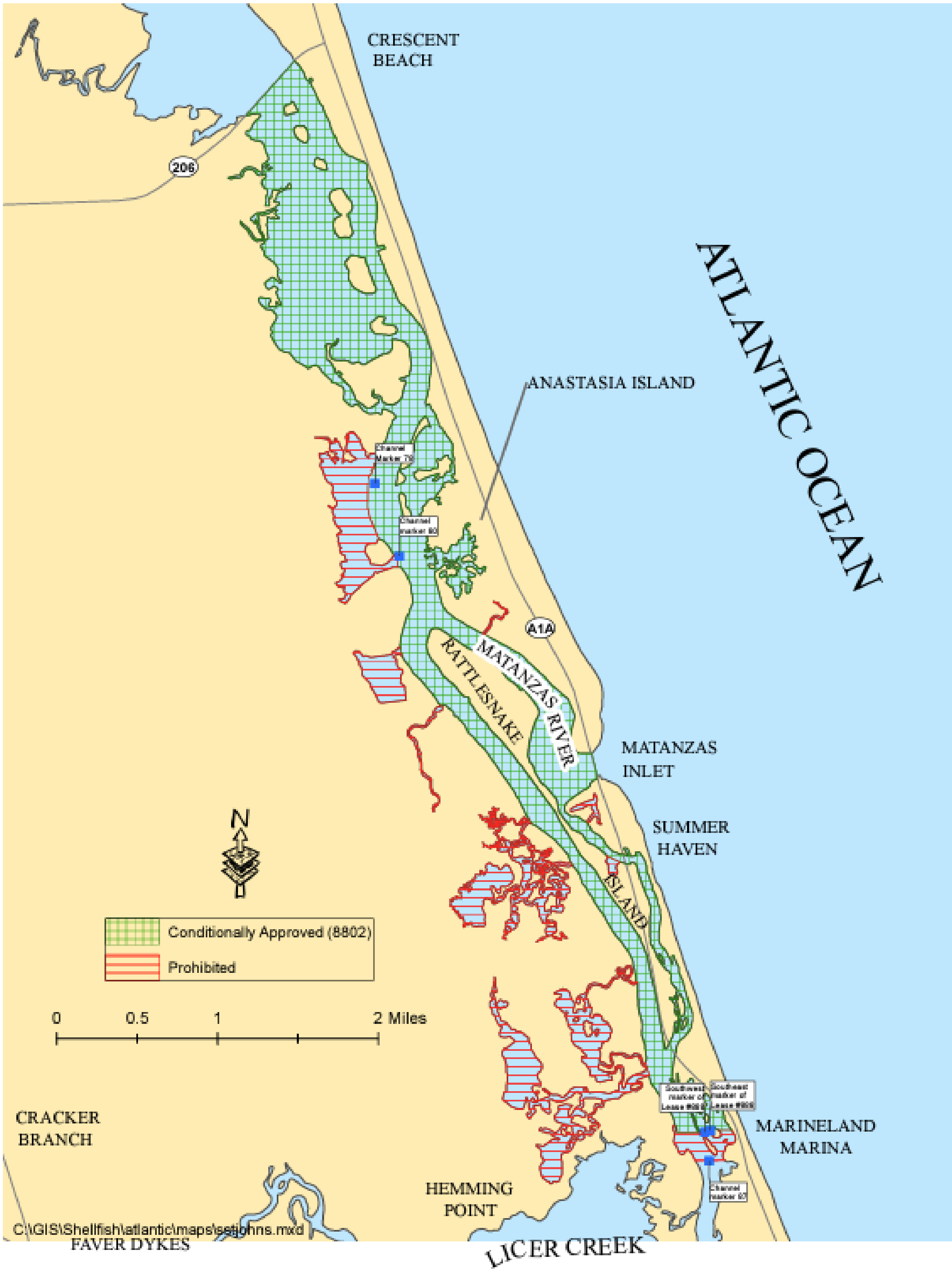

Figure 3.5.2. Map of open and prohibited shellfish harvesting areas within shellfish harvesting area 88 in St. Johns County as of August 27, 2018. Blue squares represent channel markers. Map provided by FDACS.

Results

Mean Water Quality Conditions

There are three SJRWMD WQ monitoring stations located within the PCPU (Fig. 3.5.1). Data collection in the Matanzas River (MAT) began in 1986, whereas Pellicer Creek (MRT) and ICW (JXTR26) began in 1997. Mean values for physical and chemical WQ parameters were similar across all three stations (Table 3.5.1, Table 3.5.2). Of the 13 parameters analyzed, conductivity, color, and TSS had the greatest variability.

Monthly Variability

Investigation of monthly variability among WQ parameters through KW tests showed differences for water temperature, pH, DO, Secchi, TP, TN, color, and Chl-*a* at all stations (Table 3.5.3). Differences across months were also found at the Matanzas River and ICW stations for DOC and for salinity and conductivity at the ICW station.

Station Water Quality Trends and Criteria

Increases in salinity, conductivity, pH, DOC, and Chl-*a* and decreases in TN, color, and TSS were found at the **Pellicer Creek station (MRT)** (Table 3.5.3). Over the 26-year period of record (1997-2023) water temperature remained near 23.0°C and salinity increased from 32.0 to 34.0 (Fig. 3.5.3A-B). The trendline for DO and all individual concentrations was above the 3.0 mg/L hypoxia threshold (Fig. 3.5.3C). While no change in TP was found from 1997-2023, individual concentrations occasionally exceeded the nutrient criteria value (Fig. 3.5.3D). TN decreased from 0.50 to 0.39 mg/L and Chl-*a* increased from 4.3 to 5.3 mg/m^3 between 1997 and 2023, with individual concentrations for both parameters often exceeding their respective nutrient criteria values (Fig. 3.5.3E-F).

At the **Matanzas River station (MAT)**, positive trends were found for pH, TP, DOC and Chl-*a* and a negative trend was found for TSS (Table 3.5.3). Water temperature was near 24.0°C from 1986 to 2023 and salinity around 32 from 1997 to 2023 (Fig. 3.5.4A-B). All individual DO concentrations and trendline were above 3.0 mg/L (Fig. 3.5.4C). Over the 37-year period of record (1986-2023), TP increased by 0.02 mg/L (0.07 mg/L to 0.09 mg/L) while TN remained near 0.48 mg/L, with individual TP and TN concentrations periodically exceeding respective nutrient criteria values (Fig. 3.5.4D-E). Chl-*a* increased from 5.8 mg/m^3 in 1986 to 6.9 mg/m^3 in 2023, with the trendline consistently above the 5.5 mg/m^3 numeric nutrient criteria value (Fig. 3.5.4F).

Table 3.5.1. Mean (±) standard deviation of physical WQ parameters for stations within the PCPU.

Station	Years	Temp. (°C)	Sal.	Cond. (µmhos/cm)	pH	DO (mg/L)	Secchi (m)
MRT	1997 - 2023	22.78 ± 5.38	31.42 ± 5.03	48098.70 ± 7090.95	7.84 ± 0.25	6.60 ± 1.31	0.86 ± 0.38
MAT	1986 - 2023	23.35 ± 5.42	30.64 ± 5.13	46796.23 ± 7193.07	7.76 ± 0.28	6.55 ± 1.22	0.92 ± 0.34
JXTR26	1997 - 2023	23.65 ± 5.65	28.66 ± 6.18	44227.65 ± 8793.28	7.74 ± 0.24	6.46 ± 1.32	0.96 ± 0.34

Table 3.5.2. Mean (±) standard deviation of chemical WQ parameters for stations within the PCPU.

Station	Years	TP (mg/L)	TN (mg/L)	NOx-D (mg/L)	DOC (mg/L)	Color (PCU)	TSS (mg/L)	Chl-*a* (mg/m^3)
MRT	1997 - 2023	0.09 ± 0.06	0.48 ± 0.24	0.02 ± 0.02	5.65 ± 4.34	49.01 ± 72.74	25.56 ± 17.23	5.55 ± 3.86
MAT	1986 - 2023	0.09 ± 0.06	0.50 ± 0.23	0.03 ± 0.05	5.91 ± 3.37	38.80 ± 43.58	30.23 ± 27.38	6.99 ± 4.34
JXTR26	1997 - 2023	0.10 ± 0.07	0.53 ± 0.22	0.03 ± 0.03	6.89 ± 3.80	41.17 ± 38.71	20.69 ± 14.94	7.34 ± 4.94

Table 3.5.3. Summary statistics for physical and chemical parameters at WQ monitoring stations in the PCPU.

	Temp. (°C)	Sal.	Cond. (µmhos/cm)	pH	DO (mg/L)	Secchi (m)	TP (mg/L)	TN (mg/L)	NOx-D (mg/L)	DOC (mg/L)	Color (PCU)	TSS (mg/L)	Chl-*a* (mg/m³)
	Station MRT (Pellicer Creek)												
KW (p)	**< 0.0001**	0.3260	0.3229	**< 0.0001**	**< 0.0001**	**< 0.0001**	**0.0015**	**< 0.0001**	0.9048	0.0699	**0.0252**	0.0970	**< 0.0001**
Kendall tau	0.060	0.092	0.097	0.132	-0.063	0.055	-0.074	-0.166	-0.055	0.169	-0.211	-0.246	0.105
Kendall (p)	0.1309	**0.0159**	**0.0107**	**0.0011**	0.1340	0.1857	0.0765	**0.0001**	0.4413	**0.0046**	**< 0.0001**	**< 0.0001**	**0.0093**
	Station MAT (Matanzas River)												
KW (p)	**< 0.0001**	0.1506	0.0981	**< 0.0001**	**< 0.0001**	**< 0.0001**	**< 0.0001**	**< 0.0001**	0. 7978	**0.0150**	**0.0001**	0.2703	**< 0.0001**
Kendall tau	0.026	0.025	0.064	0.291	-0.025	-0.049	0.114	-0.064	-0.082	0.137	-0.090	-0.370	0.107
Kendall (p)	0.5875	0.5133	0.1594	**< 0.0001**	0.5875	0.3407	**0.0156**	0.1824	0.2196	**0.0360**	0.0698	**< 0.0001**	**0.0271**
	Station JXTR26 (ICW)												
KW (p)	**< 0.0001**	**0.0295**	**0.0438**	**< 0.0001**	**< 0.0001**	**< 0.0001**	**< 0.0001**	**< 0.0001**	0.2948	**0.0069**	**0.0008**	0.1697	**< 0.0001**
Kendall tau	0.060	0.012	0.017	0.015	-0.111	0.022	0.005	-0.051	-0.118	0.134	-0.184	-0.340	0.046
Kendall (p)	0.1272	0.7877	0.3955	0.7014	**0.0067**	0.5454	0.8771	0.2337	0.0527	**0.0455**	**< 0.0001**	**< 0.0001**	0.2500

If $p < 0.05$ for the Kruskal–Wallis (KW) test, a Seasonal Kendall test was performed. If $p \geq 0.05$ for the KW test, a Mann-Kendall test was performed. Significant test results ($p < 0.05$) are shown in bold.

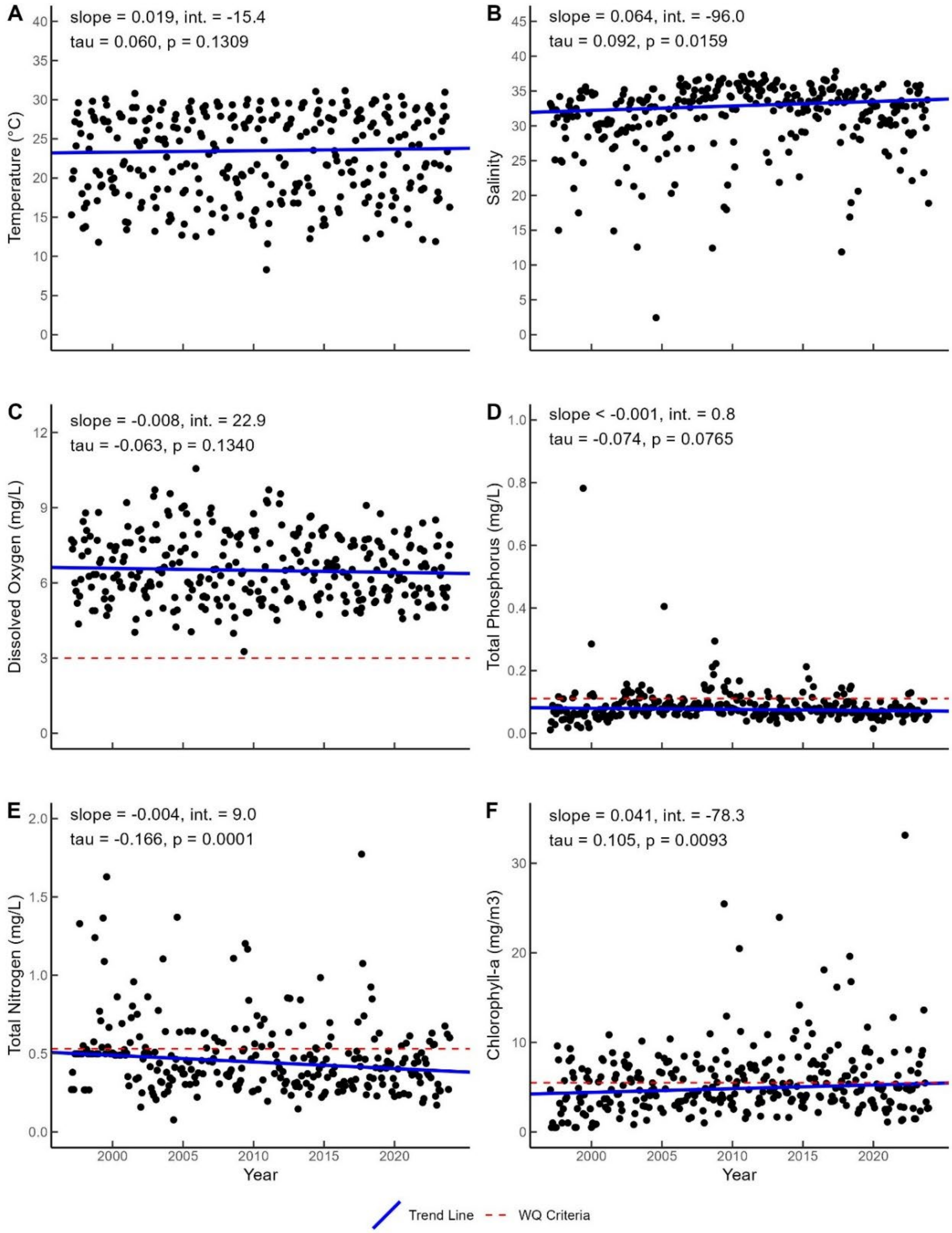

Figure 3.5.3. Trends for temperature, salinity, DO, TP, TN, and Chl-*a* (A-F) at the Pellicer Creek station (MRT). Seasonal Kendall = temperature, DO, TP, TN, Chl-*a*; Non-seasonal Kendall = salinity. The solid blue line depicts the monotonic trend for each parameter, with $p < 0.05$ denoting significant trends. The dashed red line indicates WQ criteria based on the 3.0 mg/L hypoxia stress threshold for DO and annual geometric mean values for TP, TN, and Chl-*a* outlined in F.A.C. Rule 62-302.532.

