

# Chapter 3: Water Quality in the Everglades Protection Area

Mailin Sotolongo Lopez<sup>1</sup>, Luke Hudson<sup>1</sup>, Grover G. Payne<sup>2</sup>, and Shi Kui Xue<sup>3</sup>



Water quality sampling in Water Conservation Area 2

## Highlights

Similar to previous water years, water quality in the Everglades Protection Area was in compliance with existing state water quality criteria during Water Year 2023 (May 1, 2022–April 30, 2023) with a few exceptions. Excursions of applicable Class III water quality criteria were observed for four parameters: dissolved oxygen, alkalinity, pH, and specific conductance. Similar to prior periods, these excursions were localized to specific areas of the Everglades Protection Area, and all these parameters exhibited excursions in past water years.

<sup>1</sup> Florida Department of Environmental Protection, Office of Water Policy and Ecosystems Restoration, Tallahassee, Florida.

<sup>2</sup> Florida Department of Environmental Protection, Division of Environmental Assessment and Restoration, Water Quality Standards Program, Tallahassee, Florida.

<sup>3</sup> South Florida Water Management District, West Palm Beach, Florida.

Approximately 98% of the interior Everglades Protection Area is below an annual geometric mean total phosphorus concentration of 15 micrograms per liter ( $\mu\text{g/L}$ ), and approximately 95% is below an annual geometric mean total phosphorus concentration of 10  $\mu\text{g/L}$ .

Similar to previous years' reporting, the five-year (Water Year 2019–Water Year 2023) results for the total phosphorus criterion assessment indicate that unimpacted portions of each Everglades Water Conservation Area passed all four parts of the compliance test. In Water Years 2022 and 2023, WCA2F3 and LOXA124 appear to have met the criteria to move from the impacted to unimpacted network. Additional analyses will be conducted to verify the change to unimpacted and will be included in future South Florida Environmental Reports.

Total phosphorus loads from surface sources, including internal transfers within the Everglades Protection Area, totaled approximately 99.1 metric tons (t) in Water Year 2023. This represents an increase of approximately 23% compared to Water Year 2022 (80.4 t) due to a 20% increase of flow. Of the 99.1 t of total phosphorus load, 68.4 t was from external surface water inflows to the Everglades Protection Area with a TP flow-weighted mean concentration of 27  $\mu\text{g/L}$ . Another 296 t of total phosphorus are estimated to have entered the Everglades Protection Area through atmospheric deposition.

---

## SUMMARY

---

This chapter is intended to (1) provide an assessment of water quality within the Everglades Protection Area (EPA) during Water Year 2023 (WY2023; May 1, 2022–April 30, 2023), (2) fulfill numerous reporting requirements of the Everglades Forever Act (EFA), (3) provide a preliminary assessment of total phosphorus (TP) criterion achievement, and (4) provide an annual update of the comprehensive overview of nitrogen and phosphorus concentrations and loads throughout the EPA. The information provided in this chapter is an update to Chapter 3 of the *2023 South Florida Environmental Report (SFER) – Volume I* (Sotolongo Lopez et al. 2023). Information on mercury in biota in the Everglades can be found in Appendix 3-2 and other annual permit reports in Volume III.

## WATER QUALITY CRITERIA EXCURSION ANALYSIS

The analyses and summaries presented provide a synoptic view of water quality conditions in the EPA on a regional scale, including the Arthur R. Marshall Loxahatchee National Wildlife Refuge (LNWR; also known as Water Conservation Area [WCA] 1), WCA-2, WCA-3, and Everglades National Park (ENP). For parameters with water quality criteria, regional analyses were conducted based on the frequency of exceedances of the applicable criteria, similar to the methods employed in the *1999 Everglades Interim Report, 2000–2004 Everglades Consolidated Reports, and 2005–2023 SFERs*<sup>4</sup>. For WY2023, water quality parameters that did not meet existing standards were classified based on excursion frequencies that were statistically tested using the binomial hypothesis test. These categories are (1) **concern** – any parameter with a criterion exceedance frequency statistically greater than 10%, (2) **potential concern** – any parameter with an exceedance frequency statistically greater than 5% but less than 10%, and (3) **minimal concern** – any parameter with an exceedance frequency less than 5% but greater than zero.

Similar to the previous water years, water quality was in compliance with existing state water quality criteria during WY2023 with a few exceptions. During WY2023, excursions of applicable Class III water quality criteria were observed for four parameters: dissolved oxygen (DO), alkalinity, pH, and specific conductance. Similar to prior periods, these excursions were localized to specific areas of the EPA, and all these parameters exhibited excursions in past water years.

---

<sup>4</sup> McCormick et al. 1999, 2000; Bechtel et al. 2000; Weaver et al. 2001, 2002, 2003, 2007, 2008, 2009; Weaver and Payne 2004, 2005, 2006; Payne et al. 2007, 2008, 2010, 2011; Payne and Xue 2012; Julian et al. 2013, 2014b, 2015b, 2016, 2017, 2018, 2019, 2020, 2021; Gilhooly et al. 2022; Sotolongo Lopez et al. 2023.

For WY2023, a summary of excursions of the DO, alkalinity, pH, and specific conductance criteria, as well as the status of pesticides, sulfate, phosphorus, and nitrogen within the EPA, are presented below:

- Due to excursions of the site-specific alternative criterion (SSAC), DO was classified as potential concern for the interior portions of WCA-3 and ENP, and concern for the interior portions of LNWR and WCA-2. DO SSAC excursions were observed at NE1 within ENP. Inflow, outflow, and rim canal monitoring locations were assessed using the current Class III water quality standard. DO was classified as concern for inflows to LNWR, WCA-2, WCA-3, and ENP. Rim canal portions of LNWR were classified as minimal concern, while outflow portions of WCA-2 were classified as potential concern, and WCA-3 was categorized as concern. DO in outflow from LNWR improved showing no exceedances for WY2023.
- Alkalinity was categorized as minimal concern for LNWR interior; however, the Florida Department of Environmental Protection (FDEP) considers the relatively low values to be representative of the range of natural conditions for this ecosystem.
- LNWR interior and WCA-3 inflow were categorized as minimal concern for exceedances of the pH criterion.
- Specific conductance was categorized as potential concern for the WCA-2 inflow, minimal concern for WCA-2 interior regions and LNWR rim canal, and concern for LNWR inflow.
- No exceedances of total ammonia nitrogen (TAN), turbidity, or total iron were observed in the EPA.
- In total, 18 pesticides or pesticide breakdown products were detected at concentrations above their respective method detection limits (MDLs) within the EPA, including dichlorophenoxyacetic acid (2,4-D), ametryn, atrazine, alpha-Hexachlorocyclohexane, bromacil, diazinon, dieldrin, diuron, hexazinone, imidacloprid, metalaxyl, metolachlor, metribuzin, norflurazon, prometryn, simazine, toxaphene and atrazine-desethyl. Toxaphene exceeded Class III water quality criteria.
- Annual mean inflow sulfate concentrations ranged from 64.9 milligrams per liter (mg/L) for LNWR inflows to 11.9 mg/L for WCA-3. The annual mean sulfate concentrations at interior marsh regions ranged from 42.5 mg/L for WCA-2 to 2.0 mg/L for ENP.
- Annual geometric mean inflow TP concentrations ranged from 4.6 micrograms per liter ( $\mu\text{g/L}$ ) for ENP to 28.2  $\mu\text{g/L}$  for WCA-3. Annual geometric mean TP concentrations at interior regions ranged from 4.6  $\mu\text{g/L}$  for ENP to 10.0  $\mu\text{g/L}$  for WCA-2. Annual geometric mean TP concentrations for individual interior marsh monitoring stations ranged from 2.75  $\mu\text{g/L}$  in some unimpacted portions of the marsh to 32.42  $\mu\text{g/L}$  in impacted portions of the marsh. Of the interior marsh station sites, 76% exhibited annual geometric mean TP concentrations of 10.0  $\mu\text{g/L}$  or less, while 83% had annual geometric mean TP concentrations of 15.0  $\mu\text{g/L}$  or less throughout the larger ambient monitoring network.
- Approximately 98% of the interior EPA area is below an annual geometric mean TP concentration of 15  $\mu\text{g/L}$ , and approximately 95 % is below an annual geometric mean TP concentration of 10  $\mu\text{g/L}$  (based on spatial cumulative distribution).
- Annual geometric mean inflow orthophosphate ( $\text{OPO}_4$ ) concentrations ranged from 1.1  $\mu\text{g/L}$  (ENP) to 4.7  $\mu\text{g/L}$  (LNWR). Annual geometric mean interior  $\text{OPO}_4$  concentrations were below 2.0  $\mu\text{g/L}$  for all areas of the EPA.
- Similar to previous years' reporting, the five-year (WY2019–WY2023) results for the TP criterion assessment indicate that unimpacted portions of each WCA passed all four parts of the compliance test. In WY2022 and WY2023, WCA2F3 and LOXA124 appear to have

met the criteria to move from the impacted to unimpacted network, respectively. Additional analyses will be conducted to verify the change to unimpacted and will be included in future SFER reports. In contrast and as expected, impacted portions of each water body failed one or more parts of the test. The impacted portions of the WCAs routinely exceeded the annual and five-year network TP concentration limits of 11 and 10 µg/L, respectively.

- TP loads from surface sources, including internal transfers within the EPA, totaled approximately 99.1 metric tons (t) in WY2023. This represents an increase of approximately 23% compared to WY2022 (80.4 t) due to a 20% increase of flow. Of the 99.1 t of TP load, 68.4 t was from external surface water inflows to the EPA with a TP flow-weighted mean concentration (FWMC) of 27 µg/L. Another 296 t of TP are estimated to have entered the EPA through atmospheric deposition (Redfield 2002).
- Annual geometric mean inflow total nitrogen (TN) concentrations ranged from 1.0 mg/L for ENP to 2.0 mg/L for LNWR. The annual geometric mean TN concentration at interior marsh regions ranged from 1.0 mg/L for ENP to 1.7 mg/L for LNWR.
- Where sampled, annual geometric mean inflow dissolved inorganic nitrogen (DIN) concentrations ranged from 0.05 mg/L for WCA-3 to 0.26 mg/L for LNWR. The annual geometric mean DIN concentration at interior marsh regions ranged from 0.02 mg/L for LNWR to 0.03 mg/L for WCA-2, WCA-3 and ENP.

---

## PURPOSE

---

The primary purpose of this chapter is to provide an assessment of water quality within the EPA during WY2023, and an update to the information provided in Chapter 3 of the 2023 SFER – Volume I (Sotolongo Lopez et al. 2023). The chapter is intended to fulfill the EFA requirement for an annual report to “identify water quality parameters, in addition to phosphorus, which exceed state water quality standards or are causing or contributing to adverse impacts in the Everglades Protection Area.” Additionally, this chapter provides an annual update of the comprehensive overview of nitrogen and phosphorus concentrations and loads throughout the EPA, along with an assessment of TP criterion achievement utilizing the protocol provided in the 2007 SFER – Volume I, Chapter 3C (Payne et al. 2007).

More specifically, this chapter and its associated appendices use water quality data collected during WY2023 to achieve the following objectives:

1. Summarize areas and times where water quality criteria are not being met and indicate trends in excursions over space and time.
2. Discuss factors contributing to excursions from water quality criteria and provide an evaluation of natural background conditions where existing standards may not be appropriate.
3. Present an updated review of pesticide and priority pollutant data made available during WY2023.
4. Present a preliminary TP criterion achievement assessment for different areas within the EPA for the most recent five-year period (WY2019–WY2023).
5. Summarize phosphorus and nitrogen concentrations measured in surface waters within different portions of the EPA.

6. Summarize the flow and nutrient loads entering different portions of the EPA and describe spatial and temporal trends observed.
7. Describe and discuss factors contributing to any spatial and temporal trends observed.

---

## METHODS

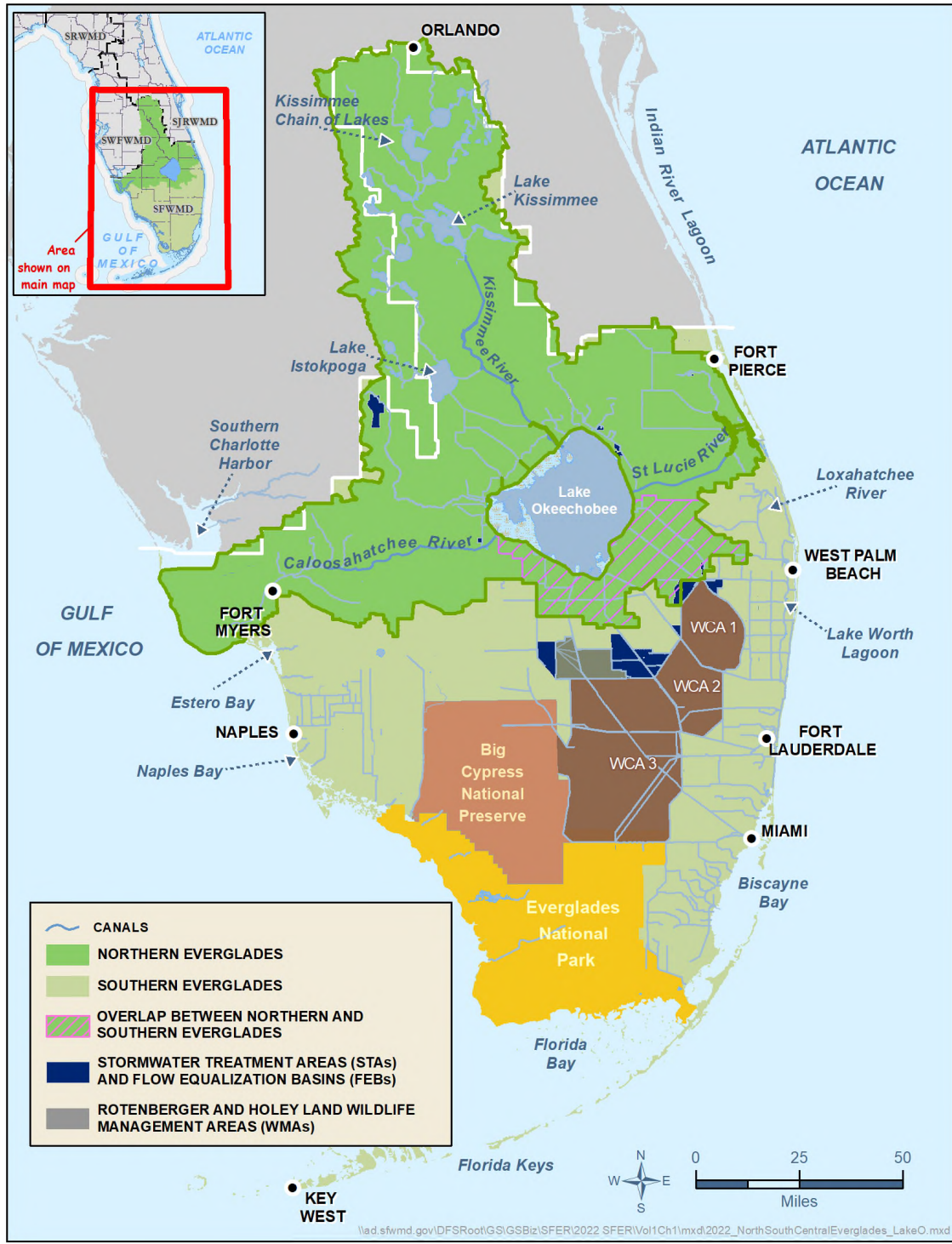
---

A regional synoptic approach, used for water quality evaluations in the *1999 Everglades Interim Report*, *2000–2004 Everglades Consolidated Reports*, and 2005–2023 SFERs, was applied to phosphorus and nitrogen data for WY2023 to provide an overview of water quality status within the EPA. Consolidating regional water quality data provides the ability to analyze data over time but limits spatial analyses within each region. However, spatial analyses can be made between regions because most inflow and pollutants enter the northern third of the EPA, and the net water flow is from north to south.

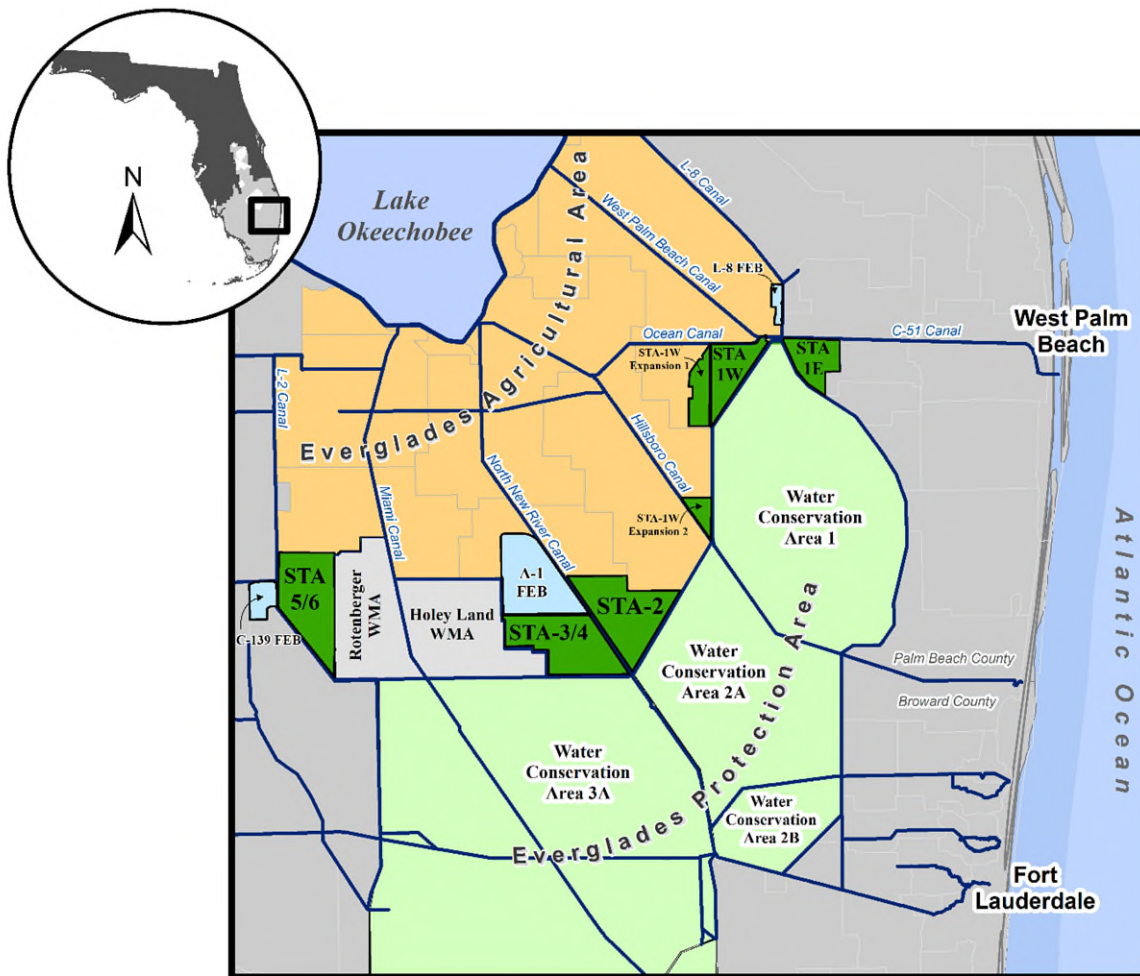
## AREA OF INTEREST

The EPA is a complex system of marsh areas, canals, and levees with inflow and outflow water control structures that covers almost 2.5 million acres (> 1 million hectares or > 10 billion square meters) of former Everglades marsh and, currently, is divided into large distinct shallow impoundments (Bancroft et al. 1992). In addition to rainfall inputs, surface water inflows regulated by water control structures from agricultural tributaries, such as the Everglades Agricultural Area (EAA) to the north and the C-139 Basin to the west, feed the EPA. The EPA also receives surface water inflows originating from Lake Okeechobee to the north and from predominantly urbanized areas to the east. The timing and distribution of the surface inflows from the tributaries to the EPA are based on a complex set of operational decisions that account for natural and environmental system requirements, water supply for urbanized and natural areas, aquifer recharge, and flood control. The major features of the EPA and surrounding area are illustrated in **Figures 3-1** and **3-2**.

The designated use of the EPA, as delineated by FDEP is for “fish consumption, recreation, propagation and maintenance of a healthy, well-balanced population of fish and wildlife” and, as such, is classified as a Class III water body. Surface waters of the state, their designated uses, and classification are described in Rule 62-302.400, Florida Administrative Code (F.A.C.).



**Figure 3-1.** Major geographic features of the South Florida environment within the South Florida Water Management District’s boundaries. The EPA region includes Arthur R. Marshall Loxahatchee National Wildlife Refuge (LNWR), also known as Water Conservation Area (WCA) 1, WCA-2, WCA-3, and Everglades National Park (ENP).

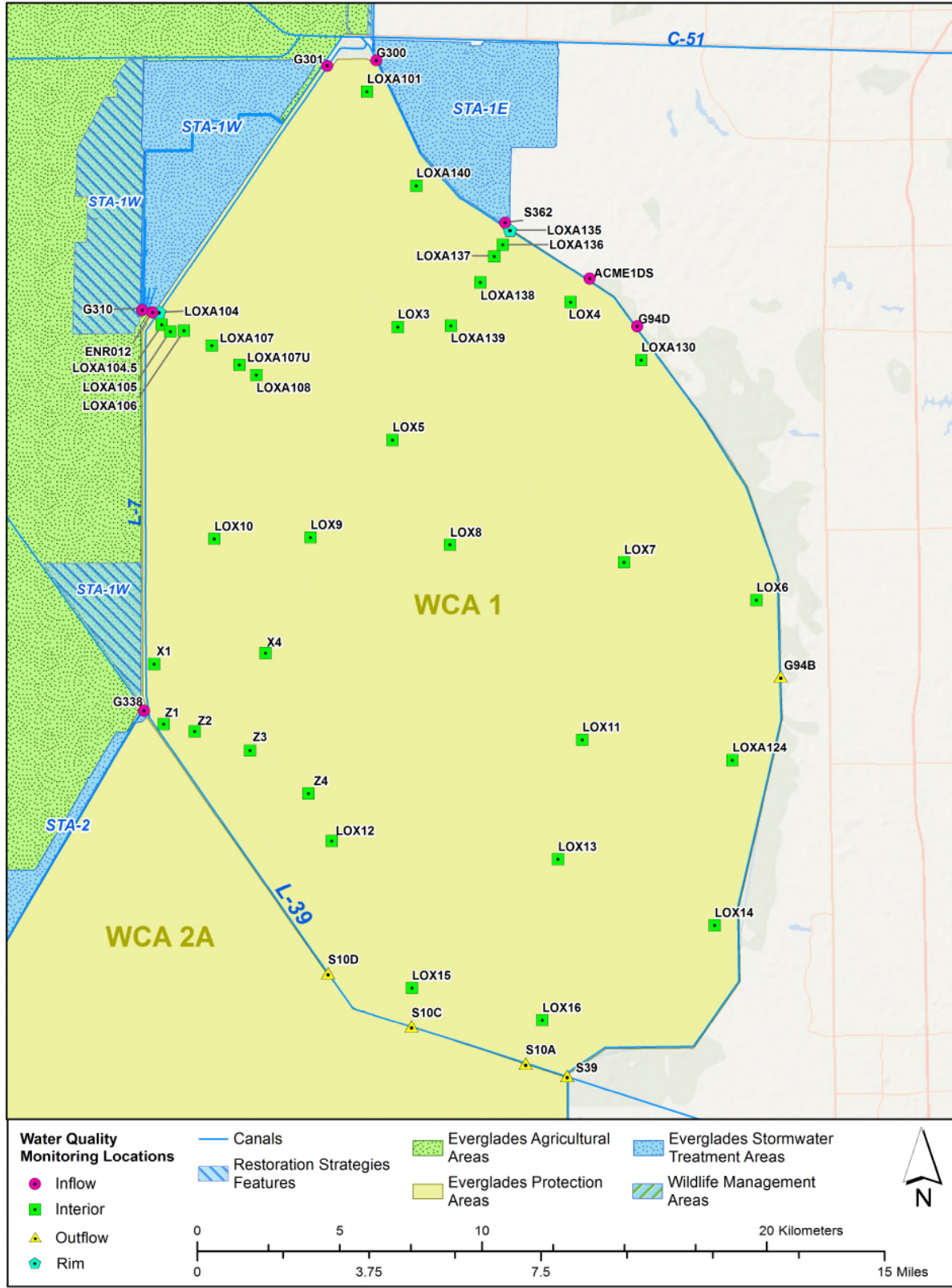


**Figure 3-2.** Location of the Everglades Stormwater Treatment Areas (STAs) (green); STA-1W Expansion Area; and A-1, L-8, and C-139 flow equalization basins (FEBs) in relation to Lake Okeechobee (light blue), WCAs (lime green), and other landscape features of South Florida. (Note: WMA – Wildlife Management Area.)