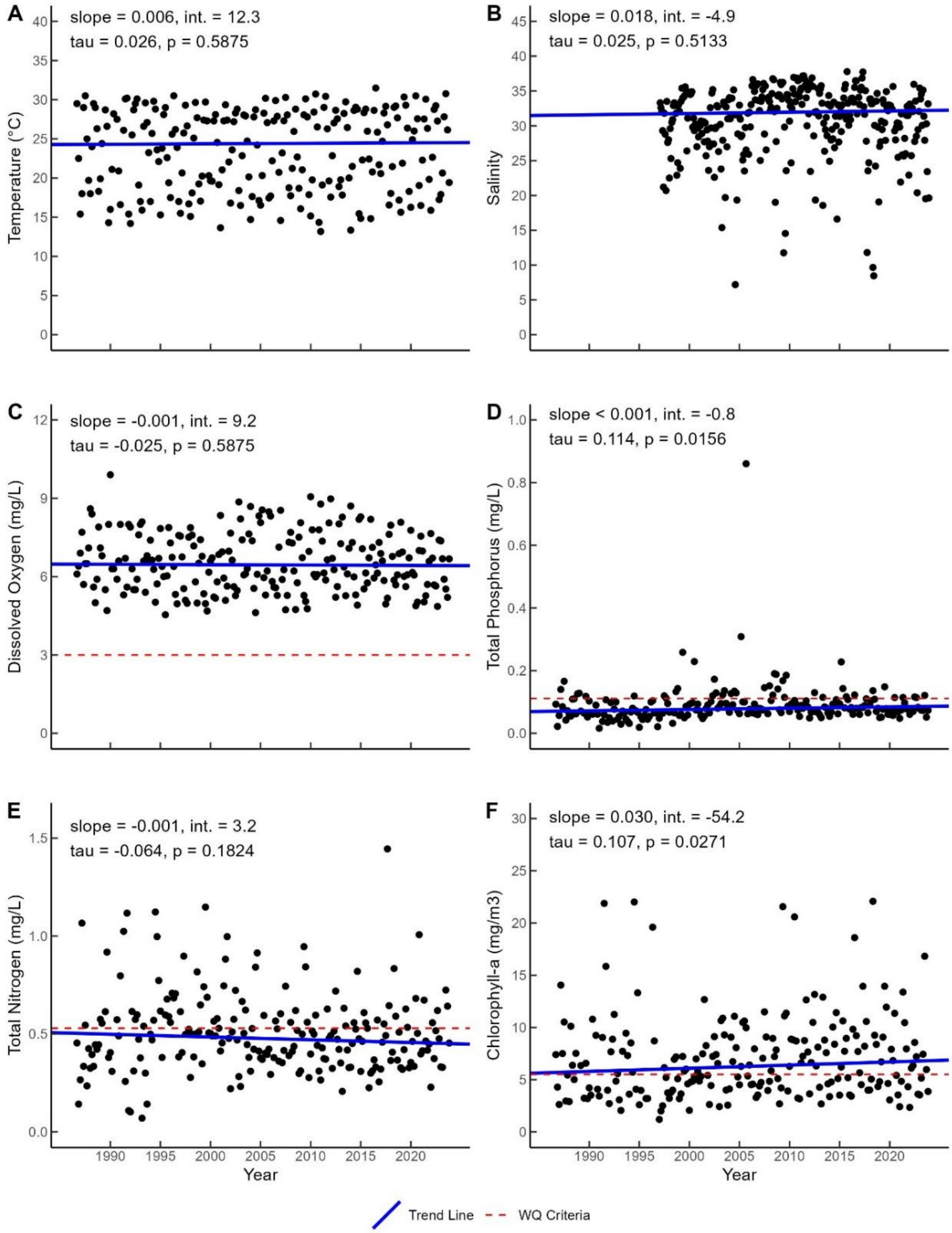

Figure 3.5.4. Trends for temperature, salinity, DO, TP, TN, and Chl-*a* (A-F) at the Matanzas River station (MAT). Seasonal Kendall = temperature, DO, TP, TN, Chl-*a*; Non-seasonal Kendall = salinity. The solid blue line depicts the monotonic trend for each parameter, with $p < 0.05$ denoting significant trends. The dashed red line indicates WQ criteria based on the 3.0 mg/L hypoxia stress threshold for DO and annual geometric mean values for TP, TN, and Chl-*a* outlined in F.A.C. Rule 62-302.532.

Evidence of a marginally significant increasing trend in DOC and decreasing trends in DO, color and TSS were found at the **ICW station (JXTR26)** (Table 3.5.3). Water temperature was near 24.0°C and salinity near 30 from 1997 to 2023 (Fig. 3.5.5A-B). Across the 26-year period DO decreased from 6.6 mg/L to 6.2 mg/L, however all individual concentrations were above 3.0 mg/L (Fig. 3.5.5C). No changes in TP, TN, or Chl-*a* were found (Fig. 3.5.5D-F). Most TP concentrations were below 0.25 mg/L and TN below 1.0 mg/L. The trendline and many individual Chl-*a* concentrations were above the 4.5 mg/m^3 nutrient criteria value for this location.

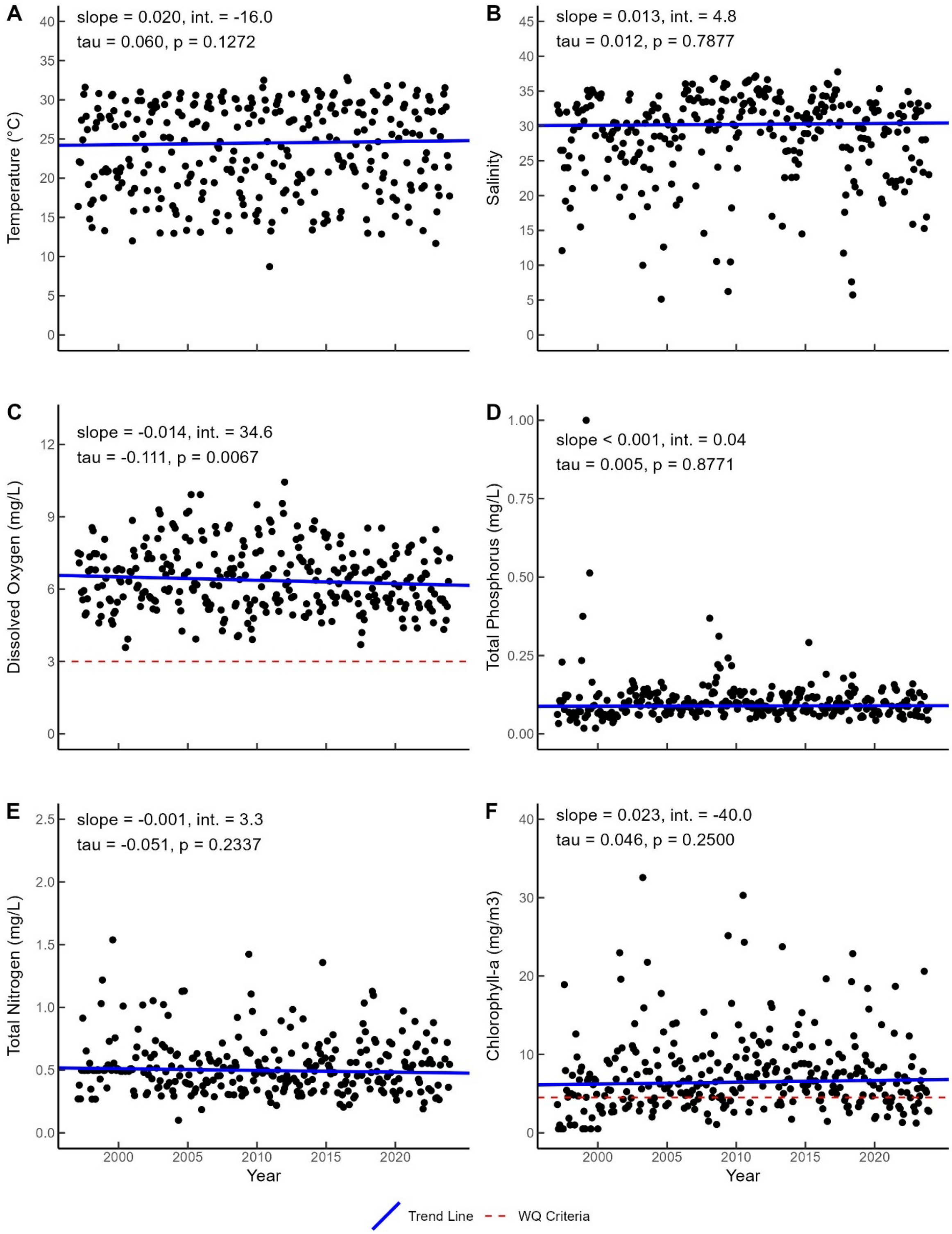

Figure 3.5.5. Seasonal Kendall trends for temperature, salinity, DO, TP, TN and Chl-*a* (A-F) at the ICW station (JXTR26). The solid blue line depicts the monotonic trend for each parameter, with p < 0.05 denoting significant trends. The dashed red line indicates WQ criteria based on the 3.0 mg/L hypoxia stress threshold for DO and annual geometric mean values for TP, TN, and Chl-*a* outlined in F.A.C. Rule 62-302.532.

Discussion

Pellicer Creek is a tributary to the GTM estuary, which is a well-mixed, ocean dominated (high salinity) estuarine system (Hart et al. 2015, Kimball and Eash-Loucks 2021, Chaya et al. 2023) and it was expected that mean WQ parameters would be relatively similar across the stations. Physical and chemical WQ parameters were relatively uniform among the three monitoring stations, with average conditions being polyhaline or euhaline. Station pH fell within typical ranges reported for estuaries (Ohrel and Register 2006). The high variation observed for color and TSS was also expected due to their sensitivity to rainfall and tidal mixing (FLW 2004, Phlips et al. 2004). Additionally, monthly differences were found for most WQ parameters, which are consistent with the results of other WQ studies conducted throughout this region (Hart et al. 2015, Balthis et al. 2017).

Trends in WQ parameters were observed at all three monitoring stations, with changes ranging from low ($tau < 0.3$) to moderate ($0.3 \leq tau < 0.5$). The small increase in salinity observed at the Pellicer Creek station, located near the Matanzas Inlet, remained within the typical range for this region and may be attributed to decreased precipitation or increased tidal flushing, both of which can elevate salinity (Hart et al. 2015, Chaya et al. 2023). From a management perspective, small, gradual changes to salinity are unlikely to negatively impact estuarine aquatic communities. Instead, freshwater pulses from storm events pose a greater concern for this system. Historically, the estuary has experienced acute decreases in salinity resulting from high rainfall associated with Hurricanes Charley, Frances, Ivan, and Jeanne in 2004 (Sheng et al. 2008) and Irma in 2017 (Kimmel et al. 2024). Prolonged periods of low salinity can negatively impact fish survival and development (Bachman and Rand, 2008) and reduce oyster recruitment, survival and growth (La Peyre et al. 2013, Parker et al. 2013). In addition, oyster reef faunal communities in low salinity waters typically show reduced abundance, biomass and diversity compared to those in higher salinities (Tolley et al. 2005, Marshall et al. 2019).

Increasing pH was also observed at two locations in the PCPU. At the Pellicer Creek station, this trend likely reflects a concurrent increase in salinity, consistent with findings by Chaya et al. (2023) in the GTM estuary. Elevated pH at both stations may also be linked to coinciding increases in Chl-*a* concentrations, as phytoplankton biomass has been positively correlated with pH (Carstensen and Duarte 2019, Raven et al. 2020, Shen et al. 2020, Hall et al. 2023) in other estuarine systems. Although overall pH changes were low, higher pH levels can be beneficial for oyster growth (Waldbusser et al. 2011a, Keppel et al. 2016) and reduce oyster shell dissolution rates (Waldbusser et al. 2011b).

Although a decline in DO was observed at the ICW station, levels remain well above the 3.0 mg/L hypoxia stress threshold, with the trendline nearly double this value and all individual measurements exceeding it. The decrease in DO may be linked to a coinciding increase in DOC, which has been associated with increases in biological oxygen demand (Stedmon et al. 2006, McCabe et al. 2021). An inverse relationship between DO and fluorescent DOM has also been reported for the GTM estuary (Chaya et al. 2023). In aquatic environments, DOC provides food for bacteria, which can contribute to DO depletion from the water column (Kerr et al. 2013). While DO concentrations appear satisfactory across all three stations, a portion of the brackish region of Pellicer Creek (WBID 2580B) is currently impaired for DO. Nutrients have been identified as the causative pollutant based on FDEP's 2022-2024 WQ Assessment, TMDLs, and BMAPs. This segment also has a TMDL for fecal coliforms and although not currently impaired for Chl-*a*, the annual geometric mean for this parameter has failed to meet the established criterion for the past three years, suggesting recent contractions have been elevated.

Nutrients and Chl-*a* trends varied across the PCPU stations. Regardless of trend direction, all three stations exhibited elevated Chl-*a*, with trendlines near or exceeding nutrient criteria values. Nutrient enrichment is well known to elevate phytoplankton growth within estuarine systems (Boyer et al. 2009, Lapointe et al. 2015, Lapointe et al. 2017), with increasing Chl-*a* at the Matanzas River station likely explained by the coinciding increase in TP. Conversely, the increase in Chl-*a* at the Pellicer Creek station was unexpected, given the concurrent decrease in TN and the absence of significant changes in other parameters (e.g., water temperature) that might otherwise account for elevated Chl-*a*. However, the trend for TN at this station may have been affected by changes to the MDL which was higher during the late 1990s to early 2000s compared to following years. Changes in data variability can affect the statistical power of Kendall tests to detect true changes over time (Meals et al. 2011, Helsel et al. 2020), therefore TN may not have declined as much as the data indicated. Nonetheless, modeling nutrient-phytoplankton relationships in dynamic coastal systems is challenging due to the dampening effects of high tidal flushing or freshwater discharge on phytoplankton biomass (Paerl et al. 2014). Studies within the GTM estuary have emphasized the importance of water residence time, rainfall and tidal flushing in the regulation of phytoplankton biomass (Hart et al. 2015, Chaya et al. 2023). While multiple mechanisms are likely to contribute to rising Chl-*a* throughout the PCPU, the observation of elevated Chl-*a* is supported by Hart et al. (2015) and existing WQ impairments. Currently a segment of the ICW (WBID 2363D) is impaired for TP, TN, and Chl-*a* and has a TMDL for nutrients. Additionally, FDEP's 2022-2024 WQ Assessment, TMDLs, and BMAPs reported elevated Chl-*a* for WBID 2580A (adjacent to WBID 2580B) in Pellicer Creek.

Overall, observed changes in physical WQ parameters across stations do not raise immediate concern within the PCPU. However, increasing trends in TP and Chl-*a*, along with Chl-*a* levels approaching or exceeding nutrient criteria values present potential management concerns for the long-term health of the system. For example, eutrophic conditions from high nutrient inputs may contribute to coastal habitat change by increasing mangrove freeze resistance (Feller et al. 2023) and accelerating mangrove range-expansion (Dangremond et al. 2020). Furthermore, elevated phytoplankton production can lower WQ by introducing algal toxins, decreasing light attenuation, and increasing the risk of hypoxia in bottom waters (Medina et al. 2025). Additionally, fecal coliform impairments, which are used as indicators of potential pathogenic contamination (Ohrel and Register 2006), present in Pellicer Creek

(WBIDS 2580A-B) and the Matanzas River (WBID 2363F) raise concerns for oyster populations and harvest shellfish. Maintaining good WQ is essential for sustaining healthy oyster populations (Dix et al. 2019) which provide both ecological and socioeconomic benefits in the PCPU. Oysters filter nearly 60% of the GTM estuary's total water volume (Gray et al. 2021) and help mitigate eutrophication by removing phytoplankton and nutrients from the water column (Kellogg et al. 2014). Together the data and results presented in this subchapter, and current impairments for nutrients, Chl-*a*, and fecal coliforms throughout Pellicer Creek suggest that further improvements to WQ and nutrient loading are needed within the PCPU.