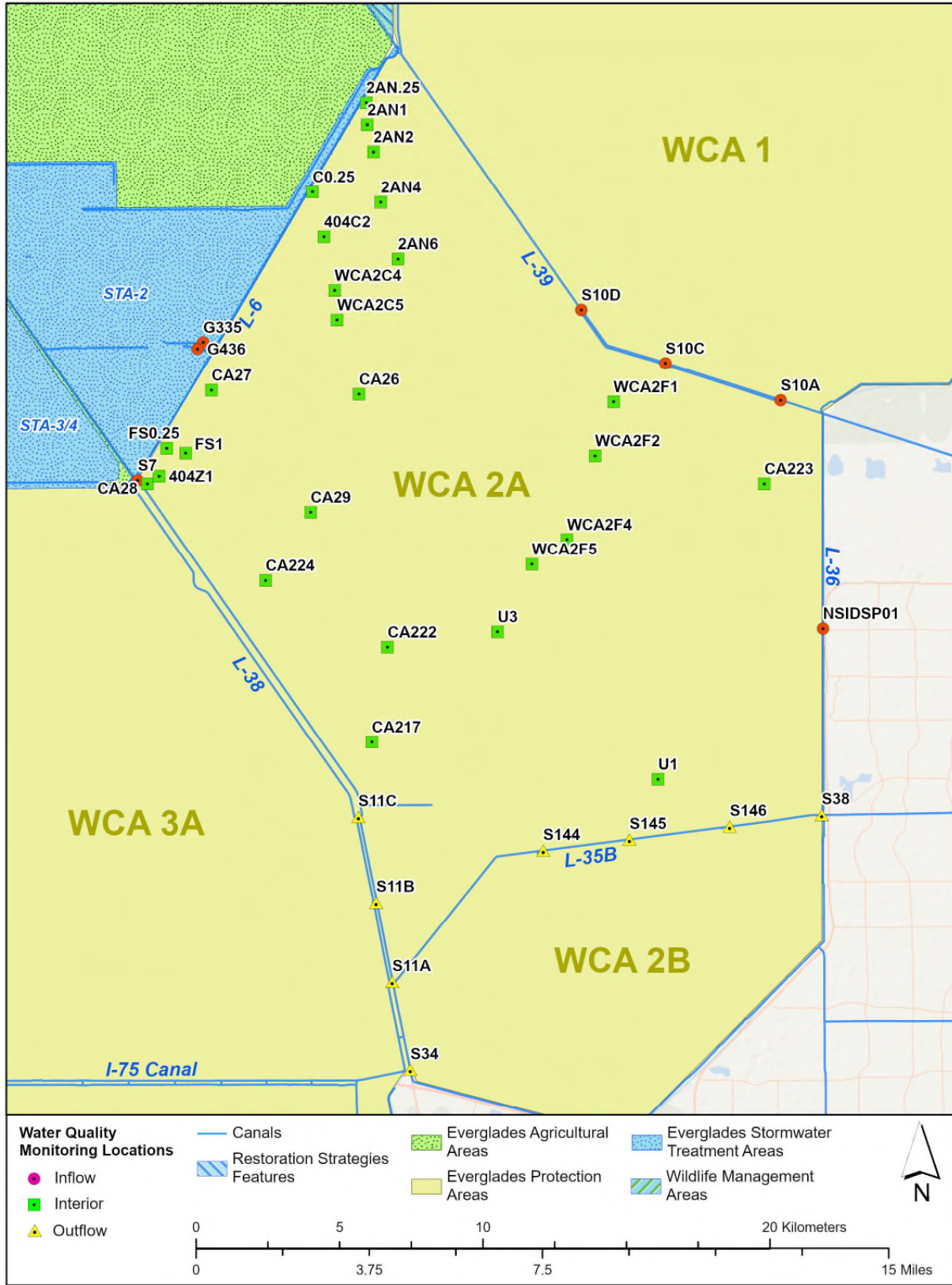
## WATER QUALITY SAMPLING STATIONS IN THE EPA

A network of water quality sampling stations has been identified (**Figures 3-3 through 3-6**) to assess annual water quality criteria exceedance and long-term trends. These stations are part of the South Florida Water Management District's (SFWMD's) long-term projects being monitored for different mandates and purposes. They were carefully selected to be representative of either the EPA boundary conditions (i.e., inflow or outflow) or ambient marsh conditions (interior). An effort has been made to utilize a consistent group of stations among previous annual consolidated reports to ensure reliable and comparable results. Table 2 in Appendix 3-1 provides a crosswalk table for each station's identifier due to the naming convention changes that have occurred for monitoring stations within the EPA throughout their monitoring period.

Water quality sampling stations located throughout the WCAs and ENP were categorized as inflow, interior, or outflow stations within each region based on their location and function (**Figures 3-3 through 3-6**). This organization of monitoring stations allows a more detailed analysis of the water quality status in each region of the EPA and assists in the evaluation of potential causes for observed excursions from Class III water quality criteria.

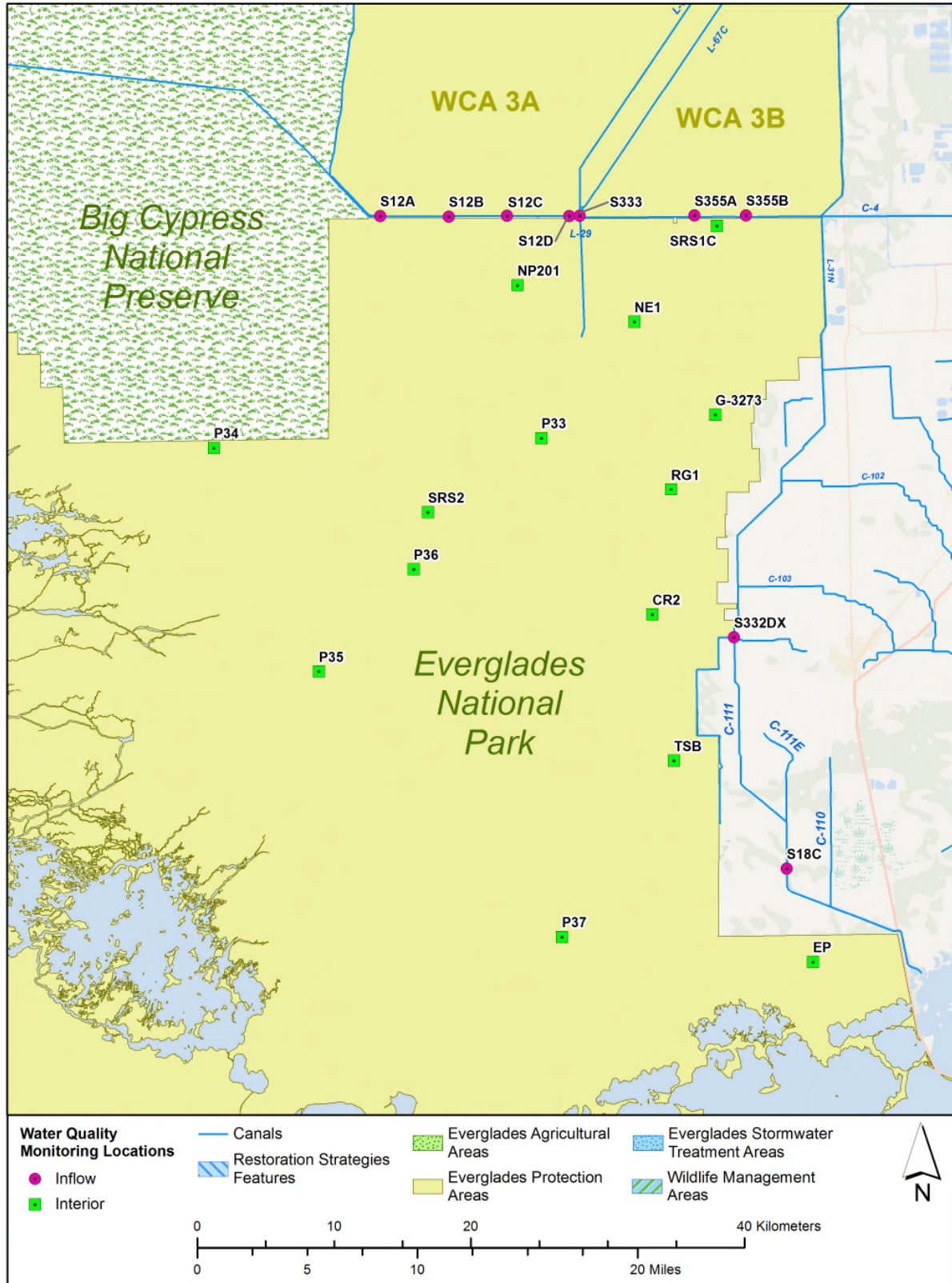


**Figure 3-3.** Location and classification of water quality monitoring stations for LNWR (WCA-1). (Notes: Stations G300 and G301 located north of LNWR are diversion structures and rarely exhibit flow into LNWR. STA-1E – STA 1 East and STA-1W – STA 1 West.)



**Figure 3-4.** Location and classification of water quality monitoring stations for WCA-2.





**Figure 3-6.** Location and classification of water quality monitoring stations for ENP.

Several interior structures convey water between different regions in the EPA and are designated as both inflow and outflow stations based on this categorization system. For example, the S-10 structures act as both outflow stations for LNWR and inflow stations to WCA-2 (**Figures 3-3 and 3-4**). The interior sites of each region consist of marsh and canal stations as well as structures that convey water within the area.

In addition to inflow, outflow, and interior stations, LNWR has a category for rim canal sites to account for water entering LNWR interior from canals that border the east and west levees (**Figure 3-3**). Waters discharged to the L-7 rim canal will either overflow into LNWR interior when canal stages exceed the ground elevation or will bypass the marsh and be discharged to WCA-2A through the S-10 structures. The extent (distance) to which rim canal overflows penetrate the marsh depends on the relative stages of the L-7 and L-40 rim canals and LNWR interior.

Water sampling is conducted by SFWMD following strict quality assurance and quality control procedures specified in the SFWMD *Field Sampling Manual*.<sup>5</sup> Sampling collection method differs by monitoring mandates specified in permits. Sampling frequency varies by station depending on its classification, parameter group, and hydrologic conditions (e.g., water depth and flow). Water control structures (inflows and outflows) were typically sampled biweekly when flowing; otherwise, sampling was performed monthly. Generally, interior monitoring stations were sampled monthly for most parameters reported in this chapter. Pesticide monitoring is conducted across the entire area SFWMD manages at 15 sites on a biannual basis. An overview of the water quality monitoring projects, including project descriptions and objectives with limited site-specific information, is available on SFWMD's website at <https://www.sfwmd.gov/science-data/environmental-monitoring>.

## ANALYSIS PERIODS

As previously noted, the primary focus of this chapter is to summarize the status of water quality within the EPA during WY2023 and describe trends or changes in water quality conditions over time. To accomplish this objective, comparisons are made across discrete periods that correspond to major restoration activities occurring upstream or within the EPA. The four periods are (1) the historical WY1979–WY1993 period (Baseline), which corresponds to the timeframe prior to implementation of the EAA Best Management Practices (BMPs) Program and the Everglades Construction Project (ECP; i.e., Everglades Nutrient Removal Project and Everglades Stormwater Treatment Areas [STAs]), (2) the intermediate WY1994–WY2004 period (Phase I), (3) the Phase II BMP and STA implementation period after WY2004 (i.e., WY2005–WY2022), and (4) the current water year, WY2023.

Phase I represents the period in which implementation of the EAA BMP Program was increasing, and all the initial Everglades STAs were constructed and became operational. The Phase II BMP and STA implementation period corresponds to when the performance of the BMPs and STAs were being optimized and enhanced. Additionally, during this period, various restoration projects were being implemented under the *Everglades Protection Area Tributary Basins Long Term Plan for Achieving Water Quality Goals* (Burns & McDonnell 2003), *Restoration Strategies Regional Water Quality Plan* (SFWMD 2012), *Comprehensive Everglades Restoration Plan* (CERP; USACE and SFWMD 1999), and other state or federal restoration projects. Because optimization, enhancement, and other restoration activities are expected to continue for years, the Phase II period will continue to expand in future SFERs to incorporate additional years of sampling. In addition, data for the current water year (WY2023) are used to make comparisons with the historical periods and are analyzed independently as the fourth period. Individual station assessments and certain mandated reporting (e.g., TP criterion achievement) were based on the previous five water years (WY2019–WY2023) rather than on the single year used for regional analysis. Reporting periods are specified in each section of this chapter.

---

<sup>5</sup> As of publication of the 2024 South Florida Environmental Report, the current SFWMD *Field Sampling Manual* is SFWMD-FIELD-FSM-001-14 and was approved on November 20, 2023.

## WATER QUALITY DATA SOURCES

Most of the water quality data evaluated in this chapter was retrieved from SFWMD’s corporate environmental database, DBHYDRO (<https://www.sfwmd.gov/science-data/dbhydro>). Additionally, water quality data from the nutrient gradient sampling stations monitored by SFWMD were obtained from SFWMD’s Water Resources Division database. Due to differences in data sources, a station crosswalk can be found in Table 2 of Appendix 3-1.

## DATA SCREENING AND HANDLING

Water quality data were screened based on laboratory qualifier codes, consistent with the FDEP Quality Assurance Rule (Chapter 62-160, F.A.C.). Any datum associated with a fatal qualifier (e.g., H, J, K, N, O, V, Q, Y, G, or ?), indicating a potential data quality problem, was removed from the analysis. Values that exceeded possible physical or chemical measurement constraints (e.g., if resulting pH is greater than 14), had temperatures well outside seasonal norms (e.g., 6 degrees Celsius [°C] in July), or represented data transcription errors were excluded. Multiple samples collected at the same location on the same day were considered as one sample, with the arithmetic mean used to represent the sampling period.

Additional considerations in the handling of water quality data are the accuracy and sensitivity of the laboratory method used. For purposes of summary statistics presented in this chapter, data reported as less than the MDL were assigned a value of one-half the MDL unless otherwise noted. All data presented in this chapter, including historical results, were handled consistently regarding screening and MDL replacement.

## WATER QUALITY DATA PARAMETERS

The District monitors numerous water quality parameters within the EPA (Payne and Xue 2012). Given this chapter’s focus on water quality criteria, the evaluation was primarily limited to parameters with Class III criteria pursuant to the FDEP’s *Surface Water Quality Standards Rule* (Chapter 62-302, F.A.C.). The parameters evaluated in this chapter include a large suite of pesticides, herbicides, and breakdown products in addition to the following water quality constituents:

- Alkalinity
- DO (in situ)
- Specific conductance (in situ)
- pH (in situ)
- Total selenium\*
- Total thallium\*
- Total zinc\*
- TAN
- TN
- Turbidity
- Total cadmium\*
- Total iron
- Total lead\*
- Total nickel\*
- Total silver\*
- Total antimony\*
- Total arsenic\*
- Total beryllium\*
- Total copper\*
- TP
- OPO<sub>4</sub>

Parameters marked with an asterisk (\*) were not measured in WY2023. However, these have been analyzed and reported in previous SFERs and, if measured in the future, will be analyzed, and reported in future SFERs. Since WY2007, monitoring of metals entering the EPA have been eliminated due to the prevalence of metals being observed below the established water quality standards identified in Section 62.302.530, F.A.C., and the lack of new sources.

## WATER QUALITY CRITERIA EXCURSION ANALYSIS

FDEP and SFWMD have developed an excursion analysis protocol for use in the annual SFER (Weaver and Payne 2005) to effectively provide a synoptic view of water quality criteria compliance on a regional scale (i.e., LNWR, WCA-2, WCA-3, and ENP). The protocol was developed to balance consistency with previous versions of the report, other State of Florida ambient water quality evaluation methodologies (e.g., Impaired Waters 303(d) designations), and the United States Environmental Protection Agency's (USEPA's) exceedance frequency recommendations, as well as provide a concise summary for decision makers and the public. This methodology ensures results will be compatible with information from other sources provided to water managers.

A multi-tiered categorical system was used in this chapter to rank the severity of excursions from state water quality criteria (see **Table 3-2** later in this chapter). Categories were assigned based on sample excursion frequencies evaluated using a statistically valid assessment methodology (i.e., binomial hypothesis test) that accounted for uncertainty in monitoring data (Weaver and Payne 2005). Parameters without exceedances were categorized as **no concern** and are not discussed further in this chapter. If a parameter had fewer than 28 samples, they were initially categorized as concern or potential concern based on excursion frequencies of greater than 20% and less than 20%, respectively. For parameters with greater than 28 samples, based on the results of the binomial test using a 90% confidence level, parameters with exceedance rates between 0 and 5% are classified as **minimal concern**, those with exceedance rates between 5 and 10% are classified as **potential concern**, and those with exceedance rates greater than 10% are classified as **concern**.

Because exceedances of the pesticide criteria can result in more immediate and severe effects to aquatic organisms and human health, a 10% excursion frequency was not used in the assessment of pesticides as recommended by USEPA (USEPA 1997, 2002). Pesticides were evaluated under the assumption that the Class III criteria values represent instantaneous maximum concentrations for which any exceedance constitutes a non-attainment of designated use. Pesticides were categorized based on whether the parameter was detected at concentrations above the MDL (**potential concern**) or at concentrations exceeding Class III criteria or chronic toxicity values (**concern**). Pesticides classified as concern have a high likelihood of resulting in an impairment of the designated use of the water body. Classification of a pesticide as a potential concern signifies that the constituent is known to be present within the basin at concentrations reasonably known to be below levels that can result in adverse biologic effects but may result in a problem at some future date or in interaction with other compounds. The **no concern** category was used to designate pesticides that were not detected at sites within a given area.

The data sources as well as the data handling and evaluation methods employed in this chapter are identical to those used in previous SFERs. Greater detail concerning the methods used can be found in Weaver and Payne (2005) and Payne and Xue (2012).

## PHOSPHORUS CRITERION ACHIEVEMENT ASSESSMENT

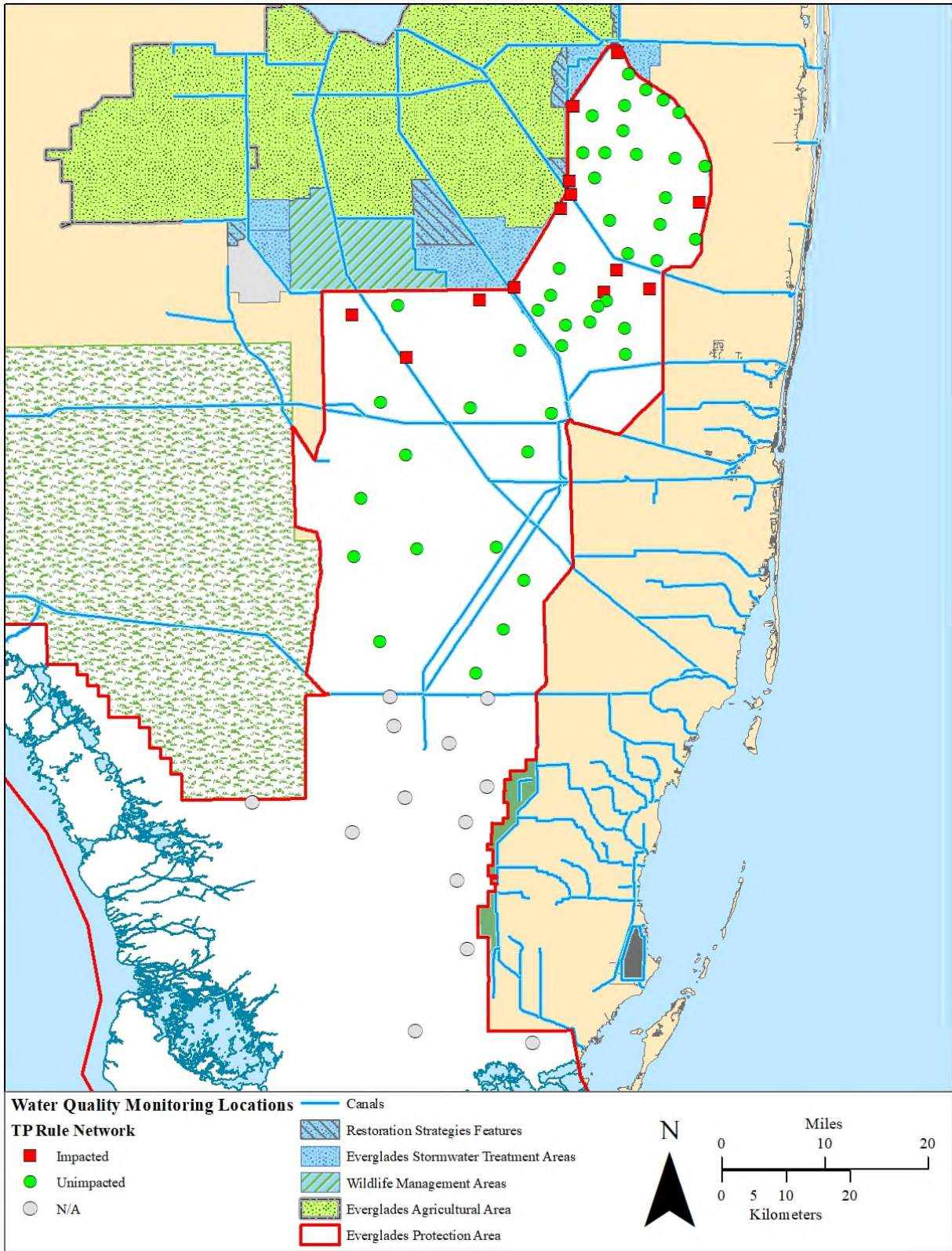
An evaluation to determine achievement of the TP criterion was performed consistent with (1) the assessment protocol presented by Payne et al. (2007), and (2) the four-part test outlined below and specified in the FDEP's *Water Quality Standards for Phosphorus within the Everglades Protection Area*, referred to as the TP Rule (Section 62-302.540, F.A.C.). Achievement of the TP Rule is assessed for networks of impacted and unimpacted, spatially explicit monitoring locations in the WCAs (i.e., LNWR [WCA-1], WCA-2, and WCA-3). Achievement of the phosphorus criterion is different for ENP than the established TP criterion for the EPA. As acknowledged by Section 62-302.530(4)(c), F.A.C., achievement of the TP criterion for ENP is assessed according to methods set forth in Appendix A of the Settlement Agreement (Case No. 88-1886-CIV-MORENO) until the Settlement Agreement is amended or terminated. Reports and supporting information related to TP assessments consistent with Appendix A of the Settlement Agreement can be found at <https://www.sfwmd.gov/our-work/toc>.

Achievement of the TP criterion is assessed by a four-part test for each WCA using two networks of stations: impacted and unimpacted. According to Subparagraph 62-302.540(4)(d)2, F.A.C., of the TP Rule, individual stations in networks shall be deemed to be unimpacted for purposes of determining compliance assessment with the TP Rule if the five-year geometric mean is less than or equal to 10 µg/L TP and the annual geometric mean is less than or equal to 15 µg/L TP (in 2002, the TP MDL changed from 0.004 to 0.002 mg/L; see Appendix 3-6 of this volume for more information). The four parts of the achievement test are as follows:

1. The five-year geometric mean averaged across all stations is less than or equal to 10 µg/L.
2. The annual geometric mean averaged across all stations is less than or equal to 10 µg/L for three of five water years.
3. The annual geometric mean averaged across all stations is less than or equal to 11 µg/L.
4. The annual geometric mean at all individual stations is less than or equal to 15 µg/L.