3.6. HALIFAX RIVER PLANNING UNIT

Summary

- There are six SJRWMD WQ monitoring stations within the Halifax River Planning Unit (HRPU) (9A). Although the NCB focuses on estuarine waters, two freshwater tributary stations, which were established for hydrodynamic modeling, were included to provide additional information on WQ trends throughout this PU.

- Mean water temperature, Secchi depth and NOx-D were similar across all six stations, with tributary stations having lower salinity, conductivity, pH, TSS, and Chl-*a* and higher DOC and color.

- Increasing water temperature is a management concern since it can directly impact WQ (e.g., decrease DO, increase Chl-*a*) and estuarine community dynamics.

- Small, gradual declines in salinity observed at several stations are not currently considered a management priority and may reflect variability in rainfall, water levels, and the frequency of episodic storm events within the region.

- Stations with decreasing DO are not currently at risk of impairment, however other regions throughout the PU have documented impairments, indicating a need for broader WQ improvements.

- Improvements to water management practices can explain declines in nutrients and Chl-*a*. However, despite these declines many areas throughout the HRPU currently have nutrient and Chl-*a* impairments, and further reductions are necessary to improve and maintain good WQ.

Introduction

The Halifax River is a 40 km (25 mi) long, shallow estuarine lagoon system that runs along the eastern coast of Volusia County and is a part of the ICW (Goolsby et al. 2016). The river extends from the Tomoka Basin southward where it connects with the Atlantic Ocean through the Ponce De Leon Inlet (Fig.3.6.1). The Halifax River basin encompasses approximately 1,035 km^2 (399 mi^2) and receives freshwater from the Tomoka River and Bulow Creek in the north and Spruce Creek in the south (Fig. 3.6.1; Dix et al. 2021, Reiter and Cho 2020). Tides within the Halifax River are semi-diurnal with a tidal amplitude of 0.6 m (2 ft.) and mean annual salinities typically range from 24-30, with significant spatial and temporal variability depending on rainfall levels and tidal mixing (Cho et al. 2020).

The Halifax River Planning Unit (HRPU) (9A) extends from the southern portion of Flagler County through Volusia County and is bound in the north by the PCPU (Fig. 3.6.1). Alterations to the Halifax River watershed including construction of the ICW and other canals, dragline ditching and impounding wetlands, dredging, and urbanization have had major impacts on circulation, tidal exchange, wetland function, and water transport into the system. In the 1920s a series of canals, known as the "Halifax Drainage District" were constructed to drain floodplains for agriculture and development by diverting water into the Halifax and Tomoka Rivers as well as Bulow and Spruce Creek (Jacoby 2023). This series of canals currently delivers more water, nutrients, and sediments into the Halifax River and its tributaries than would have occurred under natural conditions (Jacoby 2023). From the 1950s to1970s, a network of dragline ditches was created within coastal wetlands along the Halifax River to manage salt marsh mosquito populations (Cho et al. 2020, Jacoby 2023). Although reduction of mosquito populations was achieved, the construction of these ditches negatively impacted the ecological functions of these tidal wetlands (Cho et al. 2020). The Halifax River is highly urbanized and only a small portion of natural shoreline and wetlands remain (Goolsby et al. 2016, Reiter and Cho 2020). Nearly 80% of Volusia County's population (553,000 residents as of 2019) live within the Halifax River, Tomoka River and Bulow Creek watersheds, making urban runoff and associated pollutants a major concern for WQ within the system (Reiter and Cho 2020).

Methods

For a full description of WQ parameters and analyses performed refer to Chapter 2. While the focus of the HRPU is estuarine waters, freshwater tributary stations [Tomoka River south (27010579) and Spruce Creek (02248000)] were established for hydrodynamic modeling.

Most stations within the HRPU had systematic differences in sampling frequency. At the Tomoka River south station samples were collected bimonthly from 1995-1997 and monthly from 1998-2023. Since the majority of samples were collected monthly, the data from 1995-1997 was removed. Additionally, salinity data from 1998-2002 was removed since it was recorded as discrete and all other years had continuous values. Chl-*a* analysis began in 2007, due to missing data from 1999-2006.

Bimonthly data was collected for the Tomoka River north station from 1995-2014, Spruce Creek from 1995-2004, and Bulow Creek from 1986-2014, with most WQ parameters for these stations being aggregated into bimonthly bins. At Spruce Creek, salinity data from 1998-2001 was removed since it was recorded as discrete values and Secchi, NOx-D and DOC were left at monthly intervals since the period of record used began during monthly WQ data collection (start year: 2014 = Secchi, 2018 = NOx-D/DOC). At Bulow Creek data was removed from 1986-1997 for salinity, 1986-2004 for NOx-D, and 1986-2012 for Secchi since most data was missing for these parameters across the respective years.

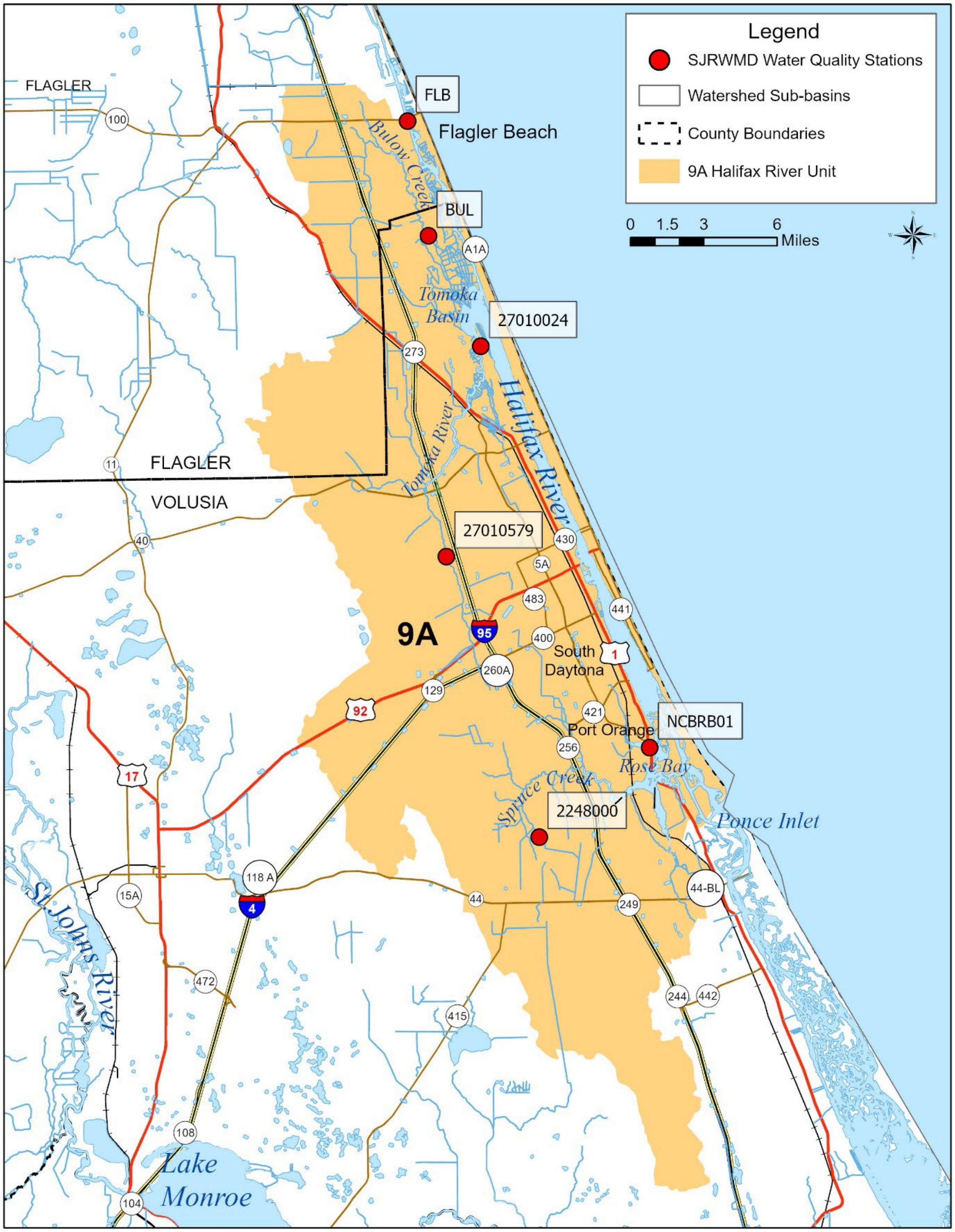

Figure 3.6.1. Map of the Halifax River (9A) Planning Unit. Red circles indicate SJRWMD WQ monitoring stations.

Results

Mean Water Quality Conditions

There are six SJRWMD WQ monitoring stations located within the HRPU (Fig. 3.6.1). Of the six stations, Bulow Creek (BUL) has the longest period of record (since 1986) while Rose Bay (NCBRB01) is the newest, starting in 2005 (Table 3.6.1). Mean water temperature, Secchi depth, and NOx-D were similar across all stations (Table 3.6.1, Table 3.6.2). The Tomoka River south (27010579) and Spruce Creek (02248000) stations had the lowest salinity, conductivity, pH, TSS and Chl-*a* and highest mean DOC and color. Additionally, TP was lowest at the Tomoka River south station and highest at the Tomoka River north station (27010024) (Table 3.6.2).

Monthly Variability

Differences across months were found for water temperature, salinity, conductivity, pH, DO, TP, TN, DOC, and Chl-*a* at all stations (Table 3.6.3). Monthly variability was also found for color at all stations except Tomoka River south. The KW tests were also significant for TSS at the ICW, Bulow Creek, Tomoka River north, and Rose Bay stations and Secchi and NOx-D at Rose Bay.

Station Water Quality Trends and Criteria

Decreases in salinity, conductivity, DO, TP, NOx-D, and TSS and an increase in color were observed at the **ICW station (FLB)** (Table 3.6.3). Significant heterogeneity was found for pH (Chi-Square Heterogeneity Test, $p = 0.0190$) with negative trends for all months except August and September, which had positive trends, and trends being significant for January, February, April, and October-December (Table 3.6.4). From 2004 to 2023, water temperature remained near 24.0°C while salinity decreased by 2.9 (31.7 to 28.8) (Fig. 3.6.2A-B). Over the 19-year period, DO decreased from 6.5 mg/L to 5.9 mg/L, however, the trendline and most individual concentrations were above the 3.0 mg/L hypoxia threshold (Fig. 3.6.2C). TP decreased from 0.12 mg/L in 2004 to 0.10 mg/L in 2023, whereas TN stayed near 0.60 mg/L, with most TP concentrations below 0.30 mg/L and TN concentrations below 1.5 mg/L (Fig. 3.6.2D-E). While no change in Chl-*a* was found, the trendline and most individual concentrations exceeded the numeric nutrient criteria value of 4.5 mg/m^3 (Fig. 3.6.2F).

Table 3.6.1. Mean (±) standard deviation of physical WQ parameters for stations within the HRPU.

Station	Years	Temp. (°C)	Sal.	Cond. (µmhos/cm)	pH	DO (mg/L)	Secchi (m)
FLB	2004 – 2023	24.02 ± 5.68	27.77 ± 6.90	43013.94 ± 9833.14	7.74 ± 0.18	6.25 ± 1.41	0.87 ± 0.37
BUL	1986 – 2023	23.65 ± 5.40	12.03 ± 9.87	19377.49 ± 15477.69	7.29 ± 0.40	4.88 ± 1.81	0.48 ± 0.19
27010024	1995 – 2023	23.72 ± 5.33	16.53 ± 9.84	25254.17 ± 14418.79	7.46 ± 0.35	5.67 ± 1.69	0.58 ± 0.27
27010579	1998 – 2023	21.02 ± 4.68	0.15 ± 0.06	310.91 ± 125.96	7.05 ± 0.43	5.43 ± 1.80	0.56 ± 0.23
NCBRB01	2005 – 2023	24.37 ± 5.73	26.95 ± 6.27	41872.43 ± 8925.10	7.78 ± 0.23	6.22 ± 1.34	0.86 ± 0.27
02248000	1995 – 2023	21.54 ± 4.22	0.19 ± 0.10	375.77 ± 197.78	7.06 ± 0.49	6.12 ± 1.35	0.34 ± 0.12

Table 3.6.2. Mean (±) standard deviation of chemical WQ parameters for stations within the HRPU.

Station	Years	TP (mg/L)	TN (mg/L)	NOx-D (mg/L)	DOC (mg/L)	Color (PCU)	TSS (mg/L)	Chl-*a* (mg/m^3)
FLB	2004 – 2023	0.12 ± 0.06	0.67 ± 0.29	0.05 ± 0.08	8.12 ± 4.20	38.32 ± 36.57	19.86 ± 13.51	11.08 ± 8.41
BUL	1986 – 2023	0.12 ± 0.06	1.11 ± 0.32	0.06 ± 0.04	17.82 ± 5.68	111.15 ± 85.57	17.41 ± 17.13	15.78 ± 18.66
27010024	1995 – 2023	0.12 ± 0.05	4.79 ± 55.80	0.05 ± 0.03	18.20 ± 8.41	151.58 ± 156.24	15.92 ± 12.96	8.61 ± 5.85
27010579	1998 – 2023	0.05 ± 0.05	1.00 ± 0.43	0.08 ± 0.04	32.38 ± 13.46	236.25 ± 213.22	4.57 ± 3.19	1.56 ± 1.02
NCBRB01	2005 – 2023	0.12 ± 0.05	0.61 ± 0.21	0.06 ± 0.07	7.95 ± 4.08	36.52 ± 39.86	18.18 ± 14.09	9.51 ± 6.72
02248000	1995 – 2023	0.13 ± 0.11	1.16 ± 0.39	0.09 ± 0.03	32.80 ± 12.82	302.29 ± 227.51	7.02 ± 9.90	4.18 ± 16.70

Table 3.6.3. Summary statistics for physical and chemical parameters at WQ monitoring stations in the HRPU.