Data from the 58 stations within the TP criterion monitoring network for the most recent five-year period (i.e., WY2019–WY2023) were utilized in the evaluation. The location of the TP criterion network monitoring stations established pursuant to the TP Rule used for the TP criterion assessment along with their classification as impacted or unimpacted are provided in **Figure 3-7**. Details concerning the selection of stations in the TP criterion monitoring networks and their classification can be found in Payne et al. (2007) and Julian (2015).

Data collection from the complete TP criterion monitoring network was initiated in January 2007. Data availability is limited for certain portions of the EPA due to extremely dry conditions that have prevailed during some of the years since WY2007. Because the results of the TP criterion compliance assessment presented in this chapter could be affected by these data limitations, this evaluation should be considered preliminary, and the results cautiously interpreted. It is expected that future assessments will improve as additional data sets are added. Data were screened according to the quality assurance and quality control procedures described in the FDEP protocol available at <https://floridadep.gov/sites/default/files/data-quality-screening-protocol.pdf> (FDEP 2004) and Appendix 3-6.



**Figure 3-7.** Location of TP criterion assessment monitoring stations and their respective classifications used in WY2019–WY2023 evaluations.

## STATISTICAL ANALYSIS

All valid data (i.e., non-qualified data) were used to compute summary statistics for all regions across the EPA (i.e., inflow, outflow, interior, and rim). Trend analysis was performed on annual geometric mean TP and TN concentrations for inflow and interior regions of the EPA using the Kendall's  $\tau$  correlation analysis (Base stats R package) and Sen's slope estimate (zyp R package; Bronaugh and Werner 2019). Trend analysis was performed on annual geometric mean TP for each monitoring station with greater than three years of data using Kendall's  $\tau$  correlation analysis and Sen's slope estimate. Estimated cumulative distribution functions (CDFs), point estimates, and 95% confidence intervals (spsurvey R package; Kincaid et al. 2019) were developed for interior (i.e., marsh) annual geometric mean TP concentrations observed during the current water year. Only stations with sufficient data (i.e., passed TP Rule screening) were included in this estimate. Calculation of CDFs were conducted. All statistical operations were performed with R© (Version 4.1.2, R Foundation for Statistical Computing, Vienna, Austria). The critical level of significance ( $\alpha$ ) was set at 0.05.

---

## WY2023 WATER QUALITY RESULTS

---

In WY2023, there were an average of 248 sampling days in each area of the EPA. ENP and WCA-3 had the greatest number of sampling days, with 360 and 362 sampling days, respectively; 129 sampling days occurred within LNWR; and 141 sampling days occurred within WCA-2. Very few samples collected during WY2023 resulted in qualified data; approximately 2.6% (1,196 qualified samples from a total of 45,222 samples collected) of the data collected was removed due to fatal qualifiers. The dominant fatal qualifier was the J qualifier (estimated value).

## WATER QUALITY CRITERIA EXCURSION ANALYSIS

Summarized by region and classification, WY2023 data is included in Appendix 3-1 of this volume. Additionally, data for the last five water years (WY2019–WY2023) summarized by region, class, and monitoring station is presented in Appendix 3-2. Comparisons of WY2023 water quality data with applicable Florida Class III water quality criteria resulted in excursions for four water quality parameters: DO, alkalinity, pH, and specific conductance (**Table 3-1**). Like previous periods, these excursions were generally isolated to specific areas of the EPA.

Water quality parameters with exceedances of applicable criteria are discussed further below, with the excursion frequencies summarized for the Baseline through the current reporting periods (WY1979–WY1993, WY1994–WY2004, WY2005–WY2022, and WY2023) to evaluate the presence of any temporal trends (**Table 3-1**). The last five water years summarized by monitoring station is provided in Appendix 3-2.

**Table 3-1.** Excursions from Florida Class III criteria in the EPA for the Baseline period (WY1979–1993), Phase I (WY1994–WY2004), Phase II (WY2005–WY2022), and WY2023.

Area	Class	Parameter	Number of Excursions / Sample Size, Percent Excursions (Category) <sup>a</sup>			
			Baseline WY1979-WY1993	Phase I WY1994-WY2004	Phase II WY2005-WY2022	WY2023
LNWR	Inflow	Alkalinity	5 / 1604, 0.3% (MC)	0 / 1158, 0% (NC)	0 / 2238, 0% (NC)	0 / 106, 0% (NC)
		Dissolved Oxygen <sup>b</sup>	550 / 1118, 49.2% (C)	950 / 2173, 43.7% (C)	1137 / 3663, 31% (C)	47 / 204, 23% (C)
		pH	8 / 1100, 0.7% (MC)	4 / 2177, 0.2% (MC)	13 / 3736, 0.3% (MC)	0 / 202, 0% (NC)
		Specific Conductance	478 / 1114, 42.9% (C)	373 / 2178, 17.1% (C)	435 / 3753, 11.6% (C)	46 / 204, 22.5% (C)
		TAN	1 / 1076, 0.1% (MC)	0 / 1622, 0% (NC)	1 / 1902, 0.1% (MC)	0 / 65, 0% (NC)
	Interior	Alkalinity	26 / 387, 6.7% (PC)	59 / 1148, 5.1% (MC)	114 / 2406, 4.7% (MC)	4 / 492, 0.8% (MC)
		Dissolved Oxygen <sup>c</sup>	8 / 30, 26.7% (C)	21 / 140, 15% (C)	98 / 528, 18.6% (C)	7 / 28, 25% (C)
		pH	59 / 253, 23.3% (C)	131 / 1394, 9.4% (PC)	135 / 5042, 2.7% (MC)	8 / 815, 1% (MC)
		Specific Conductance	6 / 153, 3.9% (MC)	1 / 1365, 0.1% (MC)	0 / 4950, 0% (NC)	0 / 815, 0% (NC)
		TAN	0 / 189, 0% (NC)	0 / 1053, 0% (NC)	0 / 2203, 0% (NC)	0 / 123, 0% (NC)
	Outflow	Alkalinity	1 / 580, 0.2% (MC)	0 / 710, 0% (NC)	1 / 796, 0.1% (MC)	0 / 25, 0% (NC)
		Dissolved Oxygen <sup>b</sup>	279 / 593, 47% (C)	258 / 697, 37% (C)	103 / 1175, 8.8% (PC)	0 / 47, 0% (NC)
		pH	1 / 581, 0.2% (MC)	4 / 693, 0.6% (MC)	2 / 1191, 0.2% (MC)	0 / 55, 0% (NC)
		Specific Conductance	130 / 597, 21.8% (C)	21 / 695, 3% (MC)	1 / 1190, 0.1% (MC)	0 / 55, 0% (NC)
		TAN	2 / 576, 0.3% (MC)	0 / 666, 0% (NC)	0 / 768, 0% (NC)	0 / 18, 0% (NC)
	Rim	Dissolved Oxygen <sup>b</sup>	19 / 96, 19.8% (C)	199 / 454, 43.8% (C)	66 / 558, 11.8% (C)	3 / 34, 8.8% (MC)
		pH	0 / 96, 0% (NC)	0 / 457, 0% (NC)	1 / 684, 0.1% (MC)	0 / 46, 0% (NC)
		Specific Conductance	27 / 96, 28.1% (C)	57 / 459, 12.4% (C)	14 / 682, 2.1% (MC)	2 / 46, 4.3% (MC)
		TAN	0 / 96, 0% (NC)	0 / 423, 0% (NC)	0 / 95, 0% (NC)	N/A <sup>d</sup>
	WCA2	Inflow	Dissolved Oxygen <sup>b</sup>	287 / 635, 45.2% (C)	290 / 950, 30.5% (C)	409 / 3080, 13.3% (C)
pH			2 / 622, 0.3% (MC)	5 / 953, 0.5% (MC)	4 / 3123, 0.1% (MC)	0 / 194, 0% (NC)
Specific Conductance			162 / 641, 25.3% (C)	129 / 954, 13.5% (C)	163 / 3127, 5.2% (MC)	20 / 198, 10.1% (PC)
TAN			2 / 616, 0.3% (MC)	0 / 766, 0% (NC)	0 / 1655, 0% (NC)	0 / 95, 0% (NC)
Interior		Dissolved Oxygen <sup>c</sup>	45 / 60, 75% (C)	43 / 88, 48.9% (C)	86 / 320, 26.9% (C)	8 / 18, 44.4% (C)
		pH	15 / 375, 4% (MC)	2 / 1469, 0.1% (MC)	2 / 2994, 0.1% (MC)	0 / 321, 0% (NC)
		Specific Conductance	69 / 263, 26.2% (C)	191 / 1505, 12.7% (C)	174 / 2987, 5.8% (PC)	12 / 321, 3.7% (MC)
		TAN	2 / 293, 0.7% (MC)	2 / 1115, 0.2% (MC)	0 / 1309, 0% (NC)	0 / 38, 0% (NC)
Outflow		Dissolved Oxygen <sup>b</sup>	427 / 1371, 31.1% (C)	379 / 1030, 36.8% (C)	426 / 1788, 23.8% (C)	7 / 72, 9.7% (PC)
		pH	4 / 1357, 0.3% (MC)	6 / 1054, 0.6% (MC)	0 / 1809, 0% (NC)	0 / 84, 0% (NC)
		Specific Conductance	42 / 1375, 3.1% (MC)	3 / 1048, 0.3% (MC)	0 / 1819, 0% (NC)	0 / 84, 0% (NC)
		TAN	0 / 1341, 0% (NC)	0 / 1029, 0% (NC)	0 / 1347, 0% (NC)	0 / 50, 0% (NC)

**Table 3-1.** continued.

Area	Class	Parameter	Number of Excursions / Sample Size, Percent Excursions (Category) <sup>a</sup>			
			Baseline WY1979–WY1993	Phase I WY1994–WY2004	Phase II WY2005–WY2022	WY2023
WCA3	Inflow	Dissolved Oxygen <sup>b</sup>	908 / 2112, 43% (C)	1272 / 3116, 40.8% (C)	6254 / 12893, 48.5% (C)	421 / 753, 55.9% (C)
		pH	17 / 2089, 0.8% (MC)	15 / 3162, 0.5% (MC)	11 / 13046, 0.1% (MC)	1 / 823, 0.1% (MC)
		Specific Conductance	58 / 2138, 2.7% (MC)	7 / 3147, 0.2% (MC)	13 / 13060, 0.1% (MC)	0 / 832, 0% (NC)
		TAN	0 / 1927, 0% (NC)	0 / 2370, 0% (NC)	0 / 4265, 0% (NC)	0 / 239, 0% (NC)
	Interior	Dissolved Oxygen <sup>c</sup>	21 / 67, 31.3% (C)	44 / 133, 33.1% (C)	36 / 222, 16.2% (C)	1 / 11, 9.1% (PC)
		pH	1 / 407, 0.2% (MC)	0 / 1935, 0% (NC)	1 / 2132, 0% (MC)	0 / 310, 0% (NC)
		Specific Conductance	4 / 297, 1.3% (MC)	0 / 1946, 0% (NC)	0 / 2142, 0% (NC)	0 / 310, 0% (NC)
		TAN	0 / 297, 0% (NC)	0 / 1430, 0% (NC)	0 / 1658, 0% (NC)	0 / 73, 0% (NC)
	Outflow	Dissolved Oxygen <sup>b</sup>	778 / 1925, 40.4% (C)	953 / 2408, 39.6% (C)	1767 / 4793, 36.9% (C)	116 / 351, 33% (C)
		pH	24 / 1891, 1.3% (MC)	22 / 2632, 0.8% (MC)	4 / 5004, 0.1% (MC)	0 / 365, 0% (NC)
		Specific Conductance	0 / 1952, 0% (NC)	0 / 2645, 0% (NC)	0 / 4987, 0% (NC)	0 / 366, 0% (NC)
		TAN	0 / 1659, 0% (NC)	0 / 1637, 0% (NC)	0 / 602, 0% (NC)	N/A <sup>d</sup>
ENP	Inflow	Dissolved Oxygen <sup>b</sup>	911 / 2287, 39.8% (C)	1250 / 3031, 41.2% (C)	2122 / 5568, 38.1% (C)	168 / 387, 43.4% (C)
		pH	26 / 2252, 1.2% (MC)	33 / 3047, 1.1% (MC)	2 / 5635, 0% (MC)	0 / 386, 0% (NC)
		Specific Conductance	0 / 2314, 0% (NC)	1 / 3019, 0% (MC)	0 / 5610, 0% (NC)	0 / 387, 0% (NC)
		TAN	0 / 2021, 0% (NC)	0 / 1962, 0% (NC)	0 / 695, 0% (NC)	N/A <sup>d</sup>
	Interior	Dissolved Oxygen <sup>c</sup>	1 / 69, 1.4% (MC)	3 / 105, 2.9% (MC)	17 / 173, 9.8% (PC)	1 / 9, 11.1% (PC)
		pH	9 / 459, 2% (MC)	27 / 1023, 2.6% (MC)	0 / 1460, 0% (NC)	0 / 269, 0% (NC)
		Specific Conductance	20 / 546, 3.7% (MC)	2 / 1055, 0.2% (MC)	2 / 1490, 0.1% (MC)	0 / 315, 0% (NC)
		TAN	3 / 455, 0.7% (MC)	1 / 966, 0.1% (MC)	0 / 989, 0% (NC)	0 / 65, 0% (NC)

a. Excursion categories of concern, potential concern, minimal concern, and no concern are denoted by C, PC, MC, and NC, respectively.

b. DO for inflow, outflow, and rim canal sampling locations were assessed using the Florida Class III freshwater water quality standard identified in Section 62-302.533, F.A.C.

c. DO SSAC was used to assess water quality excursions.

d. N/A - not available.

e. Due to low sample size, some of these estimates should be used with caution when interpreting the excursion categories.

## Dissolved Oxygen

Marsh dissolved oxygen (DO) conditions within the EPA were assessed utilizing the Everglades DO SSAC for all periods. The SSAC was developed and implemented during 2004, but to be consistent among time periods, the DO SSAC was applied across all periods, including those prior to 2004. Because a single-value criterion does not adequately account for the wide ranging natural daily fluctuations observed in the Everglades marshes, the SSAC uses an algorithm that includes sample collection time and water temperature to model the observed natural sinusoidal diel cycle and seasonal variability (Weaver 2004). The DO SSAC was originally developed to assess DO conditions within the EPA (i.e., marsh interior stations); therefore, for this analysis, DO SSAC was applied to interior monitoring locations. Compliance with the DO water quality standard for inflow, outflow, and rim canal monitoring locations was assessed using the Class III standard (discussed below); however, for informational purposes only, the DO SSAC was also applied to inflow, outflow, and rim canal monitoring locations and presented in Appendix 3-3 of this volume. The SSAC is assessed based on a comparison between the annual average measured DO concentration and average of the corresponding DO limits. DO excursion results for WY2023 for individual stations are provided in Appendix 3-3, Table 1, of this volume.

During WY2023, 16 interior stations within the entire EPA (LOXA136, LOXA104.5, LOXA105, LOX4, LOX16, Z1, Z2, WCA2F1, WCA2F2, CA27, 2AN1, 2AN2, 2AC.25, FS0.25, FS1, and NE1) exceeded the DO SSAC. Only two samples were collected at station CA36 which does not constitute enough data set for DO SSAC annual assessment. The single interior station within ENP that failed to achieve the SSAC is potentially influenced by canal flow or natural marsh processes that result in low DO, such as elevated biological oxygen demand due to accumulation of organic matter within the slough. The annual geometric mean TP concentration at this station has been at or below 5 µg/L for the past five water years, and it is more likely that low DO is a result of internal processes rather than the delivery of excess nutrients in surface water. Most interior marsh stations within LNWR, WCA-2, and WCA-3 that failed to achieve the SSAC during WY2023 reside within phosphorus-impacted areas or are heavily influenced by canal flow.

DO for inflow, outflow, and rim canal monitoring stations were assessed using the Class III freshwater water quality standard (authorized August 1, 2013), which states that “no more than 10 percent of the daily average percent dissolved oxygen (DO) saturation values shall be below... 38 percent...in the Everglades bioregions” (Section 62-302.533(1)(a), F.A.C.) or for instantaneous data (discrete measurements), the percent DO saturation values shall not exceed the limit based on the calculated time-day specific translation (FDEP 2013). For WY2023, several inflow, outflow, and rim canal stations (46 out of 67) exceeded the DO water quality standard. A detailed list of stations, summary statistics, and water quality standard pass or fail determination is presented in Appendix 3-3, Table 2. For comparison purposes only, the DO Class III (freshwater) water quality standard was applied to interior monitoring locations.

Inflow regions for LNWR, WCA-2, WCA-3, and ENP were classified as a concern for all periods. The rim canal region of LNWR was categorized as minimal concern for DO in WY2023, continuing to be an improvement over the Baseline, Phase I, and Phase II periods where it was classified as concern. Outflow regions for WCA-3 were classified as a concern for all periods. During WY2023, outflow regions for WCA-2 showed improvements moving from concern during Baseline and Phase I periods to potential concern. DO in outflow from LNWR has improved from concern during the Baseline and Phase I periods, to potential concern during the Phase II period, to no concern during WY2023 (**Table 3-1**).

Unlike most other parameters, DO is not a direct pollutant. Instead, it is a secondary response parameter that reflects changes in the wetland ecosystem by phosphorus enrichment. It is typical in wetlands for DO levels to peak during the day when photosynthesis is occurring, and approach values less than 20% at night (DeBusk and Reddy 2003). Eutrophication of wetlands increases the presence of filamentous algae and macro-litters, which contribute to high respiratory demands. During the night, areas impacted by high nutrient loads will show DO levels on the water column near zero. SFWMD is actively implementing a

comprehensive restoration program to lower TP concentrations within the phosphorus-impacted portions of the EPA. Over time, DO concentrations at the nutrient impacted sites are expected to continue to improve as phosphorus concentrations in surface water and sediment are reduced and biological communities recover.

Using the DO SSAC, interior portions of LNWR and WCA-2 were categorized as concern, and interior portions of WCA-3 were identified as potential concern in WY2023. A summary of water quality monitoring data for the five-year period of record (POR) is presented in Appendix 3-2, and analysis of the WY2023 data is provided in Appendix 3-3 for each individual monitoring location. It should be noted that no definitive conclusions regarding differences in DO excursion rates between individual water years and previous periods can be made given the large disparity in sample sizes among periods.

## Alkalinity and pH

Alkalinity is the measure of water's acid neutralization capacity and provides a measure of the water's buffering capacity. In most surface water bodies, the buffering capacity is primarily the result of the equilibrium between carbon dioxide ( $\text{CO}_2$ ), bicarbonate ( $\text{HCO}_3^-$ ), and carbonate ions ( $\text{CO}_3^{2-}$ ). The dissociation of calcium carbonate ( $\text{CaCO}_3$ ), magnesium carbonate, or other carbonate-containing compounds entering the surface water through weathering of carbonate-containing rocks and minerals (e.g., limestone and calcite) contributes to the water's buffering capacity. Therefore, in certain areas influenced by canal inflows primarily composed of mineral-rich agricultural runoff and groundwater (such as ENP, WCA-2, and WCA-3), alkalinity concentrations are relatively high (Payne and Xue 2012). Conversely, areas such as the LNWR interior, which receive their hydrologic load primarily through rainfall, have very low alkalinities. Alkalinity (i.e.,  $\text{CaCO}_3$ ) protects against dramatic pH changes, which can be lethal to sensitive organisms. The current Class III water quality criterion specifies that alkalinity shall not be lower than 20 mg/L of alkalinity as  $\text{CaCO}_3$ , or in water bodies with natural alkalinity levels below 20 mg/L, alkalinity shall not be reduced by more than 25% of natural levels (Chapter 62-302, F.A.C.).

Excursions from the alkalinity water quality criterion have historically occurred in LNWR interior (Payne and Xue 2012). However, as discussed above and in previous SFERs, LNWR interior is hydrologically dominated by rainfall, which is naturally low in alkalinity. Natural alkalinity levels in the LNWR interior were estimated by calculating the average daily alkalinity at five interior stations (LOX3, LOX5, LOX8, LOX11, and LOX13) from WY2005 to WY2014. This POR includes the first ten years of Phase II during optimization and enhancement of BMPs and the Everglades STAs. By this time, discharges through the S-6 and S-5A/S-5AS structures had been diverted to the STAs so that untreated canal water would not enter the LNWR. Using a 95% confidence interval, it was estimated that natural alkalinity in the LNWR interior is approximately 12 mg/L  $\text{CaCO}_3$ . This estimate was used to determine the excursion rate for alkalinity at LNWR interior stations for all periods. If alkalinity was reduced by more than 25% (i.e., less than 9 mg/L  $\text{CaCO}_3$ ), it was considered an exceedance. Using this revised assessment method, only one exceedance of the criterion in the LNWR interior occurred in WY2023, which was at LOX8 in the center of LNWR. While the LNWR interior was categorized as minimal concern for alkalinity (**Table 3-1**), the overall annual average alkalinity for the area was 56 mg/L  $\text{CaCO}_3$ . Given the naturally low alkalinity in this region, FDEP does not consider instantaneous exceedances of the alkalinity criteria in this region to be violations of state water quality standards.

pH is defined as the negative  $\log_{(\text{base}10)}$  of hydrogen ion ( $\text{H}^+$ ) activity. Most organisms, especially aquatic life, function best in a pH ranging from 6.0 to 9.0, although individual species have specific ideal ranges. The current Class III water quality criterion specifies that pH should not be lowered to less than 6.0 or raised above 8.5 in predominantly fresh waters. In WY2023, pH was considered a minimal concern for the LNWR interior and WCA-3 inflow. In LNWR interior, there was one exceedance at LOX9 and one exceedance at LOX11 where pH levels fell slightly below the 6.0 minimum criteria. pH excursions within LNWR interior generally occur at sites distanced from the influence of inflows and have been linked to naturally low background alkalinity conditions and FDEP does not consider the pH excursions in this area

to be a violation of state water quality standards. For the WCA-3 inflow region, a single exceedance of the criterion occurred at S-8 where pH measured 5.9 in December 2022.

### Specific Conductance

Specific conductance (conductivity) is a measure of water's ability to conduct an electrical current and is an indirect measure of the total concentration of ionized substances (e.g.,  $\text{Ca}^{2+}$ ,  $\text{Mg}^{2+}$ ,  $\text{Na}^+$ ,  $\text{Cl}^-$ ,  $\text{HCO}_3^-$ , and  $\text{SO}_4^{2-}$ )<sup>6</sup> in the water. Conductivity varies with the quantity and type of ions present in solution. The current state water quality criteria for Class III fresh water allow for a 50% increase above background conditions in specific conductance or 1,275 micro-siemens per centimeter ( $\mu\text{S}/\text{cm}$ ), whichever is greater. This limit is meant to preserve natural background conditions and to protect aquatic organisms from stressful ion concentrations. Given that background conductivities are low within the EPA, excursions were determined using the 1,275  $\mu\text{S}/\text{cm}$  criterion (Payne and Xue 2012).

For WY2023, specific conductance was categorized as a concern for LNWR inflow, potential concern for WCA-2 inflow, and minimal concern for LNWR rim canal region and WCA-2 interior. All other areas and regions were categorized as no concern (**Table 3-1**). The number of exceedances in LNWR inflow in WY2023 (46 exceedances) decreased over WY2022 (59) and occurred at G-310 (13), ENR012 (8), G-338 (21) and S-362 (4). The S-6 structure was used as a surrogate for G-338. The G-338 structure, a diversion structure for STA-2, did not discharge into LNWR during WY2023. In the LNWR rim canal, there were two exceedances of the specific conductance criteria at Z0, located directly downstream of G-338. A total of 20 exceedances were observed for WCA-2 inflow and occurred at the G-335 (13) and G-436 (7) structures that discharge water from STA-2. Most of these exceedances were reported when no visible flow was observed. For the WCA-2 interior, there were 12 exceedances with most exceedances at 404C2 (4), 2AN6 (2) and WCA2C5 (2). Elevated conductivity levels at water control structures and stations near canal inflows may be explained by groundwater intrusion into canal surface waters (Krest and Harvey 2003, Payne and Xue 2012). This groundwater intrusion can occur due to seepage into canals via pump station operation, which can pull additional groundwater into surface water because of agricultural dewatering practices. However, improvements in water management and agricultural BMPs have reduced the occurrences of these high conductivity waters entering the EPA.

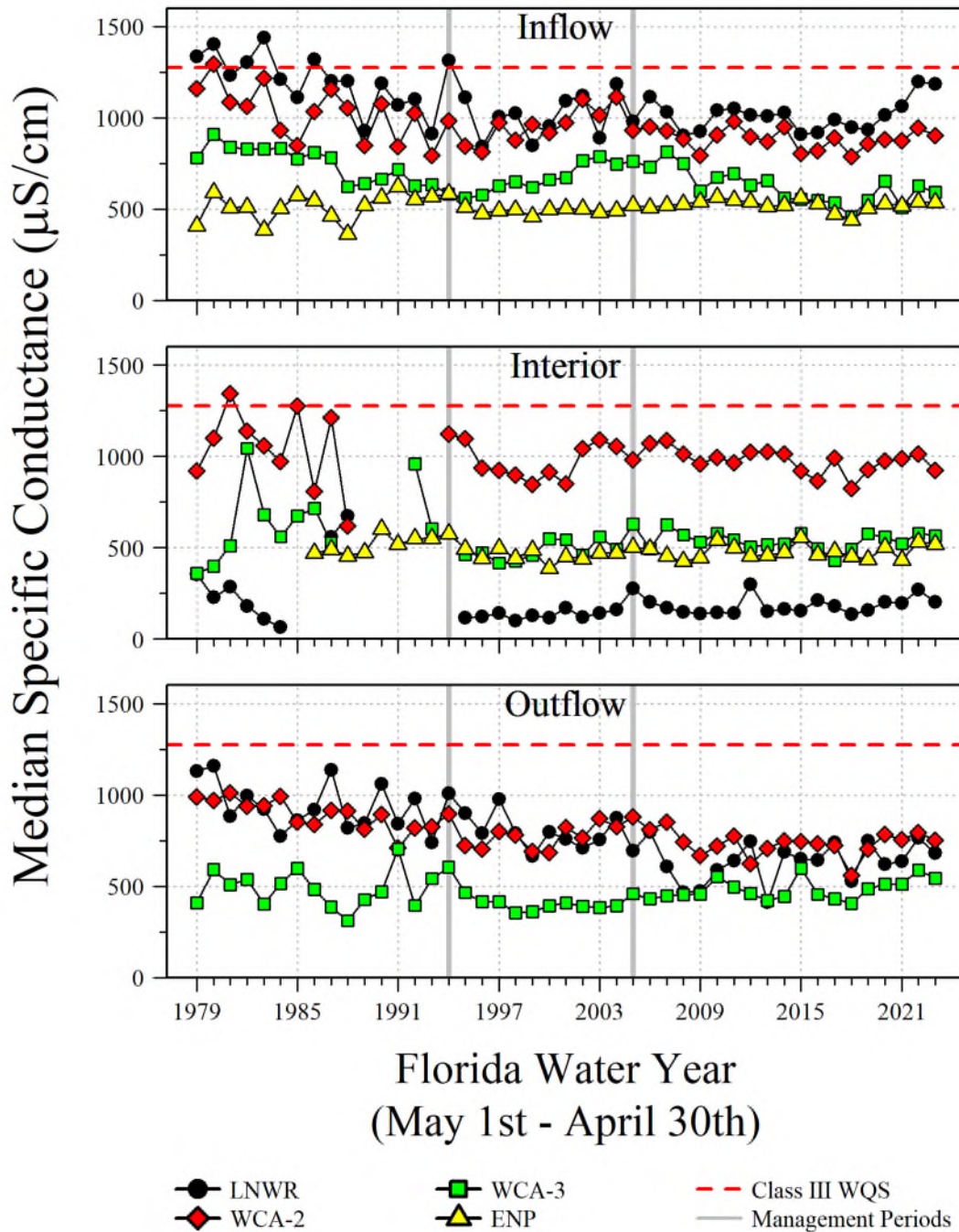
Specific conductance excursion frequency in LNWR inflows decreased from 42.9% during the Baseline period to 11.6% in Phase II and was 22.5% in WY2023. Similarly, excursion rates in the LNWR rim canal decreased from 28.1% to 2.1% from the Baseline period to Phase II and was 4.3% in WY2023. In the past, specific conductance has been categorized as a concern for WCA-2 inflows, with excursion frequencies of 25.3% and 13.5% in the Baseline and Phase I periods, respectively. Excursion rates in Phase II decreased to 5.2% (minimal concern) while for WY2023 excursion rates were 10.1% (potential concern). WCA-2 interior excursion frequency has declined from 26.2% (concern) in the Baseline period to 3.7% (minimal concern) in WY2023. Exceedances of the specific conductance criteria in WCA-3 inflows have been of minimal to no concern throughout all analysis periods and no exceedances were observed in WY2023. Throughout the entire POR, ENP inflow only experienced one exceedance observed during Phase I (**Table 3-1**).

Overall, a steady long-term decrease in specific conductance within LNWR, WCA-2, and WCA-3 inflows has occurred since WY1979 (**Figure 3-8**). Inflow median annual specific conductance levels have declined at rates (based on the Thiel-Sen Slope estimate) ranging from 6.6  $\mu\text{S}/\text{cm}$  per year for LNWR, 5.1  $\mu\text{S}/\text{cm}$  per year for WCA-2, and 5.2  $\mu\text{S}/\text{cm}$  per year for WCA-3 across the 44-year POR (WY1979–WY2023). Median annual specific conductance levels at ENP inflows have increased slightly at a rate of 0.5  $\mu\text{S}/\text{cm}$  per year but remain far below the Class III freshwater criterion. When comparing WY2023

---

$\text{Ca}^{2+}$  – calcium ion;  $\text{Mg}^{2+}$  – magnesium ion;  $\text{Na}^+$  – sodium ion;  $\text{Cl}^-$  – chloride ion;  $\text{HCO}_3^-$  – bicarbonate; and  $\text{SO}_4^{2-}$  – sulfate.

inflow median annual specific conductivity levels to WY1979 levels, LNWR declined 151  $\mu\text{S}/\text{cm}$ , WCA-2 declined 258  $\mu\text{S}/\text{cm}$ , WCA-3 declined 186.5  $\mu\text{S}/\text{cm}$ , and ENP increased 128  $\mu\text{S}/\text{cm}$ .



**Figure 3-8.** Annual median specific conductance levels in the EPA (A) inflow, (B) interior, and (C) outflow for WY1979–WY2023. (Note: WQS – Water Quality Standard.)

## Total Ammonia Nitrogen

Ammonia is the principal excretory product in aquatic animals and its mechanisms of toxicity are relatively well understood (Armstrong et al. 1978, Thurston and Russo 1981, Neil et al. 2005). The toxic effects of ammonia to aquatic species are generally considered to be caused by the un-ionized fraction ( $\text{NH}_3$ ) rather than the ionic components ( $\text{NH}_4^+$ ), which exist in equilibrium. This equilibrium is highly dependent on pH, temperature, pressure, and salinity (Hampson 1977). The current Class III freshwater water quality standard states that the 30-day average total ammonia nitrogen (TAN) concentration ( $\text{NH}_3 + \text{NH}_4^+$ ) shall not exceed the average concentration calculated from an equation (62-302.530, F.A.C.), and no single value should exceed 2.5 times the value from this equation. This standard, based on the chronic criterion calculation developed by USEPA (2013), has been adopted by the state to protect aquatic life from the toxic effects of ammonia and is not a nutrient-related criterion.

The TAN standard was applied on a rolling 30-day basis for each water year across all periods for consistency and included both the 30-day criterion and the single sample criterion. Where there were fewer than four samples in a 30-day period, the TAN concentration of a single sample was considered an exceedance if it was greater than 2.5 times the value of the equation. During WY2023, no exceedances of the TAN water quality standard were observed. Historically, there have been only 14 exceedances of the updated TAN criterion throughout all periods since WY1979, with the most recent occurring in LNWR inflow in WY2010 (**Table 3-1**).

## Pesticides

The District has been actively monitoring pesticides since 1976 (Pfeuffer 1985) and, since 1984, has established a routine pesticide monitoring program (Pfeuffer and Rand 2004). The pesticide monitoring network includes sites designated in permits for Lake Okeechobee operations and Non-Everglades Construction Projects (Non-ECP). Results of monitoring conducted as part of these permits are provided in Volume III of the annual SFER. The current EPA monitoring program consists of 14 stations and is conducted on a biannual basis. A subset of sampling stations from the entire pesticide monitoring network was used for analysis.

Surface water concentrations of pesticides are regulated under criteria presented in Chapter 62-302, F.A.C. Chemical-specific numeric criteria for several pesticides and herbicides (e.g., dichlorodiphenyltrichloroethane [DDT] and malathion) are listed in Section 62-302.530, F.A.C. Compounds not specifically listed, including many contemporary pesticides (e.g., ametryn, atrazine, and diazinon), are evaluated based on acute and chronic toxicity. A set of toxicity-based guidelines for non-listed pesticides was presented by Weaver (2001). These guidelines were developed based on the requirement in Subsection 62-302.530(62), F.A.C., which calls for Florida's surface waters to be free from "substances in concentrations, which injure, are chronically toxic to, or produce adverse physiological or behavioral response in humans, plants, or animals."

Surface water pesticide data is typically collected biannually for most monitoring locations within the network. Compliance with pesticide water quality standards is assessed annually; therefore, only WY2023 data are presented. During WY2023, 18 pesticides or pesticide breakdown products were detected at concentrations above their respective MDLs within the EPA. These compounds include 2,4-D, ametryn, atrazine, alpha-Hexachlorocyclohexane, bromacil, diazinon, dieldrin, diuron, hexazinone, imidacloprid, metalaxyl, metolachlor, metribuzin, norflurazon, prometryn, simazine, toxaphene and atrazine-desethyl. Some toxaphene samples exceeded Class III water quality standards (**Table 3-2**). Toxaphene exceedance was recorded at S8. Toxaphene has been used in the past for pest control on cotton crops. It was also used to control pests in livestock and poultry, and on other field crops. In 1982, EPA cancelled the registration of toxaphene for use as a pesticide, except for use on livestock. Other compounds were detected but did not exceed any water quality standard or guideline concentrations (**Table 3-2**).

**Table 3-2.** Surface water pesticide concentrations detected in WY2023.

Area	Parameter	Class III Water Quality Criteria or Guideline (µg/L) <sup>a</sup>	Arithmetic Mean Concentration (µg/L)	Range (µg/L)	Total Samples	Percent Above MDL (Percent Exceed Water Quality Criteria or Guideline)
LNWR	2,4-D	80	0.0044	0.0035 - 0.0054	2	100 (0)
	Ametryn	6.2	0.0275	0.0150 - 0.0400	2	100 (0)
	Atrazine	1.8	0.0625	0.0500 - 0.0750	2	100 (0)
	Atrazine Desethyl	--- <sup>b</sup>	0.0037	0.0026 - 0.0048	2	100 (0)
	Diazinon	0.17	0.0003	0.0003 - 0.0003	1	100 (0)
	Hexazinone	1020	0.0050	0.0048 - 0.0053	2	100 (0)
	Imidacloprid	--- <sup>b</sup>	0.0016	0.0010 - 0.0022	2	50 (0)
	Metalaxyl	36.5	0.0013	0.0004 - 0.0022	2	50 (0)
	Metolachlor	1.08	0.0192	0.0024 - 0.0360	2	100 (0)
	Metribuzin	64	0.0116	0.0072 - 0.0160	2	100 (0)
Prometryn	21	0.0062	0.0015 - 0.0110	2	100 (0)	
WCA-2	2,4-D	80	0.0032	0.0032 - 0.0032	1	100 (0)
	Ametryn	6.2	0.0130	0.0130 - 0.0130	1	100 (0)
	Atrazine	1.8	0.0160	0.0160 - 0.0160	1	100 (0)
	Atrazine Desethyl	--- <sup>b</sup>	0.0040	0.0040 - 0.0040	1	100 (0)
	Hexazinone	1020	0.0037	0.0037 - 0.0037	1	100 (0)
	Metalaxyl	36.5	0.0015	0.0015 - 0.0015	1	100 (0)
	Metolachlor	1.08	0.0022	0.0022 - 0.0022	1	100 (0)
	Metribuzin	64	0.0033	0.0033 - 0.0033	1	100 (0)
Prometryn	21	0.0008	0.0008 - 0.0008	1	100 (0)	
WCA-3	2,4-D	80	0.0319	0.0024 - 0.1300	20	100 (0)
	Ametryn	6.2	0.0096	0.0003 - 0.0310	20	95 (0)
	Atrazine	1.8	0.0346	0.0024 - 0.0900	20	100 (0)
	Atrazine Desethyl	--- <sup>b</sup>	0.0041	0.0007 - 0.0098	20	75 (0)
	BHC, alpha	0.0116	0.0001	0.0000 - 0.0001	19	5 (0)
	Bromacil	97	0.0005	0.0002 - 0.0038	20	10 (0)
	Dieldrin	0.0019	0.0002	0.0002 - 0.0006	19	11 (0)
	Diuron	8	0.0029	0.0010 - 0.0310	20	15 (0)
	Hexazinone	1020	0.0024	0.0002 - 0.0050	20	90 (0)
	Imidacloprid	--- <sup>b</sup>	0.0026	0.0010 - 0.0190	20	30 (0)
	Metolachlor	1.08	0.0019	0.0002 - 0.0085	20	65 (0)
	Metribuzin	64	0.0104	0.0002 - 0.1100	20	65 (0)
	Norflurazon	815	0.0029	0.0002 - 0.0150	20	65 (0)
	Prometryn	21	0.0002	0.0001 - 0.0005	20	25 (0)
Simazine	5.8	0.0003	0.0001 - 0.0013	20	5 (0)	
Toxaphene	0.0002	0.0094	0.0075 - 0.0460	20	5 (100) <sup>c</sup>	

**Table 3.2.** Continued.

Area	Parameter	Class III Water Quality Criteria or Guideline (µg/L) <sup>a</sup>	Arithmetic Mean Concentration (µg/L)	Range (µg/L)	Total Samples	Percent Above MDL (Percent Exceed Water Quality Criteria or Guideline)
ENP	2,4-D	80	0.0137	0.0010 - 0.1900	16	19 (0)
	Ametryn	6.2	0.0013	0.0003 - 0.0062	16	25 (0)
	Atrazine	1.8	0.0075	0.0006 - 0.0480	14	100 (0)
	Atrazine Desethyl	--- <sup>b</sup>	0.0015	0.0006 - 0.0057	16	25 (0)
	Bromacil	97	0.0003	0.0002 - 0.0007	16	12 (0)
	Diuron	8	0.0013	0.0010 - 0.0060	16	6 (0)
	Hexazinone	1020	0.0007	0.0002 - 0.0027	16	44 (0)
	Imidacloprid	--- <sup>b</sup>	0.0014	0.0010 - 0.0074	16	6 (0)
	Metalaxyl	36.5	0.0004	0.0003 - 0.0008	16	6 (0)
	Metolachlor	1.08	0.0008	0.0002 - 0.0037	16	56 (0)

<sup>a</sup> Class III freshwater water quality standard as referenced in 62-302.530, F.A.C. and toxicity-based guidelines for non-listed pesticides was presented by Weaver (2001).

<sup>b</sup> No class III criteria or guideline.

<sup>c</sup> Identified as concern.

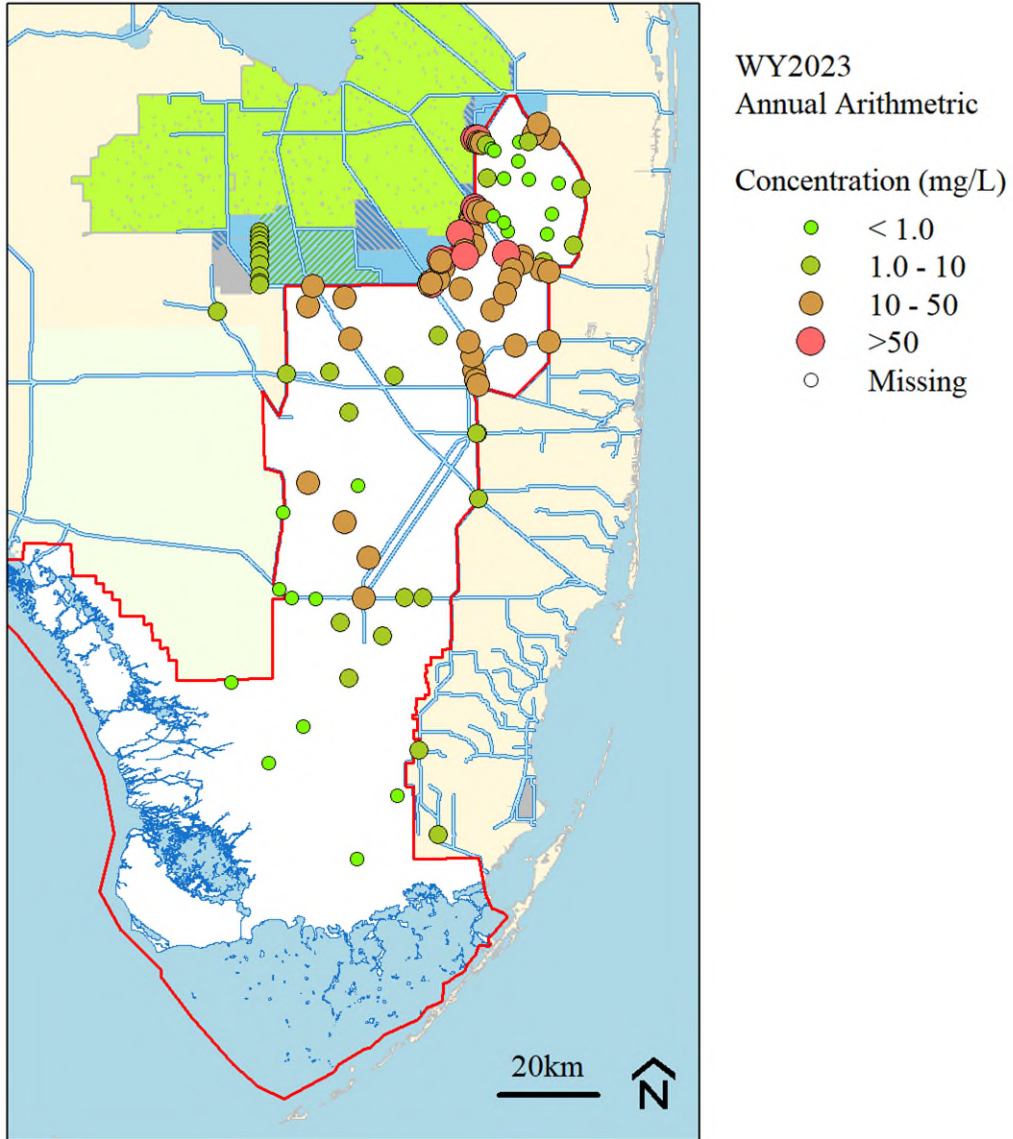
## Sulfate

Sulfur is an essential plant macro-nutrient (Bellinger and Van Mooy 2012) that enters the Everglades ecosystem primarily as sulfate (Orem et al. 2011) from various sources including the oxidation of peat soil, fertilizer application, atmospheric deposition, and groundwater. Sulfate concentrations follow a general north-to-south concentration gradient at inflow locations similar to other nutrients assessed in this report (**Table 3-3** and **Figure 3-9**). Sulfate monitoring results are presented in this section to provide an overview of current concentrations and evaluate temporal and spatial patterns. Sulfate summary statistics relative to the Baseline (WY1979–WY1993), Phase I (WY1994–WY2004), Phase II (WY2005–WY2022), and current water year (WY2023) are shown in **Table 3-3**.

Over the past several decades, multiple research studies have been conducted to understand factors that influence the formation of methylmercury (MeHg) in aquatic and semi-aquatic environments, particularly within the Everglades ecosystem. Because of its large size, extensive wetlands, and relatively high rates of mercury deposition, the Everglades is considered sensitive to mercury methylation and subsequent bioaccumulation in the food web (Wiener et al. 2003). A suite of peer-reviewed and technical publications has been produced exploring the underlying biogeochemical regulation of MeHg production within natural systems. Many of these with relevance to South Florida focus on a unimodal relationship of sulfur/sulfate and MeHg production (Gilmour et al. 1992, Benoit et al. 1999a, 1999b, 2003, Axelrad et al. 2008, 2013, Orem et al. 2011). Other studies have found that the sulfate-mercury unimodal relationship is not temporally or spatially consistent within the Everglades. At the landscape scale, mercury methylation is subject to large unexplained variations and appears to be influenced by sulfate and other environmental factors (Gilmour 2011, Julian et al. 2014a, 2015a). The complexities of MeHg production must be well understood and predictable for a control or management strategy to be effective. However, based on previous and ongoing research, sulfate can influence the mercury-MeHg cycle under some suite of ambient conditions.

**Table 3-3.** Summary statistics of sulfate concentrations in mg/L for the Baseline (WY1979–WY1993), Phase I (WY1994–WY2004), Phase II (WY2005–WY2022), and WY2023 periods. Values for inflow and outflow classes were calculated regardless of flow conditions.

Region	Class	Water Year Period	Sample Size	Arithmetic Mean	Standard Deviation	Minimum	Maximum
LNWR	Inflow	1979-1993	461	73.0	56.9	7.10	456
		1994-2004	649	54.3	34.0	1.00	461
		2005-2022	2,286	55.1	22.7	2.40	199
		2023	106	64.9	21.8	28.50	147
	Interior	1979-1993	340	16.6	21.5	2.50	220
		1994-2004	1,205	5.0	11.5	<0.05	110
		2005-2022	4,591	6.0	12.7	<0.05	95
		2023	811	6.1	13.4	<0.05	76
	Outflow	1979-1993	158	53.5	59.5	7.30	571
		1994-2004	232	44.9	32.5	1.42	419
		2005-2022	963	35.7	20.4	1.20	101
		2023	55	37.8	24.4	4.90	87
	Rim	1979-1993	66	42.2	37.1	2.5	140
		1994-2004	345	57.2	26.9	1.6	210
		2005-2021	524	49.1	20.2	0.2	185
		2022	48	58.9	22.3	25.8	100
WCA-2	Inflow	1979-1993	219	63.8	79.0	7.30	945
		1994-2004	301	49.1	29.4	6.15	419
		2005-2022	1,805	42.8	17.7	<0.05	106
		2023	129	43.3	18.2	5.40	87
	Interior	1979-1993	633	42.9	37.1	2.50	344
		1994-2004	1,269	43.8	23.9	0.10	180
		2005-2022	2,603	44.4	17.4	<0.05	128
		2023	284	42.5	14.5	0.10	71
	Outflow	1979-1993	318	40.8	30.8	2.50	224
		1994-2004	275	30.0	13.2	2.34	73
		2005-2022	1,212	28.7	14.7	2.30	86
		2023	63	29.5	12.7	10.40	53
WCA-3	Inflow	1979-1993	551	32.1	32.7	1.00	286
		1994-2004	496	20.4	14.6	0.50	73
		2005-2022	3,425	14.2	17.8	<0.05	224
		2023	241	11.9	16.3	<0.05	66
	Interior	1979-1993	450	14.9	17.3	2.00	262
		1994-2004	1,620	10.8	34.8	<0.05	1,300
		2005-2022	1,727	12.9	15.1	<0.05	126
		2023	226	9.4	11.0	<0.05	39
	Outflow	1979-1993	279	17.7	19.0	1.00	113
		1994-2004	340	6.6	9.2	<0.05	48
		2005-2022	1,124	8.0	10.4	<0.05	58
		2023	51	7.0	9.2	<0.05	38
ENP	Inflow	1979-1993	268	17.4	19.1	1.00	113
		1994-2004	276	5.1	6.2	<0.05	36
		2005-2022	826	6.3	7.3	<0.05	46
		2023	39	5.6	6.3	<0.05	22
	Interior	1979-1993	572	9.0	19.5	0.75	206
		1994-2004	864	5.5	17.7	<0.05	403
		2005-2022	1,157	4.4	17.4	<0.05	242
		2023	234	2.0	2.2	<0.05	8



**Figure 3-9.** Annual arithmetic mean sulfate (SO<sub>4</sub><sup>2-</sup>) concentrations for all classifications at stations across the EPA in WY2023. Annual arithmetic mean values for structures are represented as annual arithmetic mean regardless of flow conditions.