	Temp. (°C)	Sal.	Cond. (µmhos/cm)	pH	DO (mg/L)	Secchi (m)	TP (mg/L)	TN (mg/L)	NOx-D (mg/L)	DOC (mg/L)	Color (PCU)	TSS (mg/L)	Chl-*a* (mg/m³)
	Station FLB (ICW)												
KW (p)	**< 0.0001**	**0.0079**	**0.0110**	**< 0.0001**	**< 0.0001**	**< 0.0001**	**< 0.0001**	**< 0.0001**	**0.0166**	**< 0.0001**	**< 0.0001**	**0.0185**	**< 0.0001**
Kendall tau	0.094	-0.145	-0.152	-	-0.107	<0.001	-0.130	0.007	-0.223	0.102	0.144	-0.355	-0.088
Kendall (p)	0.0539	**0.0031**	**0.0019**	-	**0.0367**	1.0000	**0.0109**	0.9181	**0.0021**	0.1316	**0.0054**	**< 0.0001**	0.0879
	Station BUL (Bulow Creek)												
KW (p)	**< 0.0001**	**0.0020**	**0.0066**	**< 0.0001**	**< 0.0001**	0.1924	**< 0.0001**	**< 0.0001**	0.1176	**0.0418**	**< 0.0001**	**0.0032**	**< 0.0001**
Kendall tau	-0.051	0.053	-0.040	0.096	-0.138	-0.073	0.184	-0.011	0.112	0.071	-0.038	-0.204	-0.046
Kendall (p)	0.2768	0.3749	0.4044	**0.0488**	**0.0038**	0.5014	**0.0001**	0.8228	0.1314	0.4346	0.4186	**< 0.0001**	0.3136
	Station 27010024 (Tomoka River north)												
KW (p)	**< 0.0001**	**< 0.0001**	**< 0.0001**	**< 0.0001**	**< 0.0001**	0.0895	**< 0.0001**	**< 0.0001**	**0.0302**	**0.0034**	**< 0.0001**	**0.0097**	**0.0004**
Kendall tau	0.106	-0.011	-0.061	-	0.022	-0.029	0.072	-0.144	0.054	0.203	0.019	-0.279	-0.018
Kendall (p)	0.0580	0.8399	0.2757	-	0.6781	0.7537	0.1971	**0.0108**	0.5791	**0.0247**	0.7301	**< 0.0001**	0.7058
	Station 27010579 (Tomoka River south)												
KW (p)	**< 0.0001**	**0.0044**	**< 0.0001**	**0.0002**	**< 0.0001**	0.2212	**< 0.0001**	**< 0.0001**	0.6212	**0.0291**	0.1934	0.8265	**0.0155**
Kendall tau	0.280	0.060	0.067	0.130	-0.034	0.147	0.133	0.064	0.125	-0.386	0.303	-0.229	0.058
Kendall (p)	**< 0.0001**	0.2649	0.1316	**0.0032**	0.4074	**0.0377**	**0.0028**	0.1488	0.2294	**0.0407**	**< 0.0001**	**< 0.0001**	0.3606
	Station NCBRB01 (Rose Bay)												
KW (p)	**< 0.0001**	**0.0018**	**0.0016**	**< 0.0001**	**< 0.0001**	**< 0.0001**	**< 0.0001**	**< 0.0001**	**< 0.0001**	**< 0.0001**	**< 0.0001**	0.3234	**< 0.0001**
Kendall tau	0.111	-0.137	-0.139	-0.062	-0.145	0.168	-0.327	-0.171	0.021	0.003	0.124	-0.392	-0.247
Kendall (p)	0.0573	**0.0135**	**0.0124**	0.2709	**0.0101**	**0.0057**	**< 0.0001**	**0.0022**	0.7829	1.0000	**0.0299**	**< 0.0001**	**< 0.0001**
	Station 02248000 (Spruce Creek)												
KW (p)	**< 0.0001**	**< 0.0001**	**< 0.0001**	**0.0203**	**< 0.0001**	0.6601	**< 0.0001**	**< 0.0001**	0. 3575	**0.0240**	**< 0.0001**	**0.0171**	**0.0083**
Kendall tau	0.146	0.052	0.135	0.062	-0.063	-0.061	-0.288	-0.115	-0.068	-0.142	-0.074	-0.291	-0.074
Kendall (p)	**0.0111**	0.4305	**0.0192**	0.2912	0.2753	0.9537	**< 0.0001**	**0.0359**	0. 8508	0. 4350	0.2913	**< 0.0001**	0.4394

If $p < 0.05$ for the Kruskal–Wallis (KW) test, a Seasonal Kendall test was performed. If $p \geq 0.05$ for the KW test, a Mann-Kendall test was performed. Significant test results ($p < 0.05$) are shown in bold.

Table 3.6.4. Results for individual monthly Mann-Kendall tests run for pH at the ICW (FLB) and northern Tomoka River (27010024) stations.

	FLB		27010024	
Month	**tau**	**p-value**	**tau**	**p-value**
January	-0.346	**0.0486**	0.183	0.1789
February	-0.346	**0.0486**	-	-
March	-0.333	0.0575	0.444	**0.0010**
April	-0.368	**0.0299**	-	-
May	-0.111	0.5433	0.347	**0.0102**
June	-0.281	0.1111	-	-
July	-0.047	0.8064	0.339	**0.0121**
August	0.205	0.2337	-	-
September	0.117	0.6659	0.108	0.4536
October	-0.442	**0.0189**	-	-
November	-0.523	**0.0027**	-0.138	0.3135
December	-0.563	**0.0006**	-	-

Data for northern Tomoka River was aggregated into bimonthly bins. Significant test results ($p < 0.05$) are shown in bold.

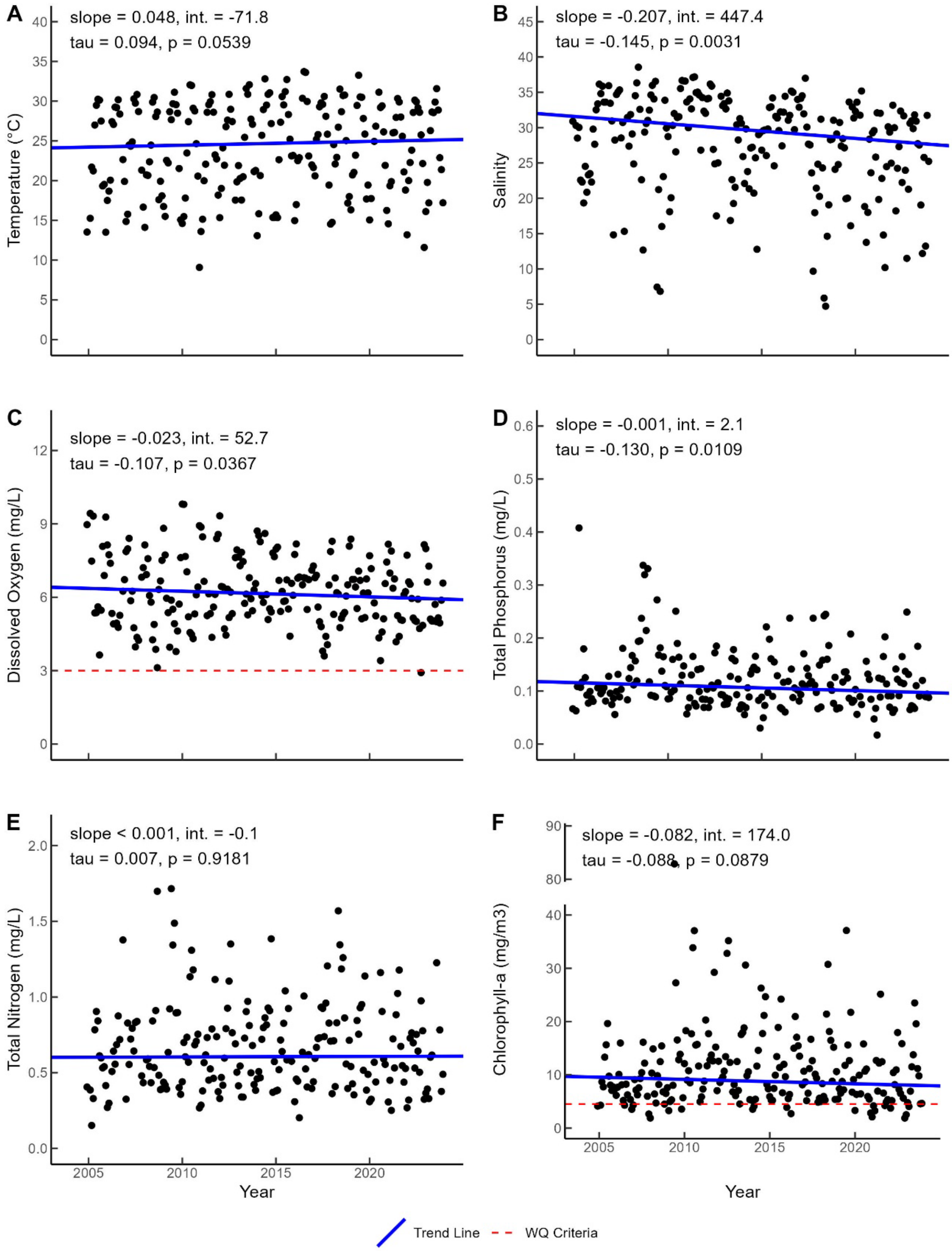

Figure 3.6.2. Seasonal Kendall trends for temperature, salinity, DO, TP, TN and Chl-*a* (A-F) at the ICW station (FLB). The solid blue line depicts the monotonic trend for each parameter, with p < 0.05 denoting significant trends. The dashed red line indicates WQ criteria based on the 3.0 mg/L hypoxia stress threshold for DO and annual geometric mean values for Chl-*a* outlined in F.A.C. Rule 62-302.532.

At the **Bulow Creek station (BUL)**, increases in pH and TP and decreases in DO and TSS were observed (Table 3.6.3). Water temperature was around 24.0°C from 1986 to 2023 and salinity between 9 and 10 from 1998 to 2023 (Fig. 3.6.3A-B). Across the 37-year (1986-2023) period of record, DO decreased from 5.1 mg/L to 4.3 mg/L, with individual concentrations periodically falling below 3.0 mg/L (Fig. 3.6.3C). From 1986 to 2023, TP increased from 0.10 mg/L to 0.13 mg/L but TN remained near 1.1 mg/L (Fig. 3.6.3D-E). While no trend was found for Chl-*a*, the trendline and many concentrations exceeded the 11 mg/m^3 criteria value (Fig. 3.6.3F).

An increase in DOC and decrease in TN were found at the **Tomoka River north station (27010024)** (Table 3.6.3). Significant heterogeneity was found for pH (Chi-Square Heterogeneity Test, $p = 0.0292$) with positive trends for all bimonthly sampling bins except November-December and significant trends being found for March-April, May-June and July-August (Table 3.6.4). From 1995 to 2023, water temperature was near 24.0°C and from 1998 to 2023 salinity was around 18 (Fig. 3.6.4A-B). Additionally, no change in DO or TP were found with most DO concentrations being above 3.0 mg/L and TP concentrations periodically exceeding the numeric nutrient criteria value of 0.132 mg/L (Fig. 3.6.4C-D). TN decreased from 1.0 mg/L in 1995 to 0.9 mg/L in 2023, with individual concentrations occasionally exceeding the 1.24 mg/L criteria value (Fig. 3.6.4E). No trend was found for Chl-*a*, however, the trendline and many of the individual concentrations exceeded the 7.2 mg/m^3 criteria value (Fig. 3.6.4F).

For the **Tomoka River south station (27010579)**, positive trends were found for water temperature, pH, Secchi, TP and color and negative trends for DOC and TSS (Table 3.6.3). Water temperature increased from 20.5°C in 1998 to 23.0°C in 2023, whereas salinity remained near 0.16 from 2003 to 2023 (Fig. 3.6.5A-B). DO was stable across years, with most individual concentrations above 3.0 mg/L (Fig. 3.6.5C). For TP, concentrations increased from 0.04 mg/L in 1998 to 0.05 mg/L in 2023, with few concentrations exceeding the 0.12 mg/L criteria value (Fig. 3.6.5D). No change was found for TN or Chl-*a*, with TN concentrations occasionally exceeding the nutrient criteria value of 1.54 mg/L (Fig. 3.6.5E-F).

Positive trends in Secchi and color and negative trends in salinity, conductivity, DO, TP, TN, TSS and Chl-*a* were observed at the **Rose Bay station (NCBRB01)** (Table 3.6.3). Across the 18-year period of record (2005-2023) water temperature remained near 24.5°C and salinity decreased from 30.0 to 26.9 (Fig. 3.6.6A-B). For DO, concentrations decreased from 6.3 mg/L in 2005 to 5.8 mg/L in 2023, with most concentrations being above 3.0 mg/L (Fig. 3.6.6C). From 2005 to 2023, TP decreased from 0.14 mg/L to 0.09 mg/L and TN from 0.63 mg/L to 0.50 mg/L (Fig. 3.6.6D-E). Chl-*a* decreased from 9.7 mg/m^3 in 2005 to 6.4 mg/m^3 in 2023, with individual concentrations frequently exceeding the 11.0 mg/m^3 criteria value (Fig. 3.6.6F).

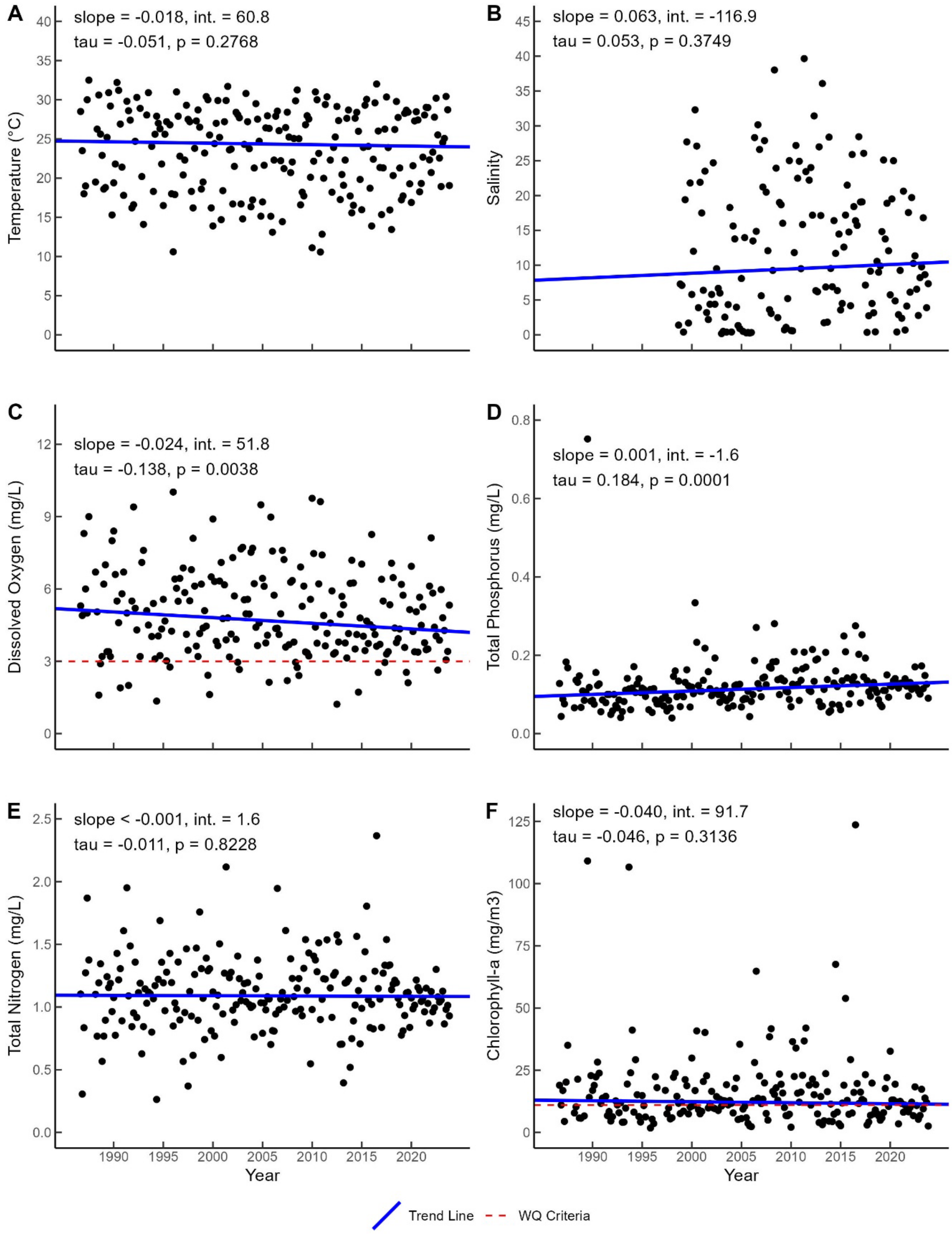

Figure 3.6.3. Seasonal Kendall trends for temperature, salinity, DO, TP, TN and Chl-*a* (A-F) at the Bulow Creek station (BUL). The solid blue line depicts the monotonic trend for each parameter, with p < 0.05 denoting significant trends. The dashed red line indicates WQ criteria based on the 3.0 mg/L hypoxia stress threshold for DO and annual geometric mean value for Chl-*a* outlined in F.A.C. Rule 62-303.353(2) and 62-303.450(1).