Annual mean sulfate concentrations observed at inflow regions to the EPA during WY2023 ranged from 64.9 mg/L for LNWR inflows, to 43.3 mg/L for WCA-2, 11.9 mg/L for WCA-3, and 5.6 mg/L for ENP (**Table 3-3**). As noted in previous years, sulfate concentrations at the S-333 and S-333N structures were elevated compared to the other inflows to ENP (**Figure 3-9**). The annual mean sulfate concentrations decreased for WCA-3 outflow and ENP inflow from WY2022 (7.4 and 6.1 mg/L, respectively) to WY2023 (7.0 and 5.6 mg/L, respectively). Inflow annual mean sulfate concentrations for the POR significantly declined for WCA-2, WCA-3 and ENP (**Table 3-4**). This declining trend was also observed during the Phase II period for WCA-3 inflows. Understanding temporal trends in sulfate across multiple monitoring stations is challenging due to incomplete data sets and inconsistencies in sample collection frequency between stations and years. For example, the number of samples collected for sulfate at ENP inflow stations increased from 9 samples in WY2015 to 39 samples in WY2023 (**Table 3-3**). Additionally, S-333 monitoring frequency increased to weekly or biweekly while all other stations retained quarterly or no collections. Therefore, S-333 is highly influential in the ENP inflow Phase II period because of an increase in sample collection volume.

**Table 3-4.** Kendall's  $\tau$  annual arithmetic mean sulfate concentration trend analysis results for each region's inflow and interior classification within the EPA for the entire POR (WY1979–WY2023) and the period of WY2005 to present.

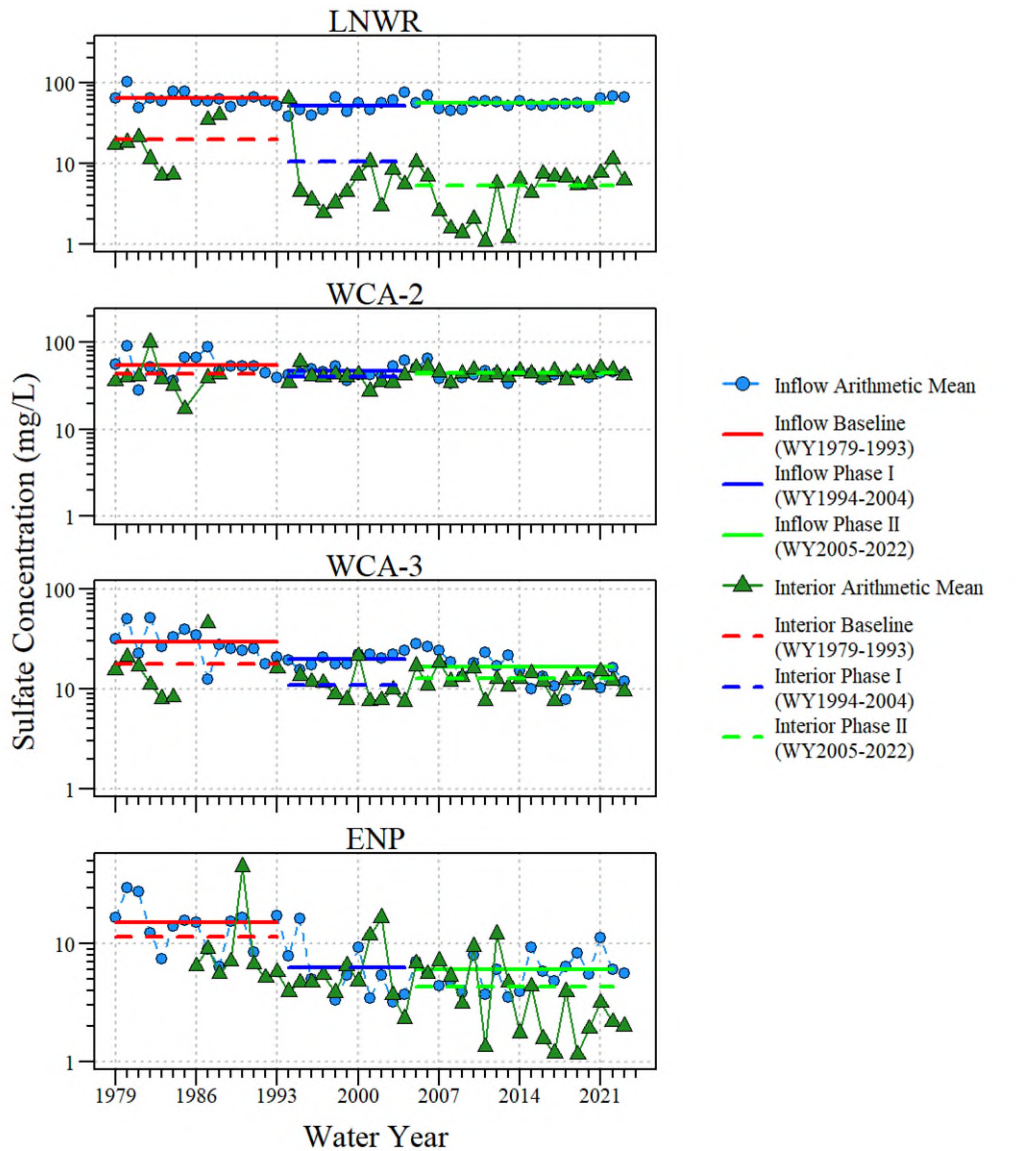
Class	Area	POR (WY1979–WY2023)			Phase II & Current Water Year (WY2005–WY2023)		
		Kendall's $\tau$	$p$ -value	Sen's Slope Estimate <sup>a</sup>	Kendall's $\tau$	$p$ -value	Sen's Slope Estimate <sup>a</sup>
Inflow	LNWR	-0.11	0.3	-0.12	0.20	0.24	0.54
	WCA-2	-0.27	<0.01	-0.28	-0.05	0.78	-0.07
	WCA-3	-0.53	<0.01	-0.44	-0.59	<0.01	-0.78
	ENP	-0.33	<0.01	-0.20	0.20	0.24	0.08
Interior	LNWR	-0.23	<0.05	-0.16	0.25	0.14	0.23
	WCA-2	0.26	<0.05	0.18	-0.08	0.68	-0.09
	WCA-3	-0.10	0.4	-0.04	-0.20	0.24	-0.14
	ENP	-0.42	<0.01	-0.14	-0.42	<0.05	-0.25

a. Expressed as mg/L per water year.

The annual mean sulfate concentrations entering the LNWR have remained relatively consistent throughout the POR and Phase II period (**Table 3-4**). This could be due to the natural conditions that exist in the eastern portion of the EAA and EPA. Historically, this region's water quality within the surficial aquifer is affected by saltwater intrusion and highly mineralized groundwater. Highly mineralized groundwater in this region is typically associated with ancient connate seawater, which was the result of the interglacial seas that inundated the area during the Pleistocene Epoch (Miller 1988). As noted in Axelrad et al. (2013), connate seawater could potentially be a relatively large source of sulfate, chloride, and dissolved solids (i.e., other minerals) to the EPA, more specifically to the LNWR. Another driving factor of interior trends are the biogeochemical processes associated with marsh dry-out. During relatively dry periods, when water levels in the marsh recede below the soil surface, oxidation of organic matter occurs readily. Once the area is reflooded, a large upward flux of nutrients occurs including sulfate from the soil to the water column. This dry-out and flux phenomena explains the relatively high annual concentrations experienced during the extremely dry period in the mid-1980s and the relatively dry period during the early to mid-2000s.

During WY2023, the WCA-2 interior had the highest annual mean sulfate concentration of 42.5 mg/L and ENP had the lowest annual mean sulfate concentration of 2.0 mg/L. Interior annual mean sulfate

concentrations significantly declined in LNWR and ENP across the entire POR (**Table 3-4**). During the Phase II period, interior annual mean sulfate concentrations declined significantly for ENP. Meanwhile, qualitative comparison of period-wide concentrations indicates sulfate concentrations within LNWR decreased between the Phase I and Phase II periods (**Figure 3-10**), which could possibly be due to the construction and operation of STA-1 West and STA-1 East in combination with the rainfall-driven hydrology. Sulfate removal in the STAs occurs primarily through microbial sulfate reduction and plant uptake but only a small percentage of sulfur entering the Everglades STAs is retained (Gabriel et al. 2008). Other factors influencing this trend of decreasing sulfate concentrations with time could be changes in water management, establishing and managing BMPs within the EAA, and potential decreasing application of elemental sulfur as a soil amendment to agricultural fields in this region of the EAA (Landing 2014).



**Figure 3-10.** Annual arithmetic mean sulfate concentrations for inflow and interior areas of LNWR, WCA-2, WCA-3, and ENP for WY1979–WY2023. The horizontal lines indicate the mean annual geometric mean TP concentrations for the Baseline (WY1979–WY1993), Phase I (WY1994–WY2004), and Phase II (WY2005–WY2022) periods. (Note: The y-axis is on a log-scale).

Notably, implemented BMPs were not used to directly mitigate sulfate but rather nutrient and sediment run-off. It is also important to note that sulfate concentrations within LNWR spiked after extended periods of dry conditions as observed between WY1985 and WY1994. This was not just isolated to water column sulfate, but phosphorus and nitrogen also spiked during these periods. The very low sulfate concentrations observed for the interior portion of LNWR indicate that either assimilation of sulfate is occurring and potentially could be in growth-limiting concentrations due to its low availability (Bellinger and Van Mooy 2012), or very little to no high-sulfate water reached the interior portion (due to topography and hydrology) of LNWR and these low concentrations reflect deposition-driven water column concentrations. To further understand marsh sulfate dynamics, sulfur speciation and a more in-depth analysis of iron biogeochemistry is needed.

Annual mean sulfate concentrations within WCA-2 are higher than other regions within the EPA. Historical EAA runoff containing both local and regional inputs of sulfate and a prolonged legacy effect is suspected in causing these internal sulfate dynamics. The hydrology of WCA-2A spreads the canal inflow broadly and WCA-2 soils have relatively high nutrient concentrations including sulfur (Everglades soil mapping data, Reddy et al. 2005). These high concentrations of sulfur within the soils could result in enhanced internal sulfur loading, which could explain why interior mean concentrations are higher than inflow mean concentrations. Due to these relatively high marsh concentrations within eutrophic/impacted portions of WCA-2, it is reasonable to suggest growth of biota within this region of the EPA is not sulfur or sulfate growth limited and corroborate results presented by Bellinger and Van Mooy (2012).

## PHOSPHORUS AND NITROGEN

Phosphorus and nitrogen are essential to the existence and growth of aquatic organisms in surface waters. The EPA and, to a larger extent, the Everglades ecosystem, is a phosphorus-limited system (Noe et al. 2001). The native flora and fauna in the Everglades are adapted to nutrient poor conditions; therefore, relatively small additions of nutrients, especially phosphorus, have dramatic effects on the ecosystem.

Until the adoption of the numeric TP criteria, both phosphorus and nitrogen concentrations in EPA surface waters were only regulated by Class III narrative criterion. The narrative criterion specifies that nutrient concentrations in a water body cannot be altered to cause an imbalance in the natural populations of aquatic flora or fauna. Because of the importance of phosphorus in controlling natural biological communities, FDEP has numerically interpreted the narrative criterion, as directed by the EFA, to establish a long-term geometric mean of 10 µg/L TP for the EPA. Currently, nitrogen does not have a numeric criterion and is still regulated by only the narrative criteria.

In addition to presenting analyses of individual TP and TN concentration, this section provides an evaluation of spatial and temporal trends in nutrient concentration and loads within the EPA as measured during WY2023 and compares the results with previous monitoring periods to provide an overview of the changes in nutrient levels within the EPA.

### Total Phosphorus Concentrations

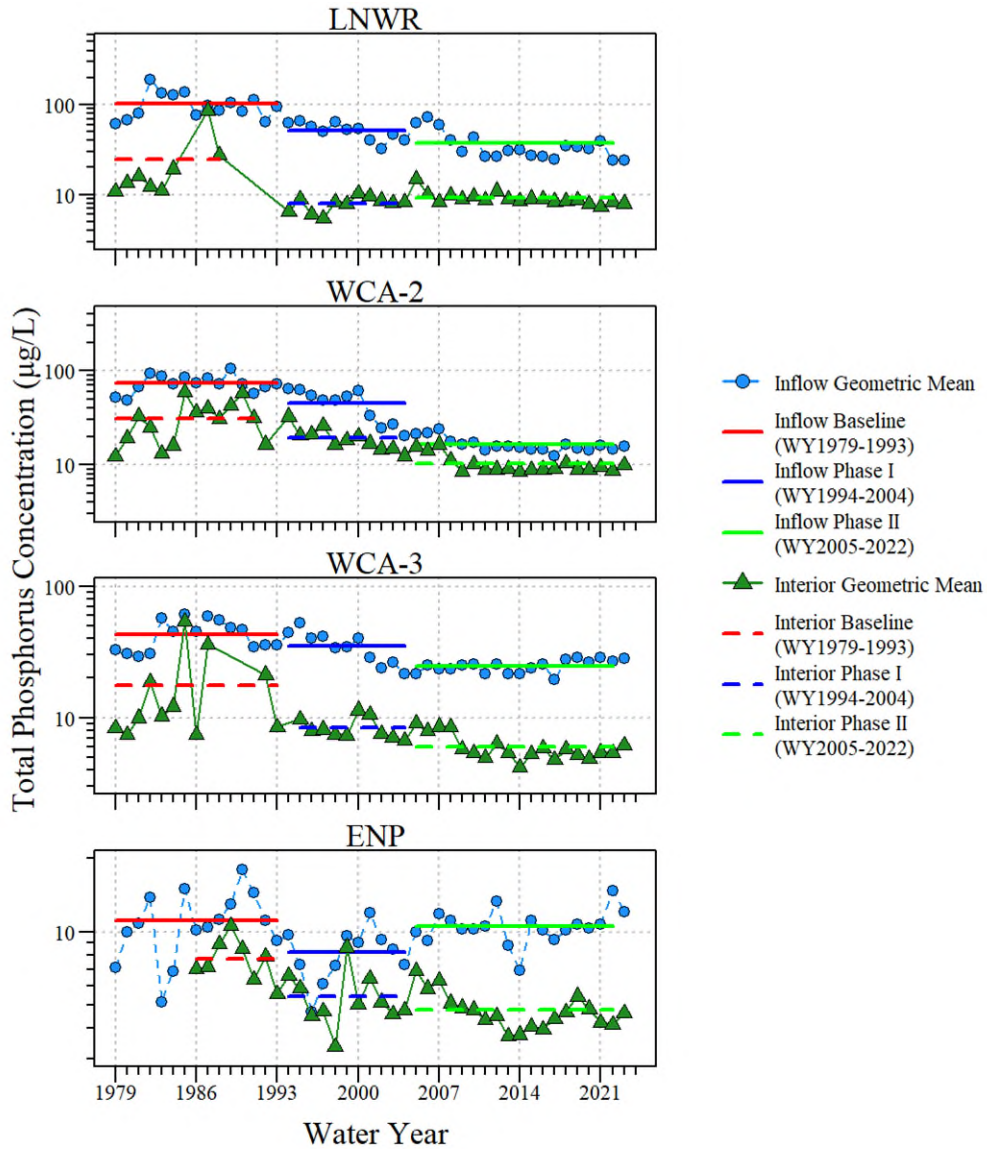
One of the primary objectives of this chapter is to document temporal changes in TP concentrations across the EPA using long-term geometric means to summarize and compare TP concentrations in accordance with the EFA and TP Rule criterion requirements. The EFA and TP Rule criterion were designed to provide long-term, ecologically protective conditions and require the use of geometric means due to the log-normal distribution of natural TP concentrations in the environment. The geometric mean employed by the criterion and the methodology used in this chapter to assess the nutrient concentrations account for short-term variability in water quality data, while providing more reliable, long-term values for evaluation and comparison of nutrient status.

Temporal changes in annual geometric mean TP concentrations during the WY1979–WY2023 POR at both inflow and interior sites of LNWR, WCA-2, WCA-3, and ENP are shown in **Figure 3-11** relative to

average geometric mean TP concentrations for the Baseline, Phase I, Phase II, and WY2023 periods. A descriptive statistics summary of TP concentrations measured within each portion of the EPA during the Baseline, Phase I, Phase II, and WY2023 periods is provided in **Table 3-5**.

During the Baseline period, annual geometric mean TP concentrations at inflow and interior marsh sites across the EPA reached peak historic concentrations and were highly variable, as shown in **Figure 3-11**. As the agricultural BMP and Everglades STA programs were initiated and became operational during the Phase I period, annual mean TP concentrations were reduced markedly and became less variable compared to levels observed during the Baseline period. Additionally, due to extreme climatic events and low water elevations during the mid-1980s, TP concentrations remained relatively high, while the 1990s experienced higher water levels and lower TP concentrations (McCormick and Stevenson 1998; also, see Appendix 2A-3 of this volume). Effectiveness of continued optimization and enhancement of BMPs and STAs on phosphorus concentrations and loads during Phase II has been difficult to assess due to climatic extremes that have occurred during this period. The Phase II period has been punctuated with periodic extreme climatic events ranging from active hurricane seasons with intense rainfall to periods of extended drought with little or no rainfall leading to extensive dry-out conditions throughout the EPA.

The greatest effect from climatic extremes was experienced during WY2005, WY2006, and WY2018 when tropical activity resulted in elevated inflow concentrations. This, combined with intense storm surge and resulting deposition of marine sediment in the lower portions of ENP and storm damage to Everglades STA vegetative communities. Additionally, it resulted in decreased STA nutrient removal for many months (Pietro et al. 2006, Castañeda-Moya et al. 2010). Decreased rainfall in WY2005 led to prolonged periods of marsh dry-out, which resulted in increased oxidation of the organic sediment and the subsequent release of phosphorus into the water column. This release, in turn, led to elevated TP concentrations at marsh sites across the EPA. After the 2004 and 2005 hurricane seasons, much of the EPA experienced varying levels of recovery from the climatic events. However, in recent years, extreme climatic periods have resulted in extensive drought and excessive rainfall causing dry-out and flood conditions that have significantly influenced annual TP concentrations observed across the ecosystem (**Figure 3-11**). As the Phase II BMP and STA implementation period is expanded, statistical results will most likely be influenced less by single atypical years (e.g., WY2005), and the long-term effects of continuing restoration efforts will become more apparent.

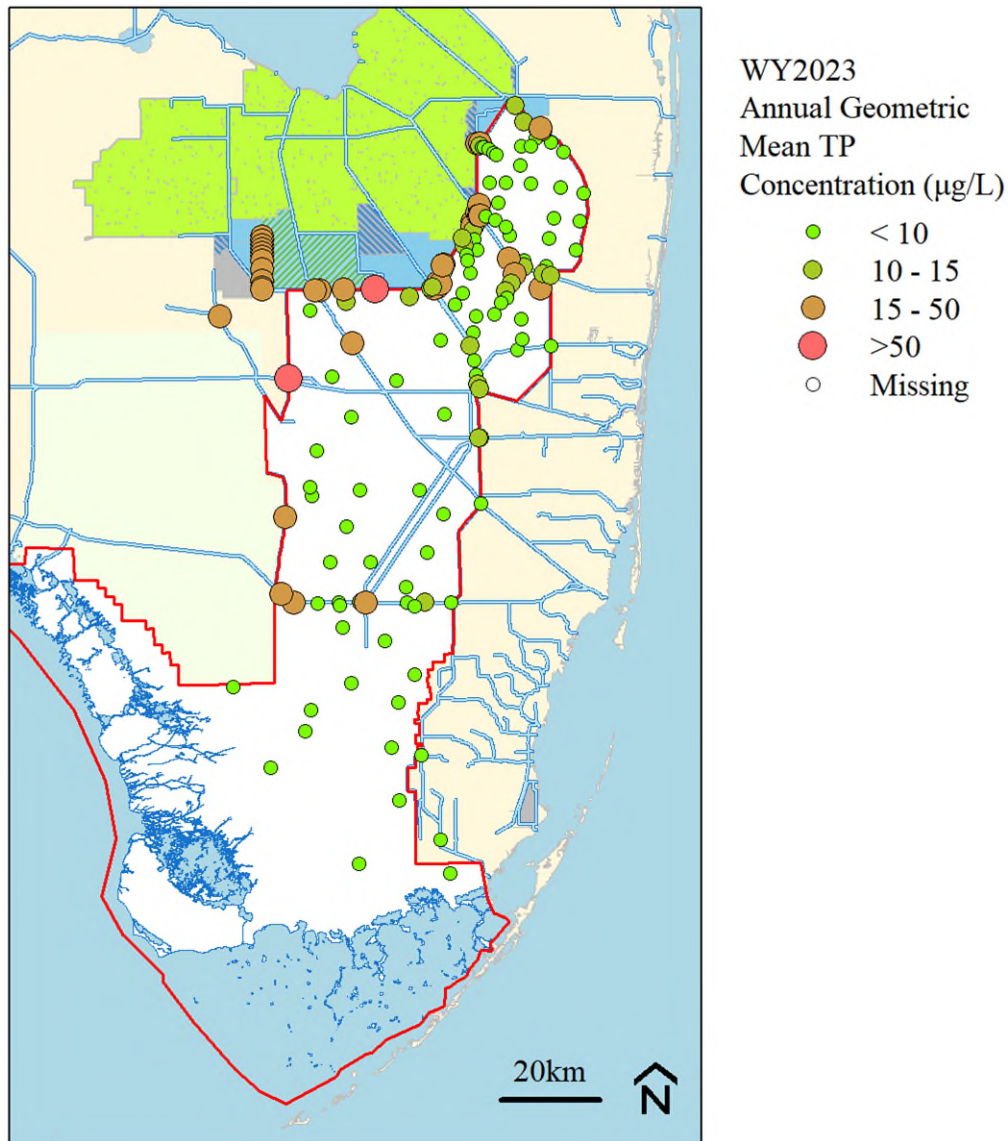


**Figure 3-11.** Annual geometric mean TP concentrations for inflow and interior areas of LNWR, WCA-2, WCA-3, and ENP for WY1979–WY2023. The horizontal lines indicate the mean annual geometric mean TP concentrations for the Baseline (WY1979–WY1993), Phase I (WY1994–WY2004), and Phase II (WY2005–WY2022) periods. (Note: The y-axis is on a log-scale).

**Table 3-5.** Summary statistics of TP concentrations in µg/L for the Baseline (WY1979–WY1993), Phase I (WY1994–WY2004), Phase II (WY2005–WY2022), and WY2023 periods.

Region	Class	Water Year Period	Sample Size	Geometric Mean	Geometric Standard Deviation	Median	Minimum	Maximum
LNWR	Inflow	1979-1993	1,686	86.8	2.3	92	<2	1,415
		1994-2004	3,326	49.8	2.1	49	2	799
		2005-2022	4,536	35.3	2.0	30	3	987
		2023	234	24.0	1.6	21	10	220.5
	Interior	1979-1993	381	13.7	2.7	12	<2	494
		1994-2004	1,493	7.9	1.6	8	2	80
		2005-2022	6,048	9.1	1.7	8	2	574
		2023	339	8.0	1.5	8	4	41
	Outflow	1979-1993	613	64.4	2.1	62	8	1,347
		1994-2004	701	45.4	1.9	43	6	495
		2005-2022	1,180	18.6	1.8	16	6	515
		2023	55	12.6	1.3	12	8	27
	Rim	1979-1993	96	84.5	1.8	89	22	473
		1994-2004	479	67.5	1.8	68	2	263
		2005-2022	685	30.6	2.0	25	10	817
		2023	46	18.6	1.4	18	13	41
WCA-2	Inflow	1979-1993	852	68.7	2.0	68	8	1,347
		1994-2004	1,332	37.0	2.1	38	7	493
		2005-2022	3,689	16.1	1.6	15	<2	245
		2023	268	15.8	1.5	15	8	79.5
	Interior	1979-1993	1,506	24.0	3.8	20	<2	3,189
		1994-2004	1,464	18.0	3.2	12	<2	5,652
		2005-2022	4,431	9.7	2.0	8	<2	278
		2023	239	10.0	1.9	9	<2	112
	Outflow	1979-1993	1,384	18.5	2.7	17	<2	556
		1994-2004	1,020	14.9	2.2	14	2	199
		2005-2022	1,769	11.2	1.7	11	3	222
		2023	84	9.5	1.5	10	4	23
WCA-3	Inflow	1979-1993	2,355	41.5	2.6	43	<2	933
		1994-2004	4,732	30.5	2.4	28	2	1,286
		2005-2022	14,364	24.6	2.0	23.5	3	1,378.5
		2023	899	28.2	1.9	28	4	976
	Interior	1979-1993	592	10.4	3.1	10	<2	438
		1994-2004	1,909	8.2	2.1	8	<2	310
		2005-2022	3,475	5.9	1.8	6	<2	180
		2023	183	6.1	1.6	6	2	41
	Outflow	1979-1993	1,958	12.4	2.3	12	<2	593
		1994-2004	2,563	10.6	2.0	10	2	171
		2005-2022	14,286	13.2	1.8	12	<2	8,540
		2023	1,195	14.6	1.7	14	4	409
ENP	Inflow	1979-1993	2,305	10.8	2.3	10	<2	593
		1994-2004	3,309	8.1	1.9	8	2	297
		2005-2022	19,673	10.7	1.8	10	<2	8,540
		2023	1,497	12.1	1.8	12	3	409
	Interior	1979-1993	505	7.7	2.7	7	2	521
		1994-2004	926	5.2	2.1	5	<2	117
		2005-2022	2,066	4.5	1.8	4	<2	291
		2023	162	4.6	1.6	5	2	14

As documented in previous years, annual geometric mean TP concentrations measured during WY2023 exhibited a general north-south concentration gradient. Annual geometric mean inflow concentrations observed during WY2023 ranged from 12.1  $\mu\text{g/L}$  for ENP to 28.2 $\mu\text{g/L}$  for WCA-3. Inflow concentrations observed for WCA-2 and LNWR were 15.8 and 24  $\mu\text{g/L}$ , respectively. The north-to-south gradient is due to spatial differences in upstream basin-wide water quality combined with canals used to transport phosphorus-rich surface water composed primarily of agricultural runoff originating in the EAA that enter the northern portions of the EPA. Settling, sorption (both adsorption and absorption), biological assimilation, and other biogeochemical processes result in decreasing concentrations as the water flows southward through the marsh (**Figure 3-12**). A detailed, site-specific summary of the TP concentrations for WY2023 is provided in Appendix 3-4 of this volume.



**Figure 3-12.** Annual geometric mean TP concentrations for all classifications for WY2023 at stations across the EPA. Annual geometric mean values for structures are represented as annual geometric mean regardless of flow conditions.

The overall temporal trend of geometric mean inflow TP concentrations observed during each distinct period (**Table 3-5**) and across the long-term POR (WY1979–WY2023) and the short-term Phase II period (WY2005–WY2023) shows a varying magnitude in change across the system. Between periods (i.e., Baseline, Phase I, and Phase II) a distinct step-down trend is apparent (**Figure 3-11**) for LNWR, WCA-2 and WCA-3. These step-down trends between periods is also reflected in annual trend results with statistically significant downward trends for LNWR, WCA-2, and WCA-3 during the entire POR (**Table 3-6**). During the Phase II period, annual inflow trends were significantly declining for LNWR and WCA-2, while WCA-3 showed a significant trend increase with a Sen’s slope of 0.26 (**Table 3-6**). Despite some year-to-year variability observed during the different periods, inflow geometric mean TP concentrations for ENP did not exhibit a clear trend between periods (**Figure 3-11**) or statistically significant annual trends over the POR or Phase II period (**Table 3-6**).

**Table 3-6.** Kendall’s  $\tau$  annual geometric mean TP concentration trend analysis results for each region’s inflow and interior classification within the EPA for the entire POR (WY1979–WY2023) and Phase II to current water year (WY2005–2023).

Class	Area	POR (WY1979–WY2023)			Phase II & Current Water Year (WY2005–WY2023)		
		Kendall's $\tau$	$\rho$ -value	Sen's Slope Estimate <sup>a</sup>	Kendall's $\tau$	$\rho$ -value	Sen's Slope Estimate <sup>a</sup>
Inflow	LNWR	-0.68	<0.01	-1.77	-0.42	<0.05	-1.40
	WCA-2	-0.75	<0.01	-1.85	-0.46	<0.01	-0.27
	WCA-3	-0.46	<0.01	-0.54	0.40	<0.05	0.26
	ENP	0.09	0.39	0.024	0.19	0.27	0.06
Interior	LNWR	-0.32	<0.01	-0.08	-0.60	<0.01	-0.12
	WCA-2	-0.60	<0.01	-0.51	-0.27	0.11	-0.08
	WCA-3	-0.58	<0.01	-0.15	-0.35	<0.05	-0.10
	ENP	-0.50	<0.01	-0.09	-0.35	<0.05	-0.07

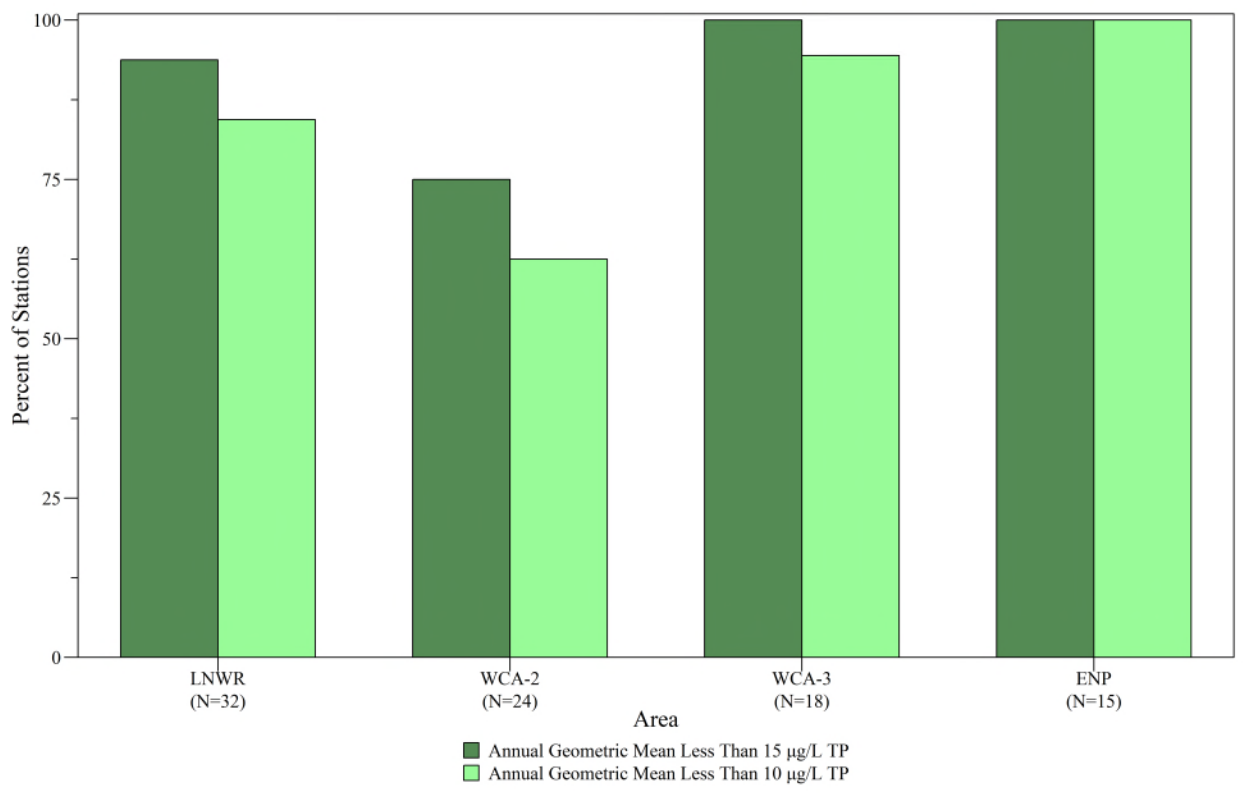
a. Expressed as  $\mu\text{g/L}$  per water year.

Interior marsh annual geometric mean TP concentrations observed across the EPA during WY2023 were lower relative to inflow structures (**Table 3-5** and **Figure 3-12**). During WY2023, annual geometric mean interior TP concentrations were 8.0  $\mu\text{g/L}$  in LNWR, 10.0  $\mu\text{g/L}$  in WCA-2, 6.1  $\mu\text{g/L}$  in WCA-3, and 4.6  $\mu\text{g/L}$  in ENP (**Table 3-5**). Marsh conditions are influenced by several factors including but not limited to marsh stage elevation; during extremely dry years, marsh concentrations become significantly elevated, as evidenced by high TP concentrations within LNWR during the late 1980s (**Figure 3-11**; WY1987: 85.4  $\mu\text{g/L}$  and WY1988: 27.5  $\mu\text{g/L}$ ).

Similar to the period average geometric mean concentrations trends observed at inflow regions, across the EPA, stepwise declines in average geometric mean concentrations declined across the Baseline, Phase I, and Phase II periods (**Table 3-5** and **Figure 3-11**). The most notable decline was observed between the Baseline and Phase I periods with continued success and declines apparent between the Phase I and Phase II periods. Across the entire POR, significantly declining trends in annual geometric mean concentrations were detected for all areas (**Table 3-6**). During the Phase II period, annual interior geometric mean TP concentration trends were significantly declining for all the regions within the EPA except WCA-2 (**Table 3-6**).

Annual geometric mean TP concentrations for individual interior marsh monitoring stations used in the assessment of the TP Rule (Rule 62-302.540 F.A.C., as detailed in Appendix 3-6 of this volume) and other ambient interior marsh monitoring stations sampled six or more times during WY2023

ranged from 2.75  $\mu\text{g/L}$  (site NP201 in ENP) in some unimpacted portions of the marsh to 32.42  $\mu\text{g/L}$  (site WCA2F1 in WCA-2). Across the entire EPA (LNWR, WCA-2, WCA-3, and ENP), 76% of the interior marsh sites exhibited annual geometric mean TP concentrations of 10.0  $\mu\text{g/L}$  or less during WY2023. Additionally, 83% of the interior sites had annual geometric mean TP concentrations of 15.0  $\mu\text{g/L}$  or less during WY2023 (**Figure 3-13**).

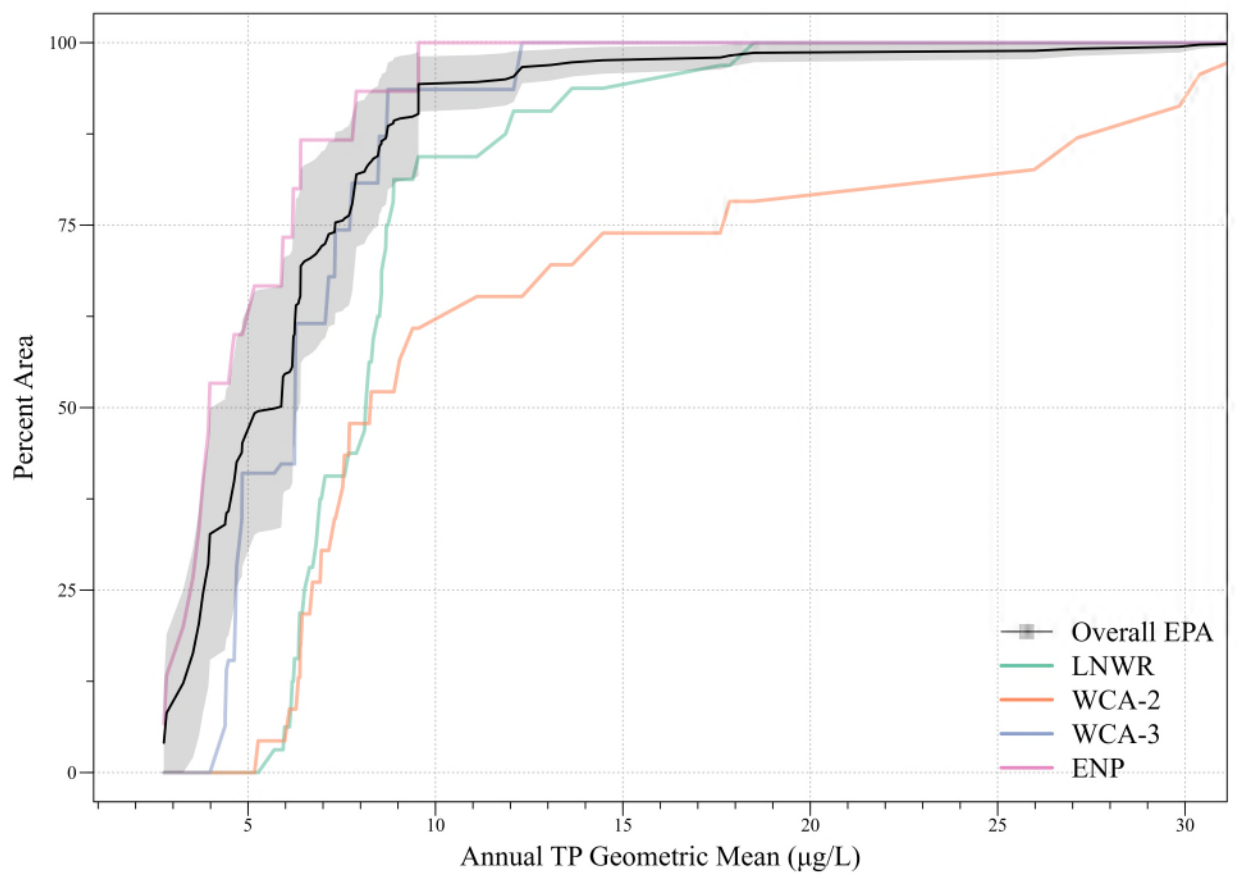


**Figure 3-13.** Percentage of stations within each region of the EPA with an annual geometric mean TP concentration less than 10 and 15  $\mu\text{g/L}$  during WY2023. The number in parenthesis indicates the number of sites used in assessment with greater than six samples (N) per year across the entire marsh monitoring network (TP Rule and ambient network).

The higher percent of interior monitoring stations meeting the 10 and 15  $\mu\text{g/L}$  limits observed for WY2023 reflects the continued recovery from recent climatic extremes, improved treatment of the inflows, and overall improvement in phosphorus conditions within the interior marsh due to ongoing restoration activities. Furthermore, this trend of improvement is apparent for stations used to assess the TP Rule. Given the relatively constant location of interior monitoring sites in recent years, temporal comparison of statistics from individual sites can be used to distinguish changes in measured concentrations.

Another way to view TP concentrations across the EPA is to look at the cumulative distribution of geometric mean across the ecosystem. The overall EPA cumulative distribution function (CDF) indicates that approximately 98% of the interior EPA is below 15  $\mu\text{g/L}$  and nearly 95% is below 10  $\mu\text{g/L}$  (**Figure 3-14**). Generally, the steepness and width of CDFs provide valuable insight into the distribution of data across a given area. As shown in **Figure 3-14**, ENP is skewed far to the left of the plot indicating that most of the area has very low annual geometric mean TP concentrations. Conversely, annual geometric mean concentrations are much more evenly distributed in WCA-2 given the shape of the CDF line (**Figure 3-14**). Amongst individual areas of the EPA, 100% of ENP and WCA-3, 97% of LNWR, and 74% of WCA-2

were below 15  $\mu\text{g/L}$  in WY2023, consistent with results of the percent stations presented above (Figure 3-13).



**Figure 3-14.** Cumulative distribution functions (CDFs) of annual geometric mean TP across the overall EPA and individual areas of the EPA including LNWR, WCA-2, WCA-3, and ENP in WY2023. Shaded region around the Overall EPA CDF represents the 95% confidence interval. Note, CDF estimated for ENP is based on four monitoring locations within Shark River Slough and may not be representative of all the freshwater portions of ENP.

### Total Phosphorus Criterion Achievement Assessment

The TP criterion specified for the EPA (62-302.540, F.A.C.) consists of a four-part test to be used to determine achievement of the criterion within LNWR, WCA-2, and WCA-3. As discussed below, the TP criterion for ENP is stipulated in the federal Settlement Agreement (Case Number 88-1886-CIV-MORENO). Each component must be achieved for a water body to be considered in compliance. Appendix 3-6 of this volume provides results of the preliminary evaluation to assess TP criterion achievement using available data for the most recent five-year period (WY2019–WY2023), impacted TP Rule station transition assessment, and TP Rule POR trend analysis. As described previously, the results of this assessment were affected by data limitations in many parts of the EPA during some years caused in part by the extremely dry conditions that have prevailed throughout the area. Additionally, monitoring at nine new sites (added to the existing sites to form the TP criterion monitoring network) was not initiated until January 2007. During WY2023, 53 of the 58 TP criterion monitoring network sites had sufficient data (i.e., six or more samples and samples in the wet and dry seasons specified by the screening protocol referenced by the TP Rule [Rule 62-302.540, F.A.C.]) to be included in the TP criterion assessment.

During WY2023, an assessment of impacted TP Rule stations was conducted based on guidance according to Subparagraph 62-302.540(4)(d)2, F.A.C. The assessment evaluated the stations to determine if they have achieved the long-term and annual limits. Individual stations in the impacted networks shall be deemed unimpacted for purposes of determining compliance with the TP Rule if the five-year geometric mean is less than or equal to 10 µg/L TP and the annual geometric mean is less than or equal to 15 µg/L TP. LOXA124 appears to have met the criteria to move from the impacted to unimpacted network. Additional trend analyses will be conducted to verify the change to unimpacted and will be included in future SFER reports. The detailed assessment can be found in Appendix 3-6.

The results of the WY2019–WY2023 TP criterion assessment indicate that, even with the data limitations, the unimpacted portions of each WCA passed all four parts of the compliance test (as expected) and are, therefore, in compliance with the 10 µg/L TP criterion. Occasionally, individual sites within the unimpacted portions of the WCAs exhibited an annual site geometric mean TP concentration above 10 µg/L, as expected, but in no case, did the values from any one unimpacted site result in an exceedance of the annual or long-term network limits. None of the annual geometric mean TP concentrations for the individual unimpacted sites during the WY2019–WY2023 period exceeded the 15 µg/L annual site limit.

In contrast, the impacted (i.e., phosphorus-enriched) portions of each water body failed one or more parts of the test and, therefore, exceeded the criteria. Impacted portions of the WCAs routinely exceeded the annual and five-year network TP concentration limits of 11 and 10 µg/L, respectively. However, some impacted stations routinely experience concentrations below the long-term and annual limits, resulting in transitioning of impacted stations to unimpacted. To date, seven impacted stations across the EPA have transitioned to unimpacted. Both in WY2022 and WY2023, WCA2F3 and LOXA124, respectively, appear to have met the criteria to move from the impacted to unimpacted network. Additional trend analyses will be conducted to verify the change to unimpacted and will be included in future SFER reports.

Within the phosphorus-impacted zone of the EPA, more specifically, the eutrophication gradient within WCA-2 (i.e., south of S-10 inflows; Juston et al. 2015), it was suggested that elevated water TP concentration could persist for many years in impacted portions of the marsh due to internal loading from phosphorus-enriched soils independent of continued reductions in inflow concentrations. This assessment was based on a combination of temporal soil and water column assessments along the highly studied eutrophication gradient within WCA-2 and provides a focused insight into ambient water column TP concentrations in the marsh, independent from inflow TP concentrations and flow events. The study also identified that there has been no significant further advancement of this enriched soil zone in WCA-2A 13 years after a ~75% reduction in inflow phosphorus load to the region (Juston et al. 2015). The suggestion of persistent elevated TP concentrations by Juston et al. (2015) is consistent with the individual station trend analysis presented in Appendix 3-4 and 3-6 of this chapter where very few significant trends are observed in impacted (or elevated TP) stations. However, some stations, such as X1 (LNWR) and WCA2F1 (WCA-2), experience significantly decreasing trends presumably due to their landscape position (edge of impacted zones) combined with ongoing restoration activities and improved boundary conditions. Those sites with significantly declining trends, assuming the magnitude of change and trend direction continues, could reach 10 µg/L in the next decade.

## Settlement Agreement

The TP Rule specifies that while the federal Settlement Agreement (Case Number 88-1886-CIV-MORENO) is in effect, compliance with the criterion in ENP will be assessed in accordance with the methodology specified in Appendix A of the Settlement Agreement using TP flow-weighted mean concentrations (FWMCs) at inflow sites instead of ambient marsh TP concentrations, as done in the other portions of the EPA. Compliance with the limits stipulated in the Settlement Agreement is expected to provide a long-term average inflow FWMC (i.e., target) of approximately 8.0 µg/L for Shark River Slough (SRS) and 6.0 µg/L for Taylor Slough and Coastal Basins. The Settlement Agreement tracking for ENP is

conducted by SFWMD and quarterly Settlement Agreement reports prepared by SFWMD are available online at <https://www.sfwmd.gov/our-work/toc>.

Assessment of TP concentrations into ENP for compliance with the Settlement Agreement is conducted on a federal water year basis (October 1–September 30). The long-term TP concentration limit for inflows to SRS is represented by concentrations delivered through the S-12A, S-12B, S-12C, and S-12D structures during the Outstanding Florida Waters baseline period of March 1, 1978, to March 1, 1979, and is adjusted for variations in flow. Inflow TP concentrations through the S-12A, S-12B, S-12C, S-12D, S-333, S-355A, and S-355B structures are compared to the long-term limits at the end of each federal water year. In August 2020, the Everglades Technical Oversight Committee reached consensus on an updated evaluation method (Method 1.5) that incorporates the S-356 seepage return pumps. In Method 1.5, the approximate volume from WCA-3 through S-356 is applied to the calculation of the long-term limit and the TP FWMC by including the lesser of the daily flow volumes from the S-356 and S-335 structures in the calculation. Beginning in federal WY2021, flow through the S-333N, which became operational in November 2020, was incorporated into the calculation of the long-term limit and TP FWMC. The addition of the S-333N to Method 1.5 calculation was approved by the Everglades Technical Oversight Committee on May 4, 2021. Compliance results for discharges entering SRS as reported for the federal WY2022 (October 1, 2021–September 30, 2022) by Mo et al. (2023) are as follows:

- A total of  $1,066 \times 10^3$  acre-feet (ac-ft) of WCA water was discharged to ENP via SRS.
- Annual TP FWMC entering SRS for federal WY2022 was 10.2  $\mu\text{g/L}$ .
- The annual long-term limit based on total flows into SRS for federal WY2022 was 7.6  $\mu\text{g/L}$ .

In addition to SRS, the Settlement Agreement also addresses discharges to Taylor Slough and Coastal Basins. Under the Settlement Agreement and associated Consent Decree, inflow TP FWMCs to ENP via Taylor Slough and Coastal Basins are compared to a fixed 11.0  $\mu\text{g/L}$  limit assessed during the federal water year. Since federal WY2007, structures S-332D and S-18C were used to assess compliance against the 11.0  $\mu\text{g/L}$  limit. During federal WY2017, two additional structures, S-328 and G-737, began operations in an effort to send more water to northeastern Florida Bay. In August 2020, the Everglades Technical Oversight Committee reached consensus on an updated evaluation method (Method 3) that incorporates flows and TP concentrations at S-328 and G-737. The method discounts S-332D flow through S-328 and the flow that enters through an adjacent detention area of the S332DX1. Compliance results for discharges entering Taylor Slough and Coastal Basins as reported for the federal WY2022 are as follows:

- Total flow entering Taylor Slough and Coastal Basins as estimated using Method 3 was  $323 \times 10^3$  ac-ft.
- TP FWMC entering Taylor Slough and Coastal Basins was 5.1  $\mu\text{g/L}$ .
- Annual reported TP FWMC for Taylor Slough and Coastal Basins for federal WY2021 was below the 11.0  $\mu\text{g/L}$  limit.