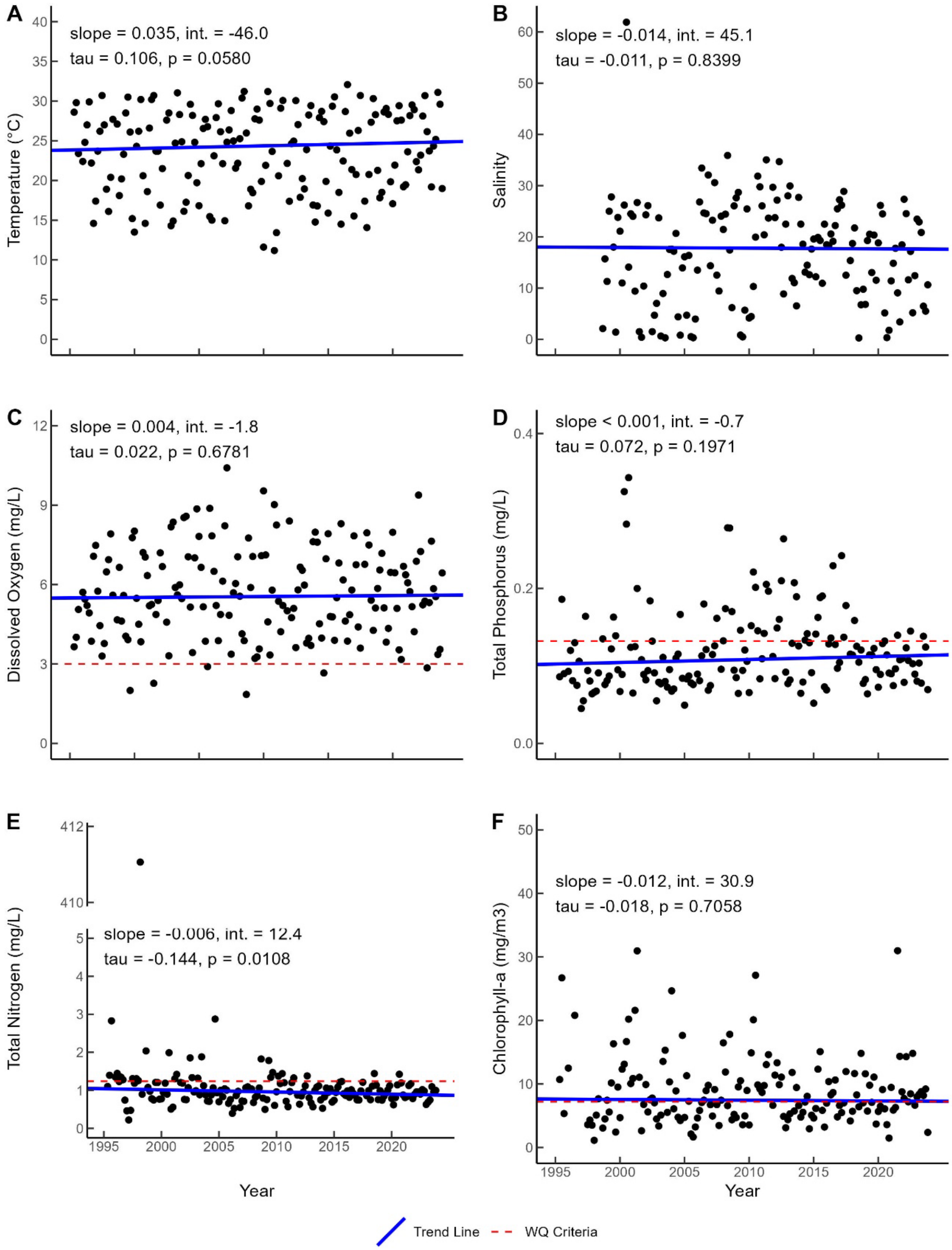

Figure 3.6.4. Seasonal Kendall trends for temperature, salinity, DO, TP, TN and Chl-*a* (A-F) at the northern Tomoka River station (27010024). The solid blue line depicts the monotonic trend for each parameter, with $p < 0.05$ denoting significant trends. The dashed red line indicates WQ criteria based on the 3.0 mg/L hypoxia stress threshold for DO and annual geometric mean values for TP, TN and Chl-*a* outlined in F.A.C. Rule 62-302.532.

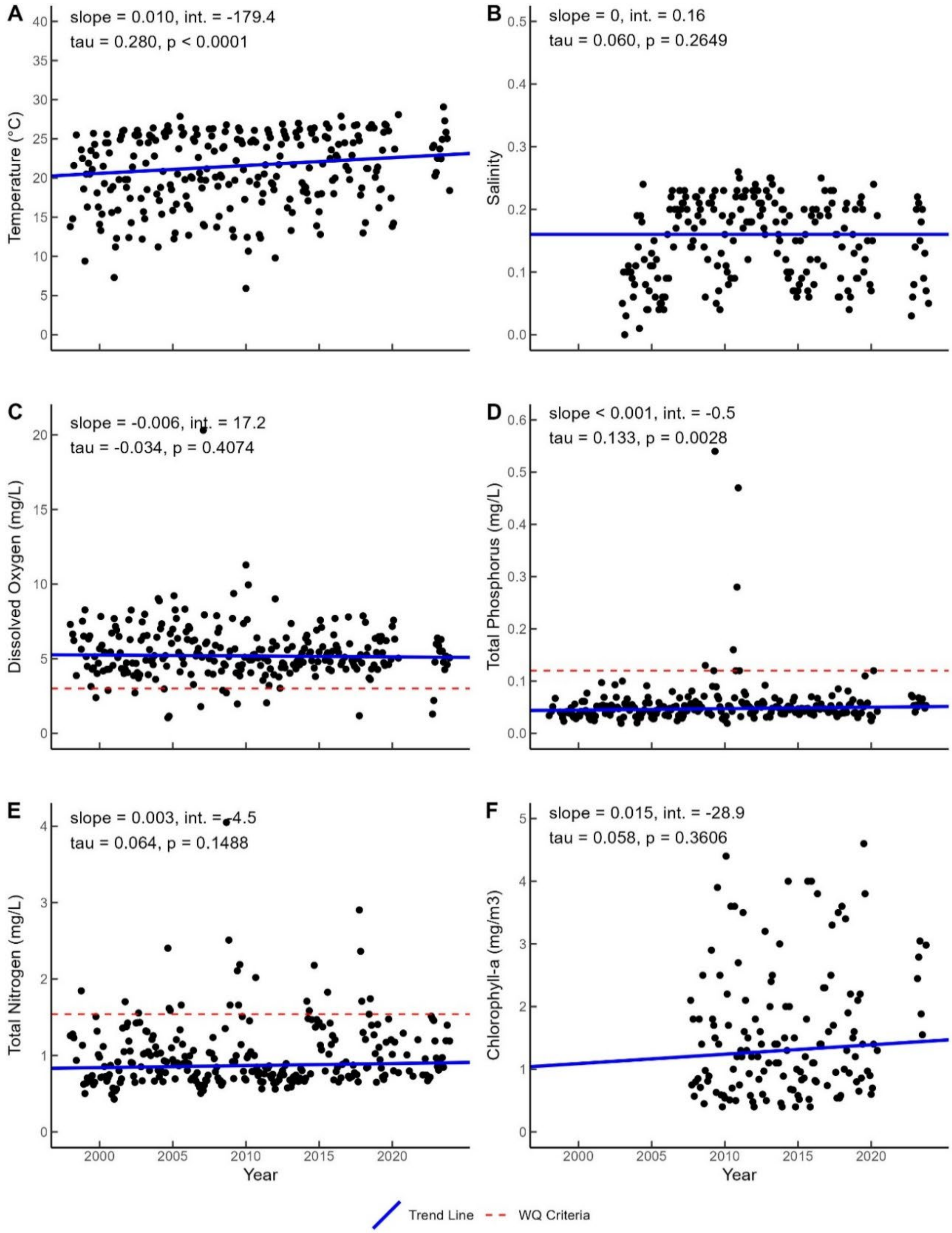

Figure 3.6.5. Seasonal Kendall trends for temperature, salinity, DO, TP, TN and Chl-*a* (A-F) at the southern Tomoka River station (27010579). The solid blue line depicts the monotonic trend for each parameter, with $p < 0.05$ denoting significant trends. The dashed red line indicates WQ criteria based on the 3.0 mg/L hypoxia stress threshold for DO and annual geometric mean values for TP and TN outlined in F.A.C. 62-302.531(2)(c). Streams do not have Chl-*a* criteria values.

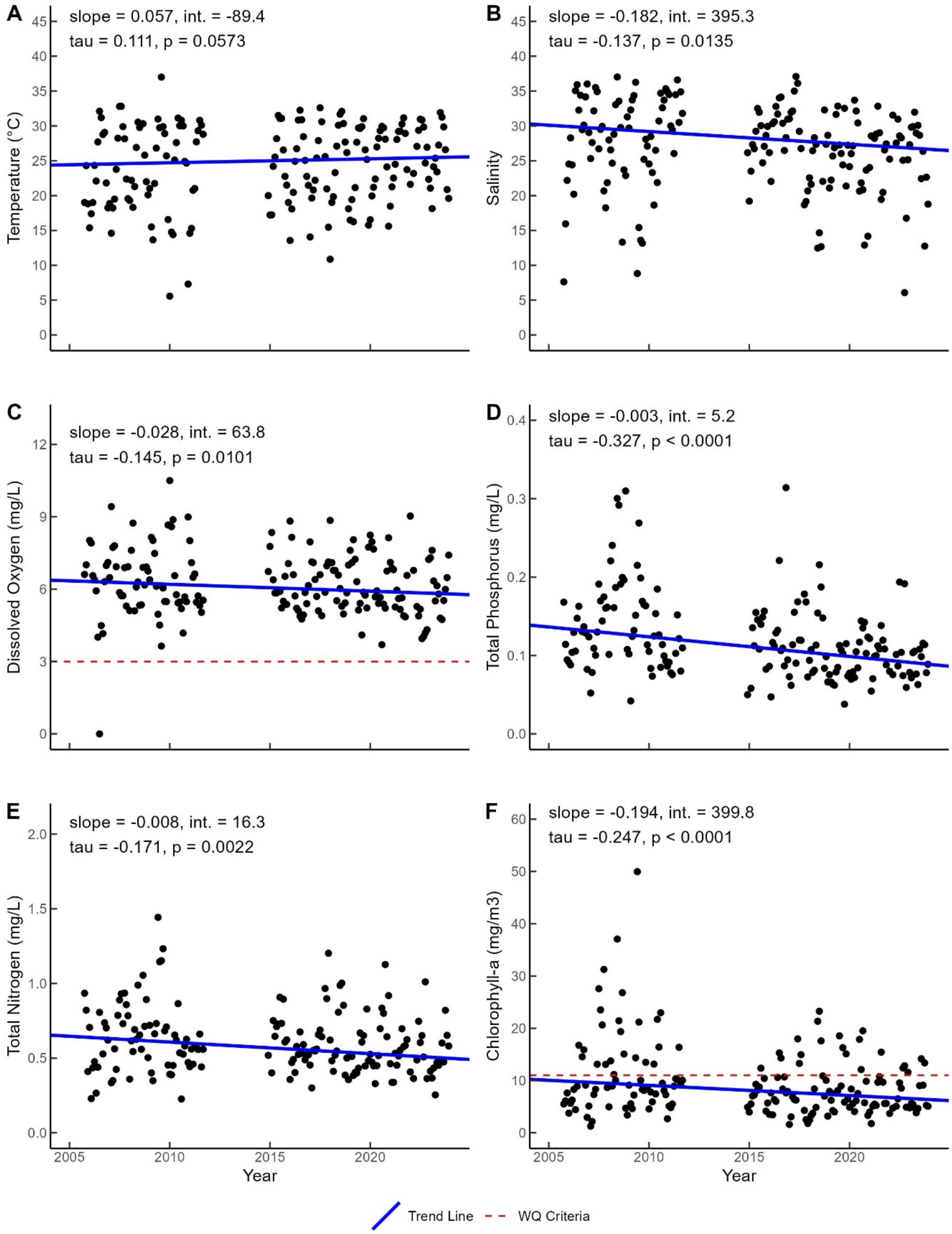

Figure 3.6.6. Seasonal Kendall trends for temperature, salinity, DO, TP, TN and Chl-*a* (A-F) at the Rose Bay station (NCBRB01). The solid blue line depicts the monotonic trend for each parameter, with $p < 0.05$ denoting significant trends. The dashed red line indicates WQ criteria based on the 3.0 mg/L hypoxia stress threshold for DO and annual geometric mean value for Chl-*a* outlined in F.A.C. Rule 62-303.353(2) and 62-303.450(1).

At the **Spruce Creek station (02248000)** increasing water temperature and conductivity and decreasing TP, TN and TSS were found (Table 3.6.3). Water temperature increased from 21.5°C in 1995 to 23.0°C in 2023, while salinity remained between 0.15 to 0.16 from 2002 to 2023 (Fig. 3.6.7A-B). Both the trendline and most individual concentrations for DO were above 3.0 mg/L (Fig. 3.6.7C). Across the 28-year period (1995-2023) TP decreased from 0.13 mg/L to 0.07 mg/L and TN decreased from 1.2 mg/L to 1.0 mg/L, with individual TP and TN concentrations periodically exceeding their respective nutrient criteria values (Fig. 3.6.7D-E). No change in Chl-*a* was found with most concentrations being below 10 mg/m^3 (Fig. 3.6.7F).

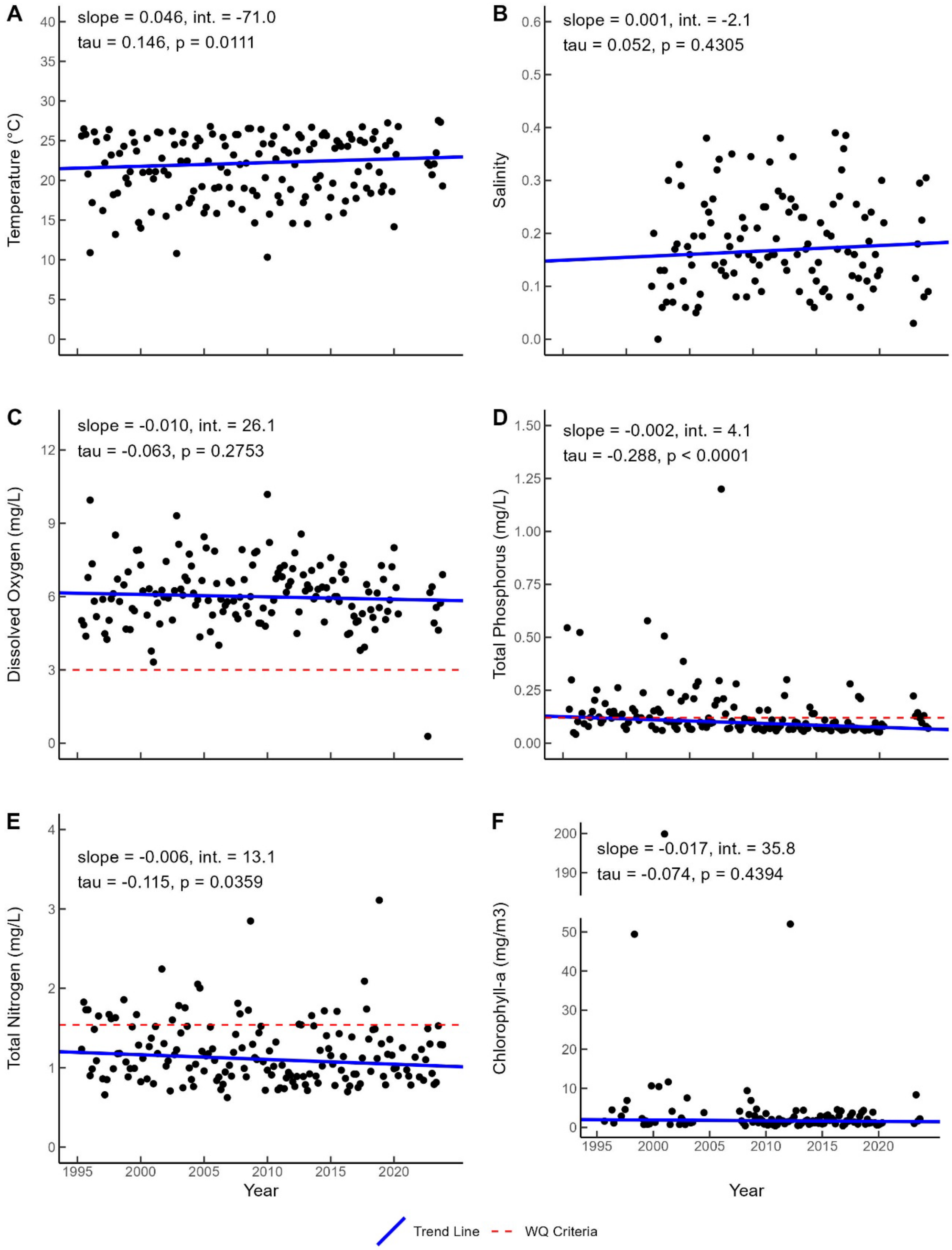

Figure 3.6.7. Seasonal Kendall trends for temperature, salinity, DO, TP, TN and Chl-*a* (A-F) at the Spruce Creek station (02248000). The solid blue line depicts the monotonic trend for each parameter, with $p < 0.05$ denoting significant trends. The dashed red line indicates WQ criteria based on the 3.0 mg/L hypoxia stress threshold for DO and annual geometric mean values for TP and TN outlined in F.A.C. 62-302.531(2)(c).

Discussion

Salinity within the HRPU is sensitive to rainfall and tidal mixing and can vary considerably both spatially and temporally (Dix et al. 2019, Cho et al. 2020). This is further supported by the data presented in this report, with salinity being lowest in tributaries and highest near inlets and the ICW, and salinity varying across months for all stations. Additionally, observations of low salinity and pH, and high DOC and color at the southern Tomoka River and Spruce Creek stations align with conditions reported for blackwater systems in Florida (Gallegos 2005, Blair et al. 2009, Williams and Kimball 2013, Chen et al. 2015, Chaya et al. 2023). High variability for DOC, color, and TSS was expected since rainfall, stormwater runoff, and freshwater inflow can add organic material or particulates into estuaries, which can lead to periodic increases in DOC, water color and suspended solids (FLW 2004, Paudel et al. 2019, Panton et al. 2020). Currents, vertical mixing of the water column, and human activities such as dredging can also resuspend particulates (Green and Coco, 2014, Wang et al. 2017) and contribute to variability in TSS.

Significant trends were found for three or more WQ parameters at each station, with changes being low (tau < 0.3) to moderate (0.3 ≤ tau < 0.5). Evidence of increasing water temperature within the HRPU is supported by other studies which report rising estuarine water temperature globally (Prum et al. 2024), including the eastern United States (Oczkowski et al. 2015, Mallick and Dunn 2024) and South Florida (Shi and Hu 2022, Shi et al. 2024). Increasing global and South Florida estuarine water temperatures have been related to increasing air temperatures (Prum et al. 2024, Shi et al. 2024), with the past several years (e.g., 2002-2023) being some of the hottest years on record for Florida (Powell, 2024). Long-term increases in water temperature are a management concern since temperature has a role in regulating biophysiochemical processes including metabolic rates, water-atmospheric gas exchange (e.g., carbon dioxide) and estuarine community dynamics (Mallick and Dunn 2024, Prum et al. 2024). Thus, warmer water temperatures may directly influence other WQ aspects such as lowering DO solubility (Chapra et al. 2021) and increasing phytoplankton production and biological oxygen demand (Harris et al. 2006, Prum et al. 2024). Additionally, warmer water temperatures, especially in the winter, may allow for the poleward expansion of subtropical estuarine species, which raises concerns over changes to biodiversity and species interactions (e.g., trophic webs) within these systems. Changes to biological communities in response to warming conditions is an important consideration for the HRPU since it overlaps with the tropical-temperate transition zone, with less cold tolerant species typically being found to the south and more cold tolerant species to the north (Kimball and Eash-Loucks 2021, Osland et al. 2021, Brewton and Lapointe 2023). Evidence of aquatic species range expansion has been reported for other regions in Florida which overlap with the tropical-temperate transition zone such as the Indian River Lagoon (Adams et al. 2024) and Cedar Key (Purtlebaugh et al. 2020).

Decreases in salinity at the ICW and Rose Bay stations is likely explained by changes in rainfall, water levels, and the frequency of episodic storm events (Schmidt and Luther 2002, Sumner and Belaineh 2005, Feher et al. 2023). The observed decreases remained within the reported salinity range for these regions (Cho et al. 2020) and gradual changes to salinity over time are not a primary management concern. Instead, management efforts focus on large, abrupt salinity shifts due to the potential adverse effects on aquatic organisms. For example, prolonged periods of low salinity (e.g., salinity < 5) can negatively impact the survival and growth of fish (Bachman and Rand, 2008) and oysters (La Peyre et al. 2013, Parker et al. 2013), and can alter oyster reef faunal communities (Tolley et al. 2005). In addition to the ICW and Rose Bay, an increase in conductivity was observed at Spruce Creek, however no coinciding increase in salinity was observed. Differences in the observed salinity and conductivity trends at Spruce Creek may be explained by differences in period of record as conductivity had an additional 7 years of data. Kendall trend and time series analyses are generally more effective with longer data records (Meals et al. 2011), as shorter periods may reduce the likelihood of detecting significant trends. This example emphasizes the importance of period of record and data collection consistency for WQ time series analyses.

Reductions in DO observed at several locations across the HRPU may be attributed to coinciding decreases in salinity in the ICW and Rose Bay, as freshwater generally has lower DO than saltwater in blackwater estuarine systems (Williams and Kimball 2013, Chen et al. 2015, Chaya et al. 2023). Despite declines in DO, the ICW and Rose Bay stations do not appear to be at risk of developing impairments. By contrast, the decline at Bulow Creek poses a management concern since this region (WBID 2620) is currently impaired for DO based on FDEP's 2022-2024 Comprehensive Verified List and 2022-2024 WQ Assessment, TMDLs, and BMAPs. However, a causative pollutant at this location has not yet been identified and is on FDEP's study list. In addition to Bulow Creek, other locations (WBIDS) within the region including creeks (2040), branches (2642, 2645), ditches (2647), canals (2664, 2666), and tributaries (2673) are currently impaired for DO. Spruce Creek (WBID 2674) also has a TMDL for DO and TP. Within estuarine systems, low DO concentrations can lead to fish kills and mortality of oyster reefs (Landsberg, et al. 2020, Coxe et al. 2023). Additionally, low DO concentrations may be a concern for biogeochemical processes within a system as DO can affect nitrogen cycling and sediment-water flux of nutrients and metals (Kemp and Dodds 2002, Skoog and Arias-Esquivel 2009).

Declines in nutrients (TP, TN and/or NOx-D) were found at several locations across the HRPU as well as decreasing Chl-*a* in Rose Bay. Improvements to eutrophication in the Rose Bay can be attributed to restoration efforts led by SJRWMD, Volusia County, the City of Port Orange, Florida Department of Transportation, and U.S. Army Corps of Engineers to improve WQ through reductions in stormwater runoff, removal of leaking septic tanks and accumulated, nutrient rich, muck at the bottom of the Bay, and the removal and replacement of an old causeway bridge which was restricting water circulation and flow (Cho et al. 2020).Other restoration initiatives throughout the HRPU, such as the Flager County Wetland Restoration Project, can also enhance WQ by reestablishing hydrological conditions that were altered by dragline ditching to their original state and improving coastal wetland function. Considering much of the Halifax River watershed is developed, nutrient loading from stormwater runoff presents a serious management concern since it has been identified as a major contributor to WQ degradation and eutrophication in urban environments (Yang and Lusk, 2018, Lusk et al. 2020). Multiple management efforts have been focused towards reducing nutrient pollution from stormwater runoff throughout the HRPU. For example, the City of South Daytona constructed a wet detention pond to detain and treat stormwater before it enters the Halifax River (Cho et al. 2020). Volusia County also initiated an assessment of stormwater infrastructure, operation, maintenance along the Halifax River to identify areas that needed improvement, and implemented best management practices, such as the Florida Friendly Fertilizer Use Ordinance, to reduce nonpoint source pollution (Goolsby et al. 2016). In addition to Spruce Creek, there are also nutrient TMDLs for TN in the Halifax River (WBID 2363B) and both TP and TN in the Tomoka River (WBID 2634), which aim to reduce nutrient loading from water treatment facilities (wastewater/demineralization concentrate discharge), stormwater, urban runoff, agriculture, and leaking wastewater collection systems and septic tanks (Cho et al. 2020).

However, despite declines in nutrients and Chl-*a*, regions with TMDLs are still impaired for these nutrients and there are additional areas (WBIDS) throughout the HRPU that have nutrient (2666, 2673, 2645) and Chl-*a* (2640, 2363J) impairments. Additionally, increasing TP was found for the Bulow Creek and southern Tomoka River stations, with the increase in the Tomoka River presenting a management concern since this WQ monitoring station overlaps with the nutrient impaired region. Currently, WQ data and impairments throughout the HRPU suggest further nutrient reductions are needed. Eutrophication can degrade WQ by increasing phytoplankton production, which may expose marine organisms to toxins, clog the gills of fish and invertebrates, elevate the risk of hypoxia, and lower light attenuation (Medina et al. 2025). It has also been linked to salt marsh loss (Deggan et al. 2012) and the accelerated expansion of mangrove range (Dangremond et al. 2020).

Overall, current impairments and observed declines in DO and increases in TP found at several stations show that improvements to WQ in the HRPU are needed to maintain a healthy ecosystem. While bacteria are not a focus of this report, both Spruce Creek and the Tomoka River have fecal coliform impairments (from anthropogenic sources), which are used as an indicator of pathogenic microorganisms (Ohrel and Register 2006). This presents a concern for Class III waters due to potential human health risks from pathogen exposure during recreational activities (McKee and Cruz 2021, Rhoden et al. 2021). Evidence of increasing water temperature also poses a management concern as water temperature can directly influence biogeochemical processes (Mallick and Dunn 2024, Prum et al. 2024), which can have direct effects on WQ, and it may lead to changes in estuarine community dynamics (Purtlebaugh et al. 2020, Adams et al. 2024). Reducing excess nutrients throughout the system is also a management priority since eutrophication can degrade WQ, harm aquatic life (Medina et al. 2025), and alter coastal habitat structure (Deegan et al. 2012, Dangremond et al. 2020).

CHAPTER 4: CONCLUSION AND RECOMMENDATIONS

The NCB is a dynamic system with differences in hydrology, land use, habitat and WQ concerns along the latitudinal gradient from Nassau County southward to the Ponce Inlet in Volusia County. Estuaries in the northern NCB (e.g., St. Marys, Nassau) have large tidal amplitudes, which increases flushing and lowers water residence time (Uncles et al. 2002) and are less developed than those in the southern NCB (e.g., Halifax). Estuarine systems with developed watersheds are at greater risk of WQ impairments due to loss of natural habitat and subsequent ecosystem services, and other anthropogenic stressors (e.g., dredging, hydrological alterations, nutrient loads) that can degrade WQ (Freeman et al. 2019). However, high tidal exchange rates across the NCB are an important consideration for WQ since they can have dampening effects on nutrients and phytoplankton biomass (Paerl et al. 2014, Hart et al. 2015), and can potentially alleviate the effects of eutrophication. The northern NCB also has greater oyster reef densities and is characterized by saltmarsh habitat, whereas estuaries in the southern NCB have less oyster cover and feature a mix of mangrove and salt marsh habitat (Dix et al. 2019, Simpson et al. 2019, Garvis et al. 2020, Dix et al. 2021). Poor WQ (e.g., eutrophication), hydrologic alterations (e.g., hardened shorelines), changes in precipitation (e.g., freshwater pulses), erosion, sea level rise (e.g., saltwater intrusion), boating traffic (e.g., boat wake), elevated water temperatures, ocean acidification and overharvesting threaten oyster populations and/or coastal habitat (Lotze et al. 2006, Beck et al. 2011, Dix et al. 2019, Knoell et al. 2021, Kimmel et al. 2024). Maintaining optimal WQ conditions to support healthy oysters and coastal habitats is a core objective of NCB WQ management. With Nassau, Duval, St. Johns, Flagler, and Volusia Counties projected to grow by nearly 670,000 people over the next 25 years (Rayer and Wang, 2024), monitoring and analyzing WQ parameters (e.g., nutrients, DO, etc.) is imperative to identifying potential threats and developing management and resiliency strategies to maintain and protect coastal waters. This report aims to evaluate long-term NCB WQ trends and address key management concerns.

Monthly Variability

Of the thirteen WQ parameters analyzed, many exhibited seasonal variability consistent with findings from previous studies in the region (Williams and Kimball, 2013, Hart et al. 2015, Bielmyer-Fraser et al. 2020, Cho et al. 2020, Chaya et al. 2023). In northeast Florida, estuarine water temperatures are lowest in the winter (December – February) and highest in the summer (June – September), whereas salinity and conductivity peak during the dry season (October – May) and decline during the wet season (June – September) due to increased freshwater input (Coffin et al., 1992, Williams and Kimball 2013, Hart et al. 2015, Bielmyer-Fraser et al. 2020). Both water temperature and salinity vary spatially, influenced by estuarine hydraulics, freshwater hydrology and tidal exchange (Kurylyk and Smith 2023). For example, salinities in the GTM estuary are highest near inlets, and summer water temperatures in these areas tend to be cooler than those farther from inlets (Hart et al. 2015).