Over the past several federal water years, TP FWMCs for SRS have begun to approach the long-term target of 8  $\mu\text{g/L}$  with a long-term average (5-year) TP FWMC between 8.4 and 9.3  $\mu\text{g/L}$  (**Table 3-7**). Inflows to Taylor Slough and Coastal Basins are consistently below the limit and achieving the long-term target concentration of 6  $\mu\text{g/L}$  with long-term average FWMCs (5-year average) less than 6.0  $\mu\text{g/L}$ . To date, great progress has been made to improve water quality entering the EPA with the recognition that progress continues upstream to complete restoration projects associated with the *Restoration Strategies Regional Water Quality Plan* (SFWMD 2012; i.e., STA expansion, flow equalization basin [FEB] construction, infrastructure improvements, etc.) to fully comply with meeting phosphorus discharge limits to the EPA. The Everglades Technical Oversight Committee has recognized that localized phosphorus transport under certain hydrologic conditions can be a more significant driver of elevated phosphorus levels into ENP than upstream anthropogenic sources, which are being mitigated through BMP and STA source

control efforts. In October 2020, the Principals to the Consent Decree designated staff from each Everglades Technical Oversight Committee member agency to form a working group to research the underlying causes of this localized phenomenon and identifying adequate solutions to address it. The working group subsequently made several recommendations in February 2021, including a sediment study and a hydrodynamic study led by the United States Department of Interior and SFWMD, respectively. These studies will better inform engineering, operational, and maintenance solutions and the results were presented to the Principals in September 2023.

**Table 3-7.** Annual SRS and Taylor Slough and Coastal Basins TP FWMCs, annual limit TP, and 5-year TP FWMC for federal WY2007–WY2022.

Federal Water Year	Shark River Slough			Taylor Slough and Coastal Basins		
	Annual FWMC TP <sup>a</sup>	Annual TP Limit <sup>a</sup>	5-Year Average FWMC TP <sup>b</sup>	Annual FWMC TP <sup>a</sup>	Fixed TP Limit <sup>a</sup>	5-Year Average FWMC TP <sup>b</sup>
2007	9.8	11.8		4.8	11.0	
2008	10.6	10.2		5.6	11.0	
2009	8.2	8.2		6.2	11.0	
2010	8.9	8.9		5.4	11.0	
2011	9.2	12.0	9.3	5.6	11.0	5.5
2012	8.9	8.8	9.2	5.7	11.0	5.7
2013	7.2	7.6	8.5	4.8	11.0	5.5
2014	10.8	9.7	9.0	4.3	11.0	5.2
2015	7.7	11.9	8.8	4.5	11.0	5.0
2016	7.2	7.6	8.4	5.3	11.0	4.9
2017	9.7	7.9	8.5	5.9	11.0	5.0
2018	7.3	7.6	8.5	6.2	11.0	5.2
2019	10.0	9.7	8.4	5.3	11.0	5.4
2020	9.3	9.5	8.7	5.3	11.0	5.6
2021	8.2	7.6	8.9	5.6	11.0	5.7
2022	10.2	7.6	9.0	5.1	11.0	5.4

a. For federal WY2007 – WY2019, both SRS and Taylor Slough and Coastal Basins results represent Method 1 calculations, where applicable. For federal WY2020, WY2021 and WY2022, the results represent Method 1.5 calculations for SRS, and Method 3 calculations for Taylor Slough and Coastal Basins.

b. 5-year TP FWMC values are computed as arithmetic average of the annual results.

## Total Phosphorus Loads

Each year, the EPA receives variable amounts of surface water inflows based on the hydrologic variability within the upstream basins. These regulated inflows contribute to the TP loading to the EPA system. **Table 3-8** provides estimates of the inflow and TP load to each portion of the EPA for WY2023. Flows and TP loads are also provided for the Baseline (WY1979–WY1993), Phase I (WY1994–WY2004), Phase II (WY2005–WY2022), and WY2023 periods for comparison.

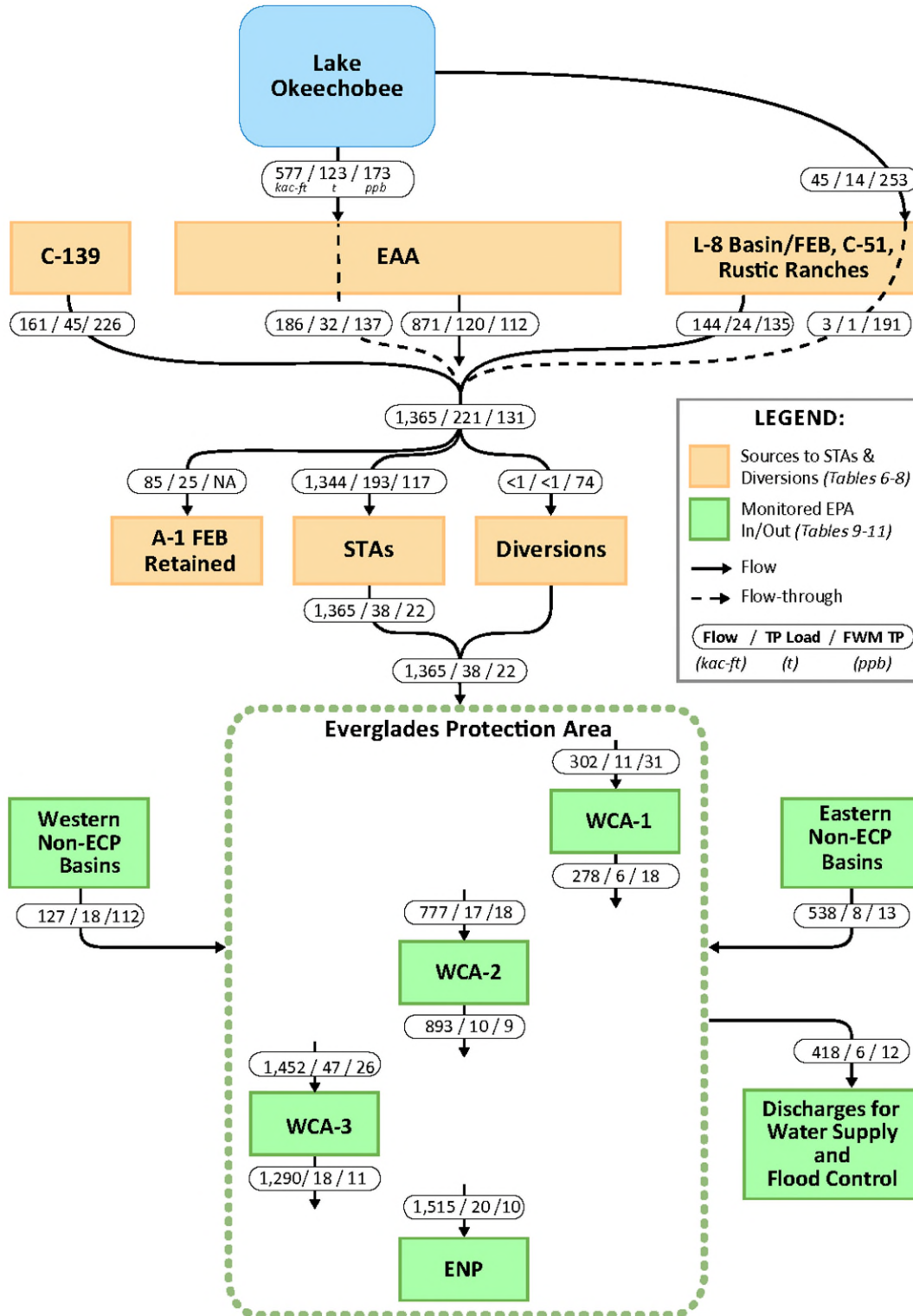
**Table 3-8.** Annual average flow, TP loads, and TP FWMCs into the EPA for Baseline (WY1979–WY1993), Phase I (WY1994–WY2004), Phase II (WY2005–WY2022), WY2023, and last five-water year period (WY2018–WY2023).

Parameter	Area	Period				
		Baseline WY1979– WY1993	Phase I WY1994– WY2004	Phase II WY2005–WY2022	Current WY2023	Last Five Water Years WY2019– WY2023
Mean Annual Flow (x 10 <sup>3</sup> ac-ft)	LNWR	506	647	302	259	302
	WCA-2	581	704	822	673	777
	WCA-3	1,181	1,396	1,373	1,442	1,452
	ENP	815	1,477	1,106	1,735	1,515
Mean Annual TP Load (t)	LNWR	111	84	20	7	11
	WCA-2	79	57	23	19	17
	WCA-3	108	84	46	48	47
	ENP	11	16	13	25	20
Mean Annual FWM TP (µg/L)	LNWR	186	100	53	22	31
	WCA-2	119	65	23	22	18
	WCA-3	72	49	27	27	26
	ENP	12	9	9	12	10

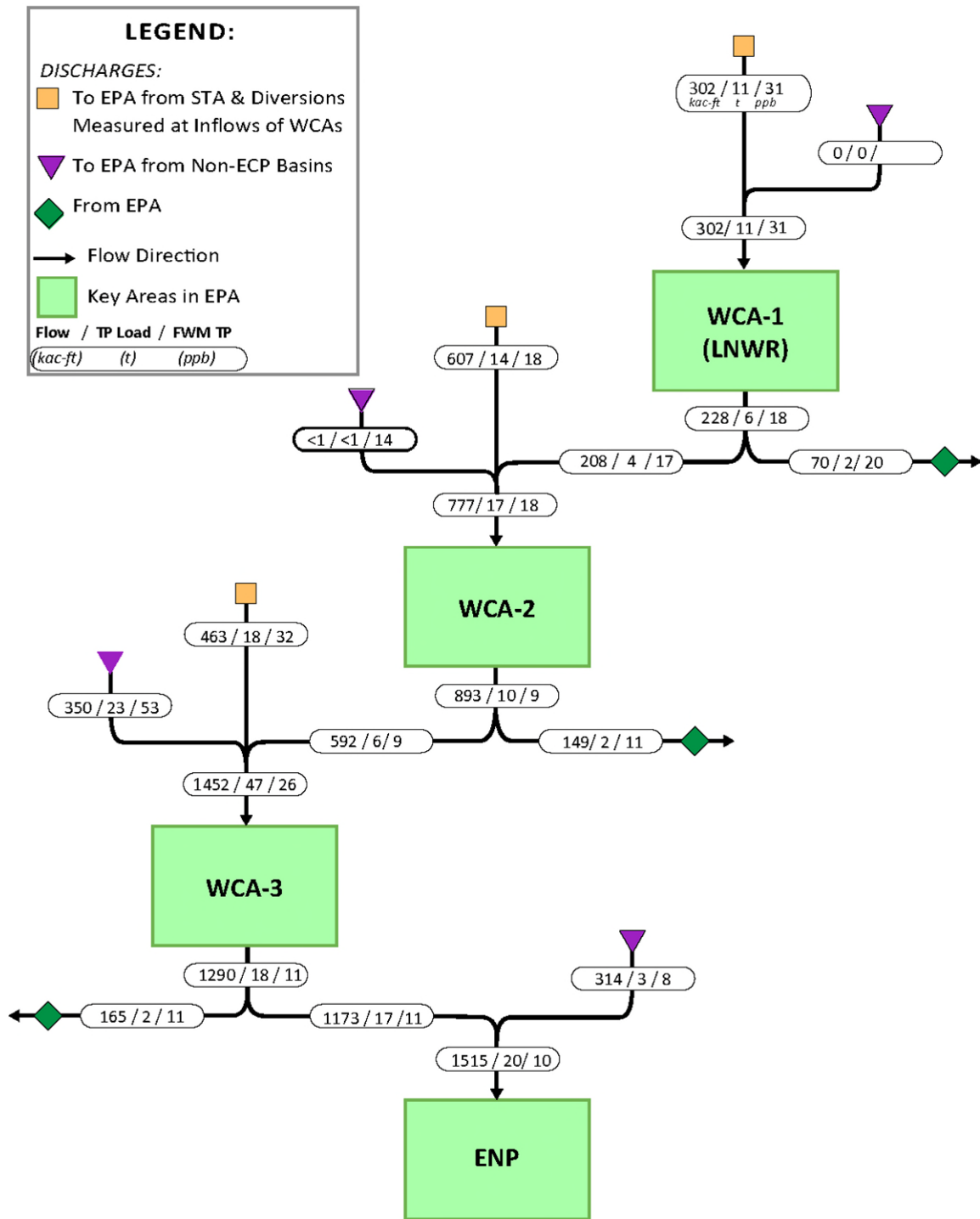
**Figure 3-15** shows five-year (WY2019–WY2023) average annual flows, TP loads, and TP FWMCs to Everglades STAs and diversions from inflow tributaries and across the EPA. Approximately 221 metric tons per year (t/yr) of TP was delivered from upstream sources (Lake Okeechobee, EAA Basin, C-139 Basin, L-8 Basin, and C-51W Basin) to the STAs, A-1 FEB, and diversions over the last five years. About 38 t/yr of TP was delivered to the EPA after treatment by the Everglades STAs and less than 1 t/yr of TP was delivered to the EPA by diversion. The A-1 FEB retained an average of 25 t/yr in WY2019–WY2023. Another 8 and 18 t/yr of TP were delivered to the EPA from the eastern and western Non-ECP basins, respectively. Data for **Figures 3-15** and **3-16** are presented in Appendix 3-5, Tables 6 through 11.

In addition to surface water inflow, atmospheric deposition contributes to the TP loading into the EPA. The long-term average range of TP atmospheric deposition to the WCAs is estimated between 82 and 146 t/yr. Atmospheric TP deposition rates are highly variable but not routinely monitored due to the high expense of such monitoring. The range (expressed spatially as 24 to 42 milligrams per square meter per year) is based on data obtained from long-term monitoring evaluated by SFWMD (Redfield 2002).

Detailed estimates of TP loads by structure for WY2023 are presented in Appendix 3-5 of this volume. This appendix summarizes contributions from all tributaries connecting to the EPA: Lake Okeechobee, EAA, C-139 Basin, other agricultural and urbanized areas, and the Everglades STAs. In some cases, surface water inflows represent a mixture of water from several sources as it passes from one area to another before arriving in the EPA. For example, water discharged from Lake Okeechobee can pass through the EAA and then through an STA before arriving in the EPA. Similarly, runoff from the C-139 Basin can pass through STA-5/6 and STA-3/4 and then into the EAA before reaching the EPA.



**Figure 3-15.** Five-year (WY2019–WY2023) average annual flows (1,000 acre-feet [kac-ft]), TP loads (metric tons [t]), and TP FWMC ( $\mu\text{g/L}$  or parts per billion [ppb]) to the Everglades STAs and diversions from inflow tributaries and across the EPA. (Note: WCD = water control district. Tables referred to in the legend are in Appendix 3-5.)

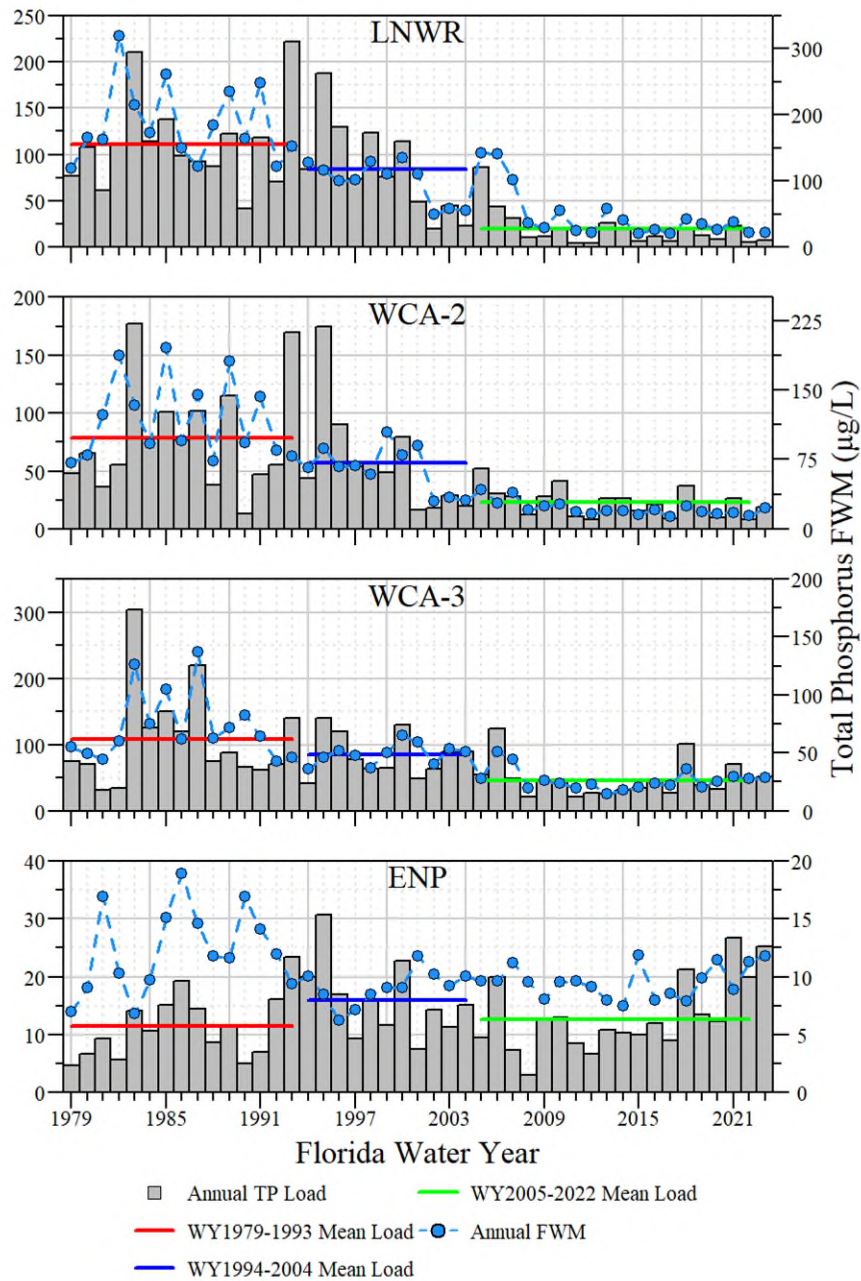


**Figure 3-16.** Five-year (WY2019–WY2023) average annual flows (kac-ft), TP loads (t), and TP FWMCs (µg/L or ppb) across the EPA. Values for each year are presented in Appendix 3-5 in the 2019 through 2023 SFRs – Volume I. The “from EPA” discharges, indicated by the green diamond, are for water supply and flood control.

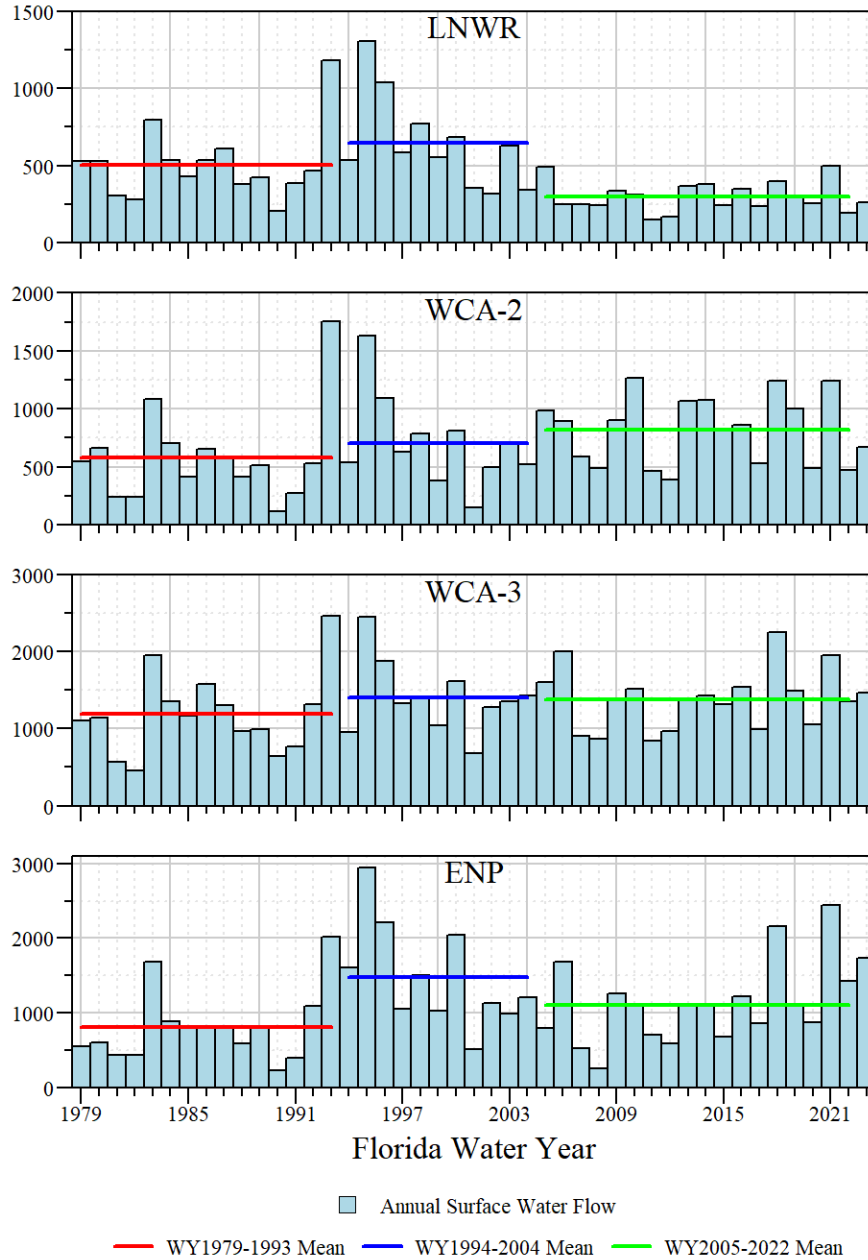
As detailed in Appendix 3-5, WY2023 annual TP loads from external surface sources (structures of the ECP and Non-ECP) to LNWR, WCA-2, WCA-3, and ENP were 68.4 t, with a TP FWMC of 27 µg/L. Another 296 t of TP is estimated to have entered the EPA through atmospheric deposition (Redfield 2002). Discharges from the EPA account for 4.8 t of TP for water supply and flood control. The 68.4 t TP load in EPA surface inflows, not accounting for internal transfers, represents an increase of approximately 19% compared to WY2022 (57.3 t). The higher TP loads to the EPA during WY2023 resulted from the increase of external source TP loads with an increase in flow volumes and concentrations entering the EPA. The EPA received  $2,046 \times 10^3$  ac-ft of surface water flow, which is a 12% increase from WY2022 volumes ( $1,835 \times 10^3$  ac-ft; Sotolongo Lopez et al. 2023). Annual TP loads to ENP from surface water sources were 25.2 t with a TP FWMC of 12 µg/L. ENP inflow loads increased 27% from WY2022 (19.8 t), due to a significant increase (22%) in surface water flow delivered to ENP in WY2023 ( $1,735 \times 10^3$  ac-ft) compared to that of the previous water year ( $1,422 \times 10^3$  ac-ft).