WQ parameters are often interrelated, with changes in one parameter influencing others. Patterns and correlations commonly emerge among DO, pH, DOM, nutrients and Chl-*a* in response to seasonal and non-seasonal shifts in water temperature and salinity. In the SJR and estuary, warmer summer water temperatures are negatively correlated with DO and positively correlated with Chl-*a* (Bielmyer-Fraser et al. 2020, Wang and Zhang, 2020), due to reduced solubility of DO and increased metabolic rates (Chapra et al. 2021, Prum et al. 2024). In the GTM estuary, salinity is positively correlated with DO and pH but negatively correlated with nutrients and florescent DOM with nutrients and Chl-*a* concentrations being highest during the wet season (Hart et al. 2015, Chaya et al. 2023). Understanding these interrelationships and their connections to climatic patterns, hydrological processes, and land use is essential for interpreting WQ trends and identifying management priorities across the NCB.

Water Temperature Trends

Increases in water temperature were found for 5 out of the 8 NCB PUs, consistent with global estuarine trends (Prum et al. 2024), as well as patterns in the eastern United States (Oczkowski et al. 2015, Mallick and Dunn 2024) and South Florida (Shi and Hu 2022, Shi et al. 2024). Rising estuarine water temperatures have primarily been correlated with increasing air temperatures (Prum et al. 2024, Shi et al. 2024). Notably, 2022 and 2023 were among the hottest years on record in Florida (Powell, 2024) with South Florida estuaries exhibiting accelerated warming rates, which in some cases surpass those of adjacent ocean waters (Shi et al. 2022, Shi et al. 2024). While air temperature is a key driver, estuarine warming rates can also be affected by landscape characteristics (e.g., vegetation cover) and land use, tidal exchange, and estuary characteristics (e.g., water depth, surface area) (Rice and Jastram 2015, Shi et al. 2022, Itsukushima et al 2024).

Water temperature is a critical component to estuarine ecosystem function and can have direct effects on WQ through changes in water-atmospheric exchange of gases (e.g., O_2, CO_2), metabolic rates (e.g., primary productivity, biological oxygen demand), and eutrophication potential (Nazari-Sharabian et al. 2018, Kurylyk and Smith 2023). Rising water temperatures are a management concern as they are expected to reduce DO concentrations and increase Chl-*a* and eutrophication risk. Throughout the NCB, every PU exhibited at least one WQ impairment related to DO, Chl-*a*, and/or nutrients, though not all PUs were impaired for all three parameters. Warming water temperatures may further exacerbate DO impairments by reducing oxygen solubility (Chapra et al. 2021) and elevate metabolic processes, including biological oxygen demand (Rubalcaba et al. 2020) and phytoplankton growth (Fernández-González et al. 2022), heightening the risk of hypoxia and blooms, particularly in summer. Additionally, warming accelerates microbial activity, enhancing nutrient release from sediments (and other sources), and increasing nutrient diffusion through reduced water viscosity (Nazari-Sharabian et al. 2018), thereby elevating the risk of nutrient and Chl-*a* impairments.

Warmer water temperatures can also directly affect estuarine community dynamics by increasing the spread of disease, altering growth and survivorship rates, and facilitating the poleward expansion of subtropical species (Okon et al. 2023, Adams et al. 2024). Increasing temperature promotes the spread of bacteria and viruses, increasing infection risk among fish, shellfish and other marine animals, with potential impacts to commercial and recreational fisheries (Okon et al. 2023, Neokye et al. 2024, Rowley et al. 2024). This is especially concerning for NCB PUs impaired for fecal coliforms, which are indicators of pathogens (Ohrel and Register 2006) and pose health risks for Class II (shellfish harvest) and Class III (recreational) waters (McKee and Cruz 2021, Rhoden et al. 2021). Additionally, prolonged exposure to elevated temperatures can increase stress, mortality, and growth deformities in oyster spat and reduces reproduction while increasing disease susceptibility of adult oysters (Okon et al. 2023, Neokye et al. 2024). Milder winter temperatures can also facilitate the range expansion of marine species such as the western Atlantic sea bream (*Archosargus rhomboidalis*) in the Indian River Lagoon (Adams et al. 2024) and common snook (*Centropomus undecimalis*) in Cedar Key (Purtlebaugh et al. 2020), raising management concerns over shifts in biodiversity and trophic interactions. Currently, there is a growing need for comprehensive, long-term monitoring and research to assess the ecological impacts of warming on WQ, (Kurylyk and Smith 2023, Itsukushima et al 2024, Prum et al. 2024), biodiversity, and habitat structure (e.g., oyster reefs).

Salinity Trends

Salinity trends across the NCB PUs were variable, reflecting both increases and decreases. Similar non-uniform trends have been reported for southeastern United States estuaries, which were attributed to local drivers (Mallick and Dunn 2024). Salinity is influenced by hydrology, freshwater inflow, precipitation, evapotranspiration, tidal mixing, and major storms (e.g., hurricanes) (Schmidt and Luther 2002, Sumner and Belaineh 2005, Feher et al. 2023) as well as when monitoring occurs within tidal cycles. While small shifts in salinity are typically not a management concern, abrupt shifts (e.g., freshwater pulses) can significantly impact the survival and development of fish (Bachman and Rand, 2008), oyster recruitment, survival and growth (La Peyre et al. 2013, Parker et al. 2013), and reduce faunal community abundance, biomass and diversity in oyster reefs (Tolley et al. 2005, Marshall et al. 2019). Although long-term variations in NCB salinity have not yet shown significant biological implications, emerging trends may signal broader shifts in precipitation and sea level dynamics.

Florida is projected to experience more frequent and intense precipitation events over the next century, raising WQ management concerns due to increased erosion, sediment transport, and nutrient loading into coastal systems (Nazari-Sharabian et al. 2018, Haque 2023). These changes may accelerate coastal habitat loss and WQ degradation through reduced ecosystem services (Barbier et al. 2011, Blair et al. 2015). Degraded WQ through increased turbidity and eutrophication can subsequently lower oyster filtration, recruitment, and growth, and increase oyster mortality (Huang et al. 2016, Neokye et al. 2024), as well as elevate the risk of harmful algal blooms and hypoxia (Haque 2023). However, factors like flow rate, tidal flushing, water residence time, and water turnover rate can mitigate these impacts by flushing pollutants and nutrients from the system (Su et al. 2004, Paerl et al. 2014, Krause et al. 2022). For example, in the GTM estuary, peak Chl-*a* concentrations dropped following high rainfall events, suggesting increased flushing and reduced residence time contributed in lowering Chl-*a*, despite increased nutrient inputs (Hart et al. 2015).

Increasing salinity may also be an indicator of sea level rise, which is another management priority due to its potential to alter estuarine hydrology, degrade WQ, and accelerate the loss of coastal habitats and oyster reefs (Dix et al. 2019, Dix et al. 2021, Haque 2023). Higher salinities also raise the risk of *Perkinsus marinus* infections in oysters, which, when coupled with elevated water temperature, can significantly increase disease-related oyster mortality (Petes et al. 2012). Currently, limited data exists on how shifts in precipitation and sea level dynamics are influencing WQ in the NCB, highlighting the need for hydrological modeling (e.g., salinity) and statistical analyses to assess the influence of rainfall and other hydrological variables (e.g., flow rates, water residence time) on key WQ parameters. Additionally, more information is needed on optimal salinity ranges for NCB oyster populations in order to evaluate whether salinity trends are affecting their health and habitat suitability.

pH Trends

Spatially variable increases and decreases in pH were found across the NCB PUs, influenced by salinity and Chl-*a* at some locations. In blackwater estuarine systems, lower salinity water is typically more acidic compared to more marine regions (Gallegos 2005, Blair et al. 2009), with salinity positively correlated with pH in the GTM estuary (Chen et al. 2015). Phytoplankton also impact pH through CO_2 absorption with positive correlations observed between phytoplankton biomass and pH (Carstensen and Duarte 2019, Raven et al. 2020, Shen et al. 2020, Hall et al. 2023). However, changes in pH may also be related to other mechanisms not assessed in this report. For example, decreases in pH linked to rising atmospheric CO_2 concentrations and ocean acidification have been reported for eight Florida estuaries (Robbins and Lisle, 2018). Although long-term pH changes in the NCB were small and fell within typical estuarine ranges (Ohrel and Register 2006), declines in pH can add stress to biological communities (Kennish 2002, Sheldon and Alber, 2011). This presents a significant management concern for oysters and other shellfish, as lower pH can negatively affect growth, development, feeding rates, nutrient absorption, and reproduction and increase shell dissolution rates (Waldbusser et al. 2011a, Waldbusser et al. 2011b, Neokye et al. 2024). More research is needed to assess whether these pH shifts are impacting shellfish populations in the NCB.

Dissolved Oxygen Trends

Similar to other WQ parameters, DO concentrations varied across the NCB, often linked to changes in salinity or Chl-*a*. Increases in salinity can elevate DO concentrations, with positive correlations observed in the GTM estuary (Chen et al. 2015). Phytoplankton blooms, while enhancing surface water DO through photosynthesis, can also increase the risk of hypoxia, especially in bottom waters, due to elevated biological oxygen demand from microbial decomposition. Declining surface DO has been reported in estuaries with reduced primary productivity (Langendorf et al. 2021, Wang et al. 2024). Management strategies targeting eutrophication and DO impairments must account for the decoupled effects of surface oxygen production versus bottom water oxygen consumption (Langendorf et al. 2021). DO impairments remain a management concern across the NCB, as low concentrations can be a stressor for aquatic animals, increasing fish and oyster mortality (Landsberg et al. 2020, Coxe et al. 2023). Low DO also promotes the release of additional nutrients (and pollutants) from sediments, and affects biogeochemical processes (e.g., nitrogen cycling) and sediment-water fluxes of nutrients and heavy metals (Kemp and Dodds 2002, Skoog and Arias-Esquivel 2009, Foster and Fulweiler 2019), further degrading WQ.

Nutrient and Chlorophyll-*a* Trends

Nutrient declines were generally observed throughout the NCB, with some localized increases in TP or TN. In contrast, Chl-*a* declines were more prevalent in the northern NCB (LSMPU, NRPU, ICWPUs) and increases in southern NCB (MRPU, PCPU). Nutrient reductions may be attributed to water management efforts, including TMDL implementation wastewater treatment upgrades septic-to-sewer conversions, stormwater system retrofits, and implementation of best management practices to reduce nonpoint source pollution. Despite these improvements, many areas across the NCB remain impaired for nutrients and Chl-*a*. Some WQ monitoring stations with stable or decreasing trends still exceeded nutrient criteria for TP, TN, or Chl-*a*. While impairment determinations by FDEP are based on annual geometric means across multiple stations, evaluating individual station trends and exceedances relative to nutrient criteria supports the identification of nutrient hotspots and regions which may be at risk of future impairments. The data presented in the report and FDEP identified impairments indicate that further nutrient reductions are needed throughout the NCB.

Eutrophication is a management priority due to its direct impacts on WQ, including increased phytoplankton (Chl-*a*) and indirect effects such as hypoxia, algal toxins, reduced light attenuation, aquatic animal mortality, and habitat change (Medina et al. 2025). Harmful algal blooms can cause low DO, increasing respiratory stress and mortality in oysters (Kellogg et al. 2014), fish, and other marine animals. In oysters, harmful algal blooms can reduce feeding efficiency, promote shell fouling, increase susceptibility to disease and predation, disrupt spawning and reproduction, and expose oyster spat to toxins (e.g., saxitoxins, domoic acids) which can lead to health issues (e.g., lower growth) and impair swimming (Neokye et al.

2024). Furthermore, toxin exposure also poses health risks to humans consuming affected oysters (James et al. 2010). Eutrophication is further linked to coastal habitat changes, contributing to salt marsh loss and degradation (Deegan et al. 2012), increased mangrove freeze resistance (Feller et al. 2023), and accelerated mangrove range-expansion in the NCB's tropical-temperate transition zone (Dangremond et al. 2020).

Trend Analysis Conclusions

Overall, analysis of long-term WQ trends combined with FDEP impairments presents management concerns related to i) increasing water temperature, ii) changes in salinity, iii) decreasing pH and DO, and iv) increases and exceedances in nutrients and Chl-*a*.

Management priorities across the NCB PUs reflect differences in ecological conditions and anthropogenic influences. Basin-wide investigations are needed to assess the impacts of rising water temperature and changes in salinity, pH, and DO on WQ, oysters, and coastal habitats. Improved understanding of estuarine ecosystem responses to long-term temperature increases is critical (Kurylyk and Smith 2023, Itsukushima et al 2024, Prum et al. 2024), as warming is expected to reduce DO, increase primary productivity and exacerbate eutrophication, potentially altering estuarine community dynamics. Furthermore, additional research is necessary to evaluate how shifts in salinity, pH, and DO influence oyster reef habitat suitability throughout the NCB. Maintaining WQ conditions that support the health and sustainability of oyster populations and surrounding habitats is a key management objective for the NCB.

Eutrophication concerns are more pronounced in the southern NCB, where watershed development is extensive. While effects of eutrophication on WQ and aquatic organisms are well documented, its impacts on coastal habitats (e.g., salt marsh, mangroves) remains comparatively understudied. Incorporating research or monitoring to investigate eutrophication driven habitat change would provide valuable insights, as shifts or losses in these habitats can degrade ecosystem services and WQ.

To address these concerns effectively, the NCB's WQ monitoring network should be evaluated to incorporate data from external agencies and identify data deficient regions. This would improve the alignment between monitoring efforts and management goals, enable more robust data analysis of WQ drivers, and support the development of targeted programs focused on oyster populations and coastal habitat dynamics in response to WQ stressors. The next section outlines key management recommendations focused on strengthening the WQ monitoring network, improving data integration, and addressing critical knowledge gaps.

4.1. Management Recommendations

Effective management of WQ and coastal habitats in the NCB requires robust and comprehensive monitoring to identify trends, assess impacts, and guide mitigation strategies. Current monitoring efforts, while valuable, reveal gaps that limit the ability to fully understand ecosystem responses to stressors like rising temperatures, shifting salinity, eutrophication, and habitat loss. To address these gaps and enhance decision-making, three primary recommendations are proposed:

i) The identification of data deficient regions, incorporation of supplemental data from external agencies, and, where feasible, implement additional WQ monitoring stations.

ii) Enhance analytical capacity by conducting regionally explicit exploratory analyses to identify mechanisms driving WQ trends.

iii) Implementation of oyster and mangrove mapping and monitoring efforts, to better understand how WQ is affecting our coastal ecosystems.

The below recommendations aim to fill critical data gaps, improve our understanding of ecosystem dynamics, and enhance our capacity to manage WQ and habitat health. Enhanced monitoring and targeted analyses will allow for proactive management strategies, better resilience planning, and more effective protection of the NCB's valuable coastal ecosystems.

Identifying Data Deficiencies and Expanding Monitoring Coverage

Effective environmental monitoring programs require continuity, longevity and geographic scale to accurately detect trends and inform management decisions (Biber 2013). Gaps in WQ data can skew statistical analyses and short data records can make distinguishing actual trends from temporal variability difficult (Meals 2011, Biber 2013). A minimum of five years of consistently collected WQ data is needed for robust time series trend analyses (Meals 2011). Additionally, limited geographic scale reduces statistical power and hinders the ability to effectively address broader management needs and concerns (Biber 2013). For example, the intended management focus of the NCB ICWPUs is to monitor WQ within the ICW and surrounding tidal creeks (e.g., Sisters Creek, Clapboard Creek); however, there are currently no stations within these creeks, and each PU is limited to a single station. The inclusion of additional WQ stations within the ICWPUs would be valuable for addressing existing data gaps, supporting the intended management objectives of these PUs, and enhancing the robustness of regional-scale WQ trend assessments. Similarly, mesohaline zones of the LSMPU and PCPU would benefit from expanded monitoring to capture data on nutrient loading and other WQ parameters, which are often masked by tidal flushing (Coffin and Hampson 1992, Chaya et al. 2023).

However, before expanding the WQ monitoring network, it is essential to identify external agencies and their existing WQ stations to prevent redundancies. Furthermore, collaboration with these local stakeholders can offer valuable insights into their management needs and priorities, highlight opportunities for collaboration, and leverage ongoing efforts by SJRWMD through the alignment of shared goals. For example, the GTMNERR technical advisory group, which includes expert scientists and managers, has identified critical gaps in WQ monitoring in the Matanzas River (Salt Run), Moultrie Creek and Pellicer Creek based on current WQ, oyster habitat, and recreation concerns in these areas. Overall, expansion of the current WQ monitoring network through additional monitoring stations will:

i) **Fill Data Gaps**: Provides critical WQ data in previously under-monitored areas, addresses informational needs identified by resource managers and stakeholders, and establishes a baseline for assessing impacts of future coastal habitat or land use changes (e.g., urban development) on WQ.

ii) **Enhance Statistical Analyses:** Additional spatial and temporal data improves the reliability of statistical analyses and modeling, providing a stronger foundation for informed, data driven management decisions.

Enhancing Analytical Capacity with Additional Water Quality Analyses

The analyses conducted in this report were designed to evaluate long-term WQ trends at individual monitoring stations using all available data. This effort marks the first synthesis of long-term trends across the NCB since the establishment of the upper PUs in 2014. While these findings provide valuable insights, further refinement to analyses and investigation into the mechanisms driving WQ trends is necessary to support the development of robust, actionable management recommendations.

To effectively inform management strategies, WQ conditions must be evaluated at both local and regional scales. Robust interpretation of regional trends requires temporally consistent datasets across monitoring stations, and the use of the most comprehensive dataset available. Inconsistent monitoring records among stations limit spatial trend comparisons, as differences among trends may reflect location or period of record rather than true environmental changes. For robust spatial comparisons across NCB PUs, all stations should share a consistent monitoring time frame. Additionally, the analyses reported herein were limited to SJRWMD monitoring data. Incorporating WQ data from other agencies would enhance spatial analyses and improve the interpretation of WQ trends and management concerns across the region.

Mann-Kendall and Seasonal Kendall tests were used in this report and while useful, have notable constraints since they: i) require balanced datasets (e.g., no large gaps or systematic inconsistencies), ii) cannot incorporate additional explanatory variables, iii) assume seasonal cycles are stationery with consistent timing and magnitude across years, and iv) assess monotonic trends across time (Beck et al. 2022, Medina et al. 2025). However, WQ data are often non-monotonic and influenced by multiple driving forces that cannot be adequately captured or incorporated into Kendall tests (Murphy et al. 2019). To address these complexities, advanced analytical methods such as generalized additive models (GAM) and mixed-effects models are recommended, as they accommodate multiple independent variables, detect trends across temporal scales, and remain robust to uncertainty, missing data, and uneven sampling (Murphy et al. 2019, Beck et al. 2022, Medina et al. 2025). Incorporating these approaches would provide a more nuanced understanding of nutrient dynamics, Chl-*a* variability, and the underlying drivers of WQ trends in the NCB. Therefore, to support effective management decisions, it is recommended to incorporate analyses that:

i) **Use a Consistent Temporal Scale:** Enables spatial comparisons of WQ trends, without the results being skewed by differences in monitoring time frames across locations.

ii) **Utilize All Available Data:** Incorporating data from external agencies can fill data gaps, enhance statistical analysis, and improve understanding of WQ trends at local and regional scales.

iii) **Allow for Non-Monotonic Trends and Multiple Explanatory Factors:** Detects shifts in WQ trends across different temporal scales, which can enhance evaluations of the effectiveness of various management strategies (e.g., TMDLs), and improves identification of key drivers and interactions influencing WQ.

Coastal Habitat Research and Monitoring Needs

Additional research and monitoring are needed to determine how shifts in WQ parameters impact oyster populations and coastal habitats in the NCB. Both WQ and coastal habitats are interconnected, such that salt marsh, mangroves, and oyster reefs store carbon, remove nutrients from the water column, and stabilize shorelines (Kellogg et al. 2014, Volety et al. 2014, Garvis et al. 2020). However, poor WQ can stress and degrade these coastal habitats, leading to salt marsh loss, mangrove decline, and oyster reef degradation (Deegan et al. 2012, Feller et al. 2015, MacDonnell et al. 2017, Dix et al. 2019). For example, eutrophication, rising water temperatures, and declines in DO and pH pose management concerns for oyster reef sustainability. Further research is needed to evaluate whether these WQ changes are affecting oyster reef habitat suitability across the NCB. Additionally, the shift from salt marsh to mangrove dominance in the southern NCB PUs has implications for ecosystem services, as mangroves and salt marshes differ in their capacity for coastal protection, erosion control, water purification, and carbon sequestration (Barbier et al. 2011). Understanding how this habitat transition affects sediment retention, nutrient uptake, water clarity, and eutrophication is crucial, as these changes can have cascading effects on oysters, fisheries and other habitat dependent species (e.g., birds). The sections below outline how enhancements to oyster monitoring efforts and the integration of mangrove mapping and monitoring will address data gaps and improve coastal ecosystem understanding and management.

Oyster Mapping and Monitoring

Oysters provide important and valuable ecosystem services including enhancing WQ through denitrification, improving water clarity, sequestering carbon, reducing coastal erosion, providing food and habitat for estuarine animals, and supporting high levels of biodiversity due to their three-dimensional structure (Kellogg et al. 2014, Volety et al. 2014, Garvis et al. 2020). Despite these benefits, shellfish reefs are declining rapidly worldwide (Beck et al. 2011) due to habitat destruction, disturbance, disease, overharvesting, and reduced WQ (Knoell et al. 2021, Kimmel et al. 2024). Maintaining WQ conditions which support healthy oyster populations is a fundamental objective of NCB WQ management, particularly in Class II waters and designated shellfish harvesting areas throughout the NCB.

In 2015, SJRWMD and UCF mapped oysters throughout the NCB to create a baseline for oyster habitat within the region to be later used to study potential changes to reef area coverage over time. One of the recommendations from this study, and previous research, is to update mapping efforts every 10 years with possibly more frequent mapping efforts in high stress areas or locations with WQ impairments (Dix et al. 2019, Garvis et al. 2020). For example, areas with high boating activity, such as the ICW, have been linked to elevated shoreline erosion and oyster mortality (Wall et al. 2005, Garvis et al. 2015, Garvis et al. 2020, Pinton and Canestrelli 2024). Boat wakes can dislodge live oysters and relocate them to unsuitable habitats or smother reefs through sediment resuspension and deposition (Pinton and Canestrelli 2024), which increases oyster mortality and reduces ecosystem services and resiliency. In addition to enhancing WQ, oyster reefs protect salt marsh habitat from erosion through wave attenuation and shoreline stabilization (Garvis et al. 2020, Pinton and Canestrelli 2024) and under favorable conditions oysters can vertically accrete at rates sufficient to keep pace with sea level rise (Rodriguez et al. 2014, Walles et al. 2015). However, when reefs are degraded by boat wakes, overharvesting, or other stressors, the ability of oyster reefs to maintain elevation and protect salt marsh from erosion decreases. Sea level rise can further exacerbate shoreline erosion, as higher water levels allow for waves to pass more readily over low-profile reefs. The interaction between sea level rise and boat wakes can create a negative feedback loop that leads to further loss of oyster and salt marsh habitat and a reduction in ecosystem resiliency to environmental change. Therefore, understanding and reducing the effects of anthropogenic and environmental stressors, is a key management priority for protecting and maintaining heathy oyster populations.

Updating mapping and increasing monitoring efforts, particularly in high stress areas, can help track oyster reef responses to management efforts aimed at reducing threats such as boat wakes, disease, and poor WQ, while also identifying mechanisms that may positively affect oyster reefs. Furthermore, WQ data can be leveraged to better understand the effects of global threats (e.g., sea level rise, acidification, rising temperature, and changing precipitation patterns) to our oyster communities. Like WQ monitoring, oyster monitoring is essential for informing management strategies and oysters have been documented as bioindicators of WQ condition (Fehrenbach et al. 2025). Additional oyster reef monitoring and health assessments across the NCB can build upon ongoing efforts by FDACS to evaluate shellfish harvesting areas through the aligned objective of ensuring WQ conditions support heathy oyster populations and comply with regulatory standards. Updating oyster mapping throughout the NCB and increasing fine-scale monitoring efforts can fill knowledge gaps by:

i) **Documenting Long-Term Changes in Oyster Reef Coverage**: Provides critically needed up-to-date information on oyster reef cover which will be compared to previous basin-wide mapping efforts by SJRWMD and UCF.

ii) **Aiding Current Water Quality and Ecosystem Health Assessments:** Changes in oyster reef abundance can be used as an early bioindicator of degraded WQ conditions and for quantifying the overall health of coastal waters.

iii) **Enabling Habitat Suitability Modeling:** In addition to WQ, information on oyster condition and other biological factors (e.g., predation, competition) is needed for determining habitat suitability, which can be used to guide restoration and enhancement efforts.

iv) **Determining Impacts of Commercial and Recreational Harvest:** Evaluation of how harvesting influences oyster reef resilience (e.g., reef height, accretion rate, size structure), which is positively correlated with WQ maintenance.

Mangrove Mapping and Monitoring

Mangroves are highly productive ecosystems that occur in the intertidal zones of tropical and subtropical regions and can enhance WQ through processes such as wave attenuation, sediment deposition, and removal of nutrients and other pollutants (Barbier et al. 2011, Blair et al. 2015, Kelleway et al. 2017). The ecotonal boundary between mangroves and salt marshes in the United States has shifted between dominance at least six times between the late 1700s and 2017, resulting from decadal-scale fluctuations in the frequency and intensity of extreme cold events (Cavanaugh et al. 2019). This regime shift is currently occurring in salt marshes and oyster reefs along the Atlantic Coast of Florida (Simpson et al. 2019, Dix et al. 2021, Walters et al. 2024), with mangroves expanding northward into Georgia (Vervaeke et al. 2024). This expansion raises questions about changes in ecosystem service provisioning, particularly regarding nutrient cycling and WQ maintenance.

Shifts in dominant coastal habitats have important implications for WQ management, due to differences in biogeochemical cycling (e.g., enzyme activity, nitrogen mineralization, NH_4^+-N removal) between mangrove and saltmarsh habitats (Alongi 2020, Steinmuller et al. 2020, Wang et al. 2022, Wang et al. 2023), which can impact nutrient concentrations within coastal waters. For example, mangroves store more organic carbon, remove more nitrate (NO_3^-N), and release less methane (CH_4) compared to salt marshes. However, they also export more particulate organic carbon, DOC, dissolved inorganic carbon, and ammonium (NH_4^+-N) to adjacent waters (Alongi 2020, Wang et al. 2022, Wang et al. 2023). Therefore, mangrove expansion is expected to enhance nitrogen and carbon uptake, supported by evidence of increased aboveground carbon storage (Doughty et al. 2016) and sediment burial (Vaughn et al. 2020). While nutrient enrichment can accelerate mangrove range expansion (Dangremond et al. 2020) through increased aboveground growth and biomass (Weaver and Armitage 2020), and freeze resistance (Feller et al. 2023), it may also increase mangroves' vulnerability to environmental stress, increasing the risk of mortality (Feller et al. 2015, Lovelock et al. 2009) and release of stored nutrients (Kelleway et al. 2017, Mack et al. 2024). Additionally, shifts in dominant vegetation can alter estuarine aquatic community dynamics. In the GTM estuary, distinct differences in nekton assemblages have been observed among marsh dominated, mixed mangrove-marsh, and mangrove dominated habitats, likely driven by changes in food availability or predator-prey dynamics between habitats (Kimball and Eash-Loucks 2021). This is an important consideration as rising water temperatures combined with habitat alteration may produce synergistic effects that accelerate shifts in biological aquatic communities. Coastal habitat monitoring can provide valuable insights into spatiotemporal changes in WQ and biological communities, driven by habitat shifts and WQ-habitat interactions, as well as relationships between WQ and habitat health; thereby supporting more robust, data driven management strategies.

Given the documented shifts in coastal habitats and the ecological implications of mangrove expansion, there is a critical need to map mangrove recruitment across the NCB using high resolution satellite imagery. As the NCB is at the forefront of mangrove range expansion, understanding the impacts on habitat structure, ecosystem function, and corresponding services, such as biogeochemical cycling, WQ, and aquatic communities dynamics, is essential for effective management. Current Land Use Land Cover (LULC) maps lack the spatial and temporal resolution necessary to capture the fine-scale dynamics of this expansion (Bardou et al. 2023). High-resolution mapping would address this gap by:

i) **Documenting Habitat Expansion in Real-Time:** Providing up-to-date information on the rate and extent of mangrove encroachment.

ii) **Integrating with Global Mangrove Datasets:** Enhancing global assessments of mangrove distribution and climate resilience.

iii) **Forecasting Habitat Resilience:** Modeling how mangrove habitats, and subsequent WQ provisioning, may respond to climatic shifts and anthropogenic pressures.

iv) **Supporting Regional Coastal Monitoring Networks:** Enabling more robust analysis of past, present, and future coastal dynamics, particularly in relation to WQ and habitat health.

Management Recommendation Conclusions

To support effective, data driven ecosystem management and protect the intended uses of Class II and Class III waters throughout the NCB, there is a clear need to strengthen the current WQ monitoring network and coastal habitat connectivity. This includes:

i) **Identifying Data Deficiencies and Expanding Monitoring Coverage**: Establishing additional WQ monitoring stations in data deficient regions to fill critical gaps and improve the robustness of statistical analyses.

ii) **Enhancing Analytical Capacity:** Utilizing advanced analytical methods, such as GAMs, can improve interpretation of trends by accounting for non-monotonic trends across time and incorporating multiple explanatory variables. Ensuring temporal consistency can also enhance the spatial comparison of trends across local and regional scales.

iii) **Oyster Reef Mapping and Monitoring:** Producing up-to-date oyster habitat maps is essential for supporting WQ assessments, habitat suitability modeling, and ecosystem health evaluations. Increased monitoring in Class II waters will also improve understanding of how commercial and recreational harvest affects reef resilience and aid in management efforts that support sustainable fisheries and ecosystem integrity.

iv) **Mangrove Mapping and Monitoring:** Establishing systematic mangrove mapping will support real time tracking of habitat expansion, aid in assessing potential changes in WQ, and contribute to forecasting coastal habitat resilience. This effort will enhance broader coastal monitoring initiatives across the NCB.

Together these improvements will provide the foundation for more targeted and adaptive management strategies and promote long-term WQ health and ecosystem resilience.

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