A summary of the annual flows and TP loads to each portion of the EPA for WY1979–WY2023 along with the annual averages for the Baseline (WY1979–WY1993), Phase I (WY1994–WY2004), and Phase II (WY2005–WY2022) periods is presented in **Figure 3-17**. The effectiveness of the BMP and STA phosphorus removal efforts is demonstrated by decreased TP loading to WCA-2 and WCA-3 during the Phase I and Phase II periods compared to the Baseline period despite increased flows (**Figure 3-18**). The effects are less apparent in ENP, where inflow concentrations have remained near background levels and TP loading responds more directly to changes in flow and climatic conditions (**Figure 3-18**).

The mean flow and TP load to the EPA, especially to LNWR, during the Phase II have been highly influenced by climatic extremes, as previously discussed. The annual TP load from all sources to LNWR was approximately 7.1 t during WY2023, which represents a 31% increase from WY2022 (5.4 t). Surface water volume increased 32% in WY2023 ( $259 \times 10^3$  ac-ft) compared to WY2022 ( $196 \times 10^3$  ac-ft). The TP FWMC decreased from 23 µg/L in WY2022 to 22 µg/L in WY2023. Other areas of the EPA experienced similar changes in flow and TP load during WY2023. WCA2 had a higher TP inflow load during WY2023 (18.6 t) relative to WY2022 (8.5 t). WCA-3 showed an increase of 10% in TP inflow load during WY2023 (48.2 t) relative to WY2022 (46.6 t) due to a surface water inflow increase of 7% (WY2023:  $1,442 \times 10^3$  ac-ft and WY2022:  $1,343 \times 10^3$  acft). ENP experienced an increase of 27% in TP inflow load during WY2023 (25.2 t) relative to WY2022 (19.8 t) due to flow increase. Increased TP loads to LNWR in WY2023 (flow:  $259 \times 10^3$  ac-ft and TP load: 7.1 t) are primarily due to increased flows comparing with in WY2022 (flow:  $196 \times 10^3$  ac-ft and TP load: 5.4 t). Although TP loads and concentrations were reduced relative to the Baseline period, more monitoring is needed before the effects of Phase II BMP and STA optimization projects can be accurately assessed.



**Figure 3-17.** Annual inflow TP load in metric tons per year (t/yr) and TP FWM in µg/L for LNWR, WCA-2, WCA-3, and ENP from WY1979 through WY2023. The horizontal lines indicate the mean annual loads for the Baseline (WY1979–WY1993), Phase I (WY1994–WY2004), and Phase II (WY2005–WY2022) periods.



**Figure 3-18.** Annual inflow surface water flow for LNWR, WCA-2, WCA-3, and ENP from WY1979 to WY2023 in  $\times 10^3$  acre-feet per year (ac-ft/yr). Horizontal lines indicate the mean annual flows for the Baseline (WY1979–WY1993), Phase I (WY1994–WY2004), and Phase II (WY2005–WY2022) periods.

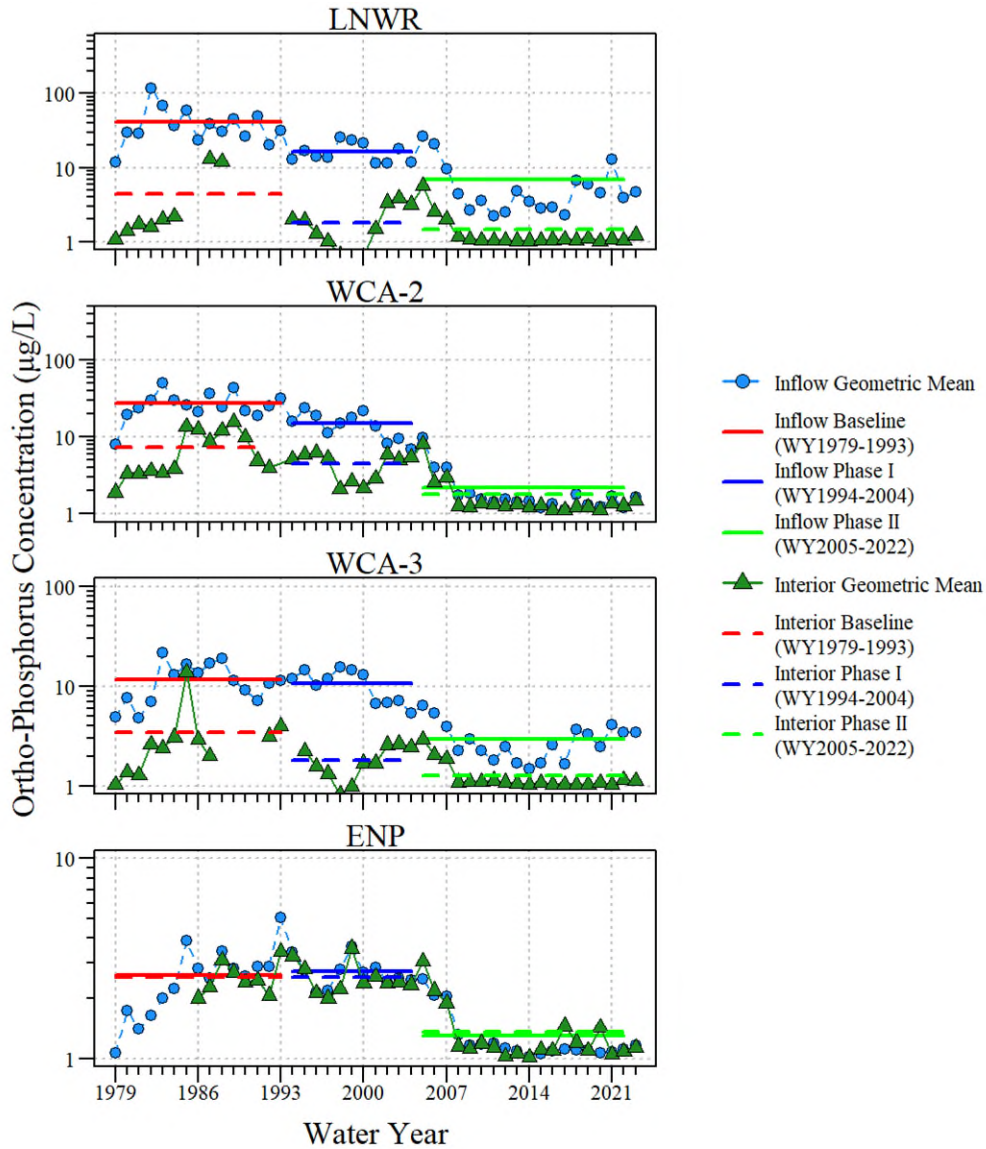
As shown in Appendix 3-5, Table 6, there was a decrease of flow from Lake Okeechobee to the Everglades STAs for WY2023 ( $43.3 \times 10^3$  ac-ft) compared with the five-year average (WY2019–WY2023;  $189.1 \times 10^3$  ac-ft). The TP loads from Lake Okeechobee decreased proportionally to 4.8 t in WY2023 compared to 32.2 t in WY2019–WY2023, as shown in Appendix 3-5, Table 7. The STAs received below average flow volumes from other upstream sources, including the EAA, C-139, L-8, and C-51W basins. Overall, the STAs received more than  $1,084 \times 10^3$  ac-ft of flow in WY2023, compared to the five-year average of approximately  $1,344 \times 10^3$  ac-ft. As a result, the STAs received below average TP loads from upstream sources and Lake Okeechobee in WY2023, as shown in Appendix 3-5, Table 7. The average periods of flows and TP loads to the Everglades STAs appear to do not explain overall outflow TP FWMCs of the STAs in WY2023 (25  $\mu\text{g/L}$ , as shown in Appendix 3-5, Table 8) compared with the outflow TP FWMC of 22  $\mu\text{g/L}$  for the WY2019–WY2023 period. Decreased TP FWMCs from STA-1E, STA-1W, and increase from STA-2 outflow partially explain the slightly decreased concentrations into LNWR (22  $\mu\text{g/L}$ ) and increase to WCA-2 (22  $\mu\text{g/L}$ ) in WY2023, compared with WY2022 (23  $\mu\text{g/L}$  into LNWR and 15  $\mu\text{g/L}$  into WCA-2; Appendix 3-5, Table 11).

### Orthophosphate Concentrations

Orthophosphate ( $\text{OPO}_4$ ) is an inorganic, soluble form of phosphorus readily utilized by biological organisms and, therefore, has the greatest and most rapid effect on the Everglades ecosystem. During WY2023, geometric mean  $\text{OPO}_4$  concentrations at inflow, interior, and outflow stations in all areas within the EPA were lower than concentrations observed during the Baseline and Phase I periods (**Figure 3-19** and **Table 3-9**).

Since WY1979,  $\text{OPO}_4$  concentrations have drastically declined for EPA inflows (**Figure 3-19**). During WY2023, geometric mean  $\text{OPO}_4$  concentrations observed at inflow regions were 4.7, 1.7, 3.4, and 1.2  $\mu\text{g/L}$  for LNWR, WCA-2, WCA-3, and ENP, respectively. Inflow geometric mean  $\text{OPO}_4$  concentrations have declined for all areas and regions throughout the POR, although inflow for WCA3 for WY2023 slightly increased when compared with Phase II WY2005-2022 (**Table 3-9**). Annual geometric mean concentrations for interior regions of the EPA have also steadily declined since the Baseline period to near background concentrations across the EPA (**Table 3-9** and **Figure 3-19**). During WY2023, geometric mean  $\text{OPO}_4$  concentrations observed at interior regions were 1.2, 1.5, 1.1, and 1.1  $\mu\text{g/L}$  for LNWR, WCA-2, WCA-3, and ENP, respectively (**Table 3-9** and **Figure 3-19**).

Since WY2008, annual geometric mean  $\text{OPO}_4$  concentrations for interior locations have been low, with regional annual geometric mean concentrations being less than 2.0  $\mu\text{g/L}$  (MDL) for all areas (**Figure 3-19**). Sustained reduction of  $\text{OPO}_4$  concentrations for both inflow and interior sites over the past several water years shows the continued recovery from past extreme climatic events, preferential removal of  $\text{OPO}_4$  by the Everglades STAs, and effects of restoration activities to improve the overall phosphorus conditions in the interior marsh areas.



**Figure 3-19.** Annual geometric mean OPO<sub>4</sub> concentrations for inflow and interior areas of LNWR, WCA-2, WCA-3, and ENP from WY1979 through WY2023. Horizontal lines indicate the mean annual geometric mean OPO<sub>4</sub> concentrations for the Baseline (WY1979–WY1993), Phase I (WY1994–WY2004), and Phase II (WY2005–WY2022) periods. (Note: The y-axis is on a log-scale).

**Table 3-9.** Summary statistics of OPO<sub>4</sub> concentrations in µg/L for the Baseline (WY1979–WY1993), Phase I (WY1994–WY2004), Phase II (WY2005–WY2022), and WY2023 periods.

Region	Class	Water Year Period	Sample Size	Geometric Mean	Geometric Standard Deviation	Median	Minimum	Maximum
LNWR	Inflow	1979–1993	1,620	30.2	4.5	42.0	<2	1,106
		1994–2004	1,242	15.8	2.9	14.0	<2	226
		2005–2022	2,876	4.9	5.3	2.0	<2	854
		2023	64	4.7	5.3	2.5	<2	208
	Interior	1979–1993	381	1.5	2.4	1.0	<2	278
		1994–2004	1,180	1.5	2.2	1.0	<2	10
		2005–2022	2,324	1.4	2.0	1.0	<2	506
		2023	122	1.2	1.4	1.0	<2	4
	Outflow	1979–1993	604	19.9	4.3	25.0	<2	1,274
		1994–2004	676	15.0	2.9	13.0	2	383
		2005–2022	1,082	1.9	2.9	1.0	<2	461
		2023	53	1.3	1.5	1.0	<2	4
	Rim	1979–1993	96	33.5	3.2	39.5	<2	408
		1994–2004	321	27.4	2.9	34.0	<2	190
		2005–2022	100	38.2	3.5	46.5	2	544
		2023	0	---	---	---	---	---
WCA-2	Inflow	1979–1993	805	23.9	4.0	31.0	<2	1,274
		1994–2004	740	12.7	2.9	10.0	2	352
		2005–2022	2,211	1.7	2.5	1.0	<2	190
		2023	122	1.7	2.1	1.0	<2	58
	Interior	1979–1993	1,504	4.6	5.1	2.0	<2	1,967
		1994–2004	1,210	4.3	4.6	4.0	<2	960
		2005–2022	1,344	1.5	2.1	1.0	<2	186
		2023	34	1.5	1.7	1.0	<2	10
	Outflow	1979–1993	1,360	3.9	3.5	2.0	<2	396
		1994–2004	977	5.2	2.4	5.0	2	156
		2005–2022	1,727	1.5	1.9	1.0	<2	153
		2023	79	1.2	1.3	1.0	<2	2
WCA-3	Inflow	1979–1993	2,157	10.7	4.5	11.0	<2	596
		1994–2004	2,143	9.5	3.6	7.0	2	431
		2005–2022	6,213	2.7	3.6	2.0	<2	510
		2023	312	3.4	4.0	2.0	<2	127
	Interior	1979–1993	581	1.8	2.6	1.0	<2	142
		1994–2004	1,679	1.7	2.3	2.0	<2	85
		2005–2022	1,672	1.3	1.6	1.0	<2	39
		2023	72	1.1	1.3	1.0	<2	3
	Outflow	1979–1993	1,684	2.7	2.3	2.0	<2	116
		1994–2004	1,713	3.0	1.7	2.0	2	97
		2005–2022	3,058	1.3	1.6	1.0	<2	180
		2023	237	1.2	1.5	1.0	<2	7
ENP	Inflow	1979–1993	2,023	2.6	2.1	2.0	<2	77
		1994–2004	1,949	2.7	1.6	2.0	2	97
		2005–2022	3,816	1.2	1.5	1.0	<2	43
		2023	295	1.2	1.4	1.0	<2	7
	Interior	1979–1993	484	2.5	1.7	2.0	2	63
		1994–2004	873	2.5	1.5	2.0	2	45
		2005–2022	1,017	1.3	1.6	1.0	<2	19
		2023	69	1.1	1.4	1.0	<2	4

## Total Nitrogen Concentrations

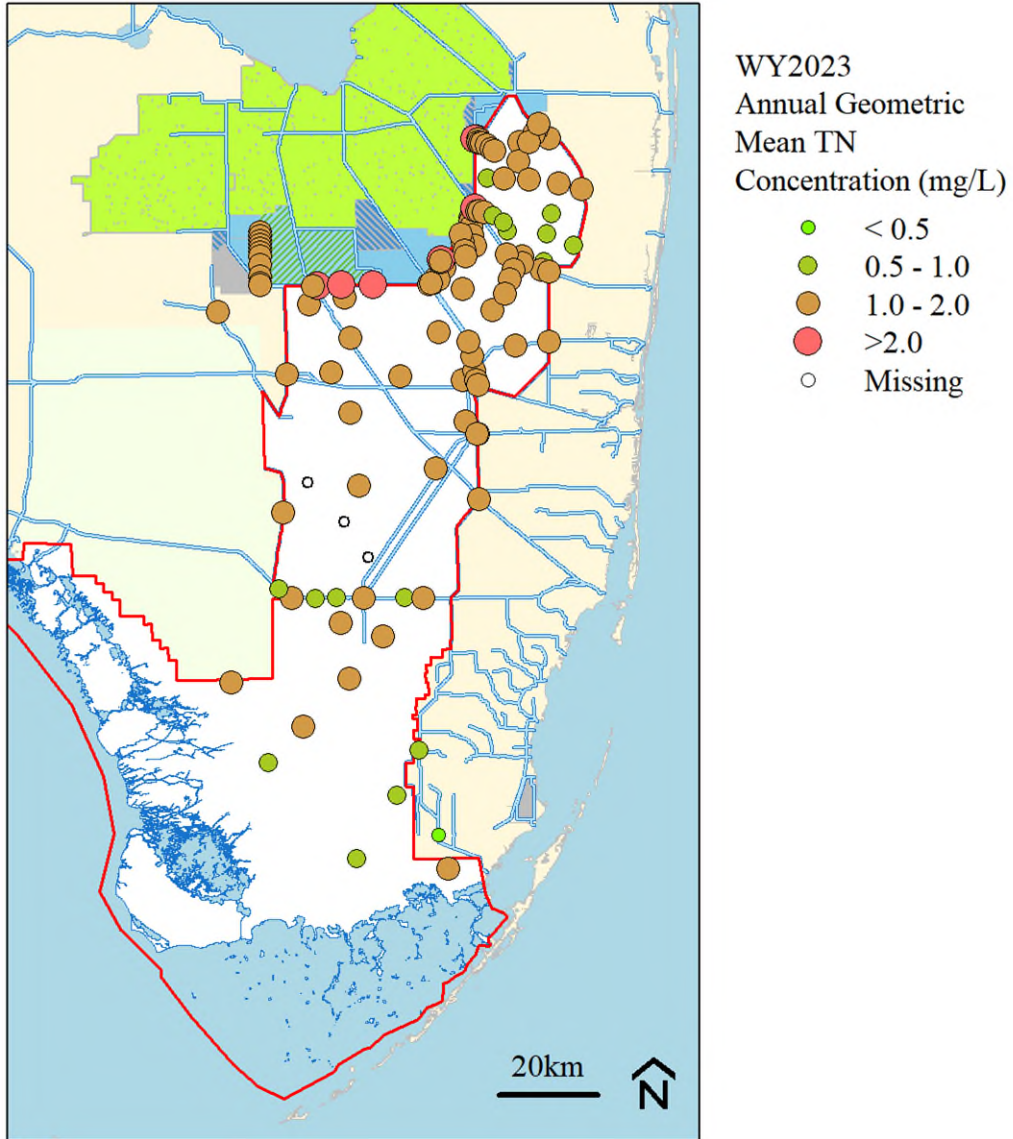
Elevated concentrations of nitrogen in freshwater ecosystems are of concern due to the role of nitrogen in eutrophication of freshwater systems, the effect on the oxygen content of receiving waters, and its potential toxicity to aquatic invertebrate and vertebrate species (Kadlec and Wallace 2009, Saunders and Kalff 2001). However, the EPA and the greater Everglades ecosystem in general are phosphorus-limited systems, which means the growth of algae and macrophytes are limited by the quantity of phosphorus input into the system. When nitrogen is limited, biota can offset this nitrogen limitation through fixation of atmospheric nitrogen gas ( $N_2$ ; Noe et al. 2001).

One of the primary objectives of this chapter is to document temporal changes in TN concentrations across the EPA using long-term geometric mean concentrations. The TN values for this chapter were calculated only for those samples for which both total Kjeldahl nitrogen and nitrate + nitrite ( $NO_x$ ) results were available, or TN was estimated from direct measure TN methods. A summary of area and class TN concentrations is presented in **Table 3-10** for the Baseline (WY1979–WY1993), Phase I (WY1994–WY2004), Phase II (WY2005–WY2022), and WY2023 periods.

As in previous years, TN concentrations during WY2023 exhibited a general north-to-south spatial gradient across the EPA (**Figure 3-20**). This gradient likely reflects the higher concentrations associated with discharges to the northern portions of the system from agricultural areas and Lake Okeechobee. A gradual reduction in TN concentrations results from the assimilative processes in the marsh as water flows southward. During WY2023, the north-to-south gradient is apparent for inflow regions within the EPA with the highest geometric mean TN concentrations being observed in LNWR inflows (2.0 mg/L), followed by inflows to WCA-2 (1.7 mg/L), WCA-3 (1.4 mg/L), and ENP (1.0 mg/L). Interior geometric mean TN concentrations were equal to or lower than inflow concentration within each region of the EPA, most likely due to marsh assimilation. During WY2023, TN concentrations in interior regions ranged from WCA-2 (1.7 mg/L), to ENP (1.0 mg/L). In interior portions of the EPA, biota (i.e., bacteria, algae, and macrophytes) are generally highly limited by phosphorus but may become nitrogen-limited in areas enriched with phosphorus, such as areas in close proximity to canals and impacted areas (Noe et al. 2001). While not specifically designed for TN reductions, BMPs implemented in the EAA have achieved approximately 50% reduction on TN loads from the Baseline period specific to BMP evaluation (WY1980–WY1988; Frydenborg and Frydenborg 2015). Since the implementation and enforcement of BMPs, changes in water management, and optimization of the Everglades STAs, the marsh condition within WCA-2 has improved based on general water quality improvements across the POR.

**Table 3-10.** Summary statistics of TN concentrations in mg/L for the Baseline (WY1979–WY1993), Phase I (WY1994–WY2004), Phase II (WY2005–WY2022), and WY2023 periods.

Region	Class	Water Year Period	Sample Size	Geometric Mean	Geometric Standard Deviation	Median	Minimum	Maximum
LNWR	Inflow	1979-1993	1,686	3.8	1.7	3.9	0.3	18.7
		1994-2004	2,296	2.6	1.6	2.5	0.3	54.8
		2005-2022	2,493	2.2	1.4	2.2	0.6	12.0
		2023	117	2.0	1.3	2.1	1	5.2
	Interior	1979-1993	378	2.4	1.6	2.3	0.7	36.7
		1994-2004	1,102	1.1	1.4	1.1	0.5	9.5
		2005-2022	3,427	1.1	1.4	1.1	0.5	8.7
		2023	232	1.0	1.3	1.0	0.5	2.4
	Outflow	1979-1993	603	2.6	1.7	2.6	0.3	22.8
		1994-2004	698	2.0	1.5	1.9	0.3	7.9
		2005-2022	1,164	1.4	1.3	1.4	0.7	6.3
		2023	55	1.2	1.3	1.2	0.9	2.1
	Rim	1979-1993	96	2.9	1.7	2.8	0.8	10.9
		1994-2004	449	2.4	1.5	2.3	0.3	9.7
		2005-2022	548	1.7	1.4	1.6	0.9	8.2
		2023	44	1.6	1.3	1.5	0.9	2.5
WCA-2	Inflow	1979-1993	848	3.0	1.6	3.0	0.5	22.8
		1994-2004	1,020	2.2	1.5	2.3	0.5	7.9
		2005-2022	2,113	1.7	1.3	1.7	0.4	6.3
		2023	168	1.7	1.2	1.7	0.9	2.9
	Interior	1979-1993	1,494	2.7	1.7	2.6	0.1	104.1
		1994-2004	1,183	2.0	1.5	2.1	0.1	16.7
		2005-2022	2,297	1.8	1.3	1.8	0.5	4.8
		2023	111	1.7	1.2	1.8	1	3.7
	Outflow	1979-1993	1,394	2.2	1.4	2.2	0.4	9.6
		1994-2004	1,019	1.6	1.4	1.7	0.3	4.4
		2005-2022	1,745	1.6	1.3	1.6	0.7	3.9
		2023	84	1.5	1.2	1.5	1	1.9
WCA-3	Inflow	1979-1993	2,222	2.0	1.6	2.0	0.3	10.8
		1994-2004	2,634	1.7	1.4	1.6	0.4	7.8
		2005-2022	6,327	1.5	1.3	1.5	0.6	12.2
		2023	399	1.4	1.2	1.3	0.8	4.5
	Interior	1979-1993	575	1.9	1.6	1.9	0.4	10.0
		1994-2004	1,433	1.2	1.4	1.2	0.1	9.0
		2005-2022	1,673	1.3	1.3	1.3	0.6	4.0
		2023	73	1.3	1.2	1.3	0.9	2.0
	Outflow	1979-1993	1,740	1.5	1.5	1.5	0.2	14.9
		1994-2004	1,699	1.1	1.4	1.1	0.3	4.1
		2005-2022	11,394	1.2	1.3	1.2	<0.02	11.1
		2023	1,077	1.2	1.3	1.3	0.5	2.1
ENP	Inflow	1979-1993	2,115	1.4	1.6	1.4	0.1	14.9
		1994-2004	1,931	0.9	1.5	0.9	0.3	3.6
		2005-2022	15,490	1.1	1.4	1.1	<0.02	11.1
		2023	1,395	1.0	1.5	1.2	0.4	2.1
	Interior	1979-1993	567	1.3	1.9	1.4	0.3	80.9
		1994-2004	940	1.1	1.6	1.1	0.3	5.7
		2005-2022	1,214	1.0	1.5	1.0	<0.02	53.0
		2023	94	1.0	1.4	1.0	0.6	4.5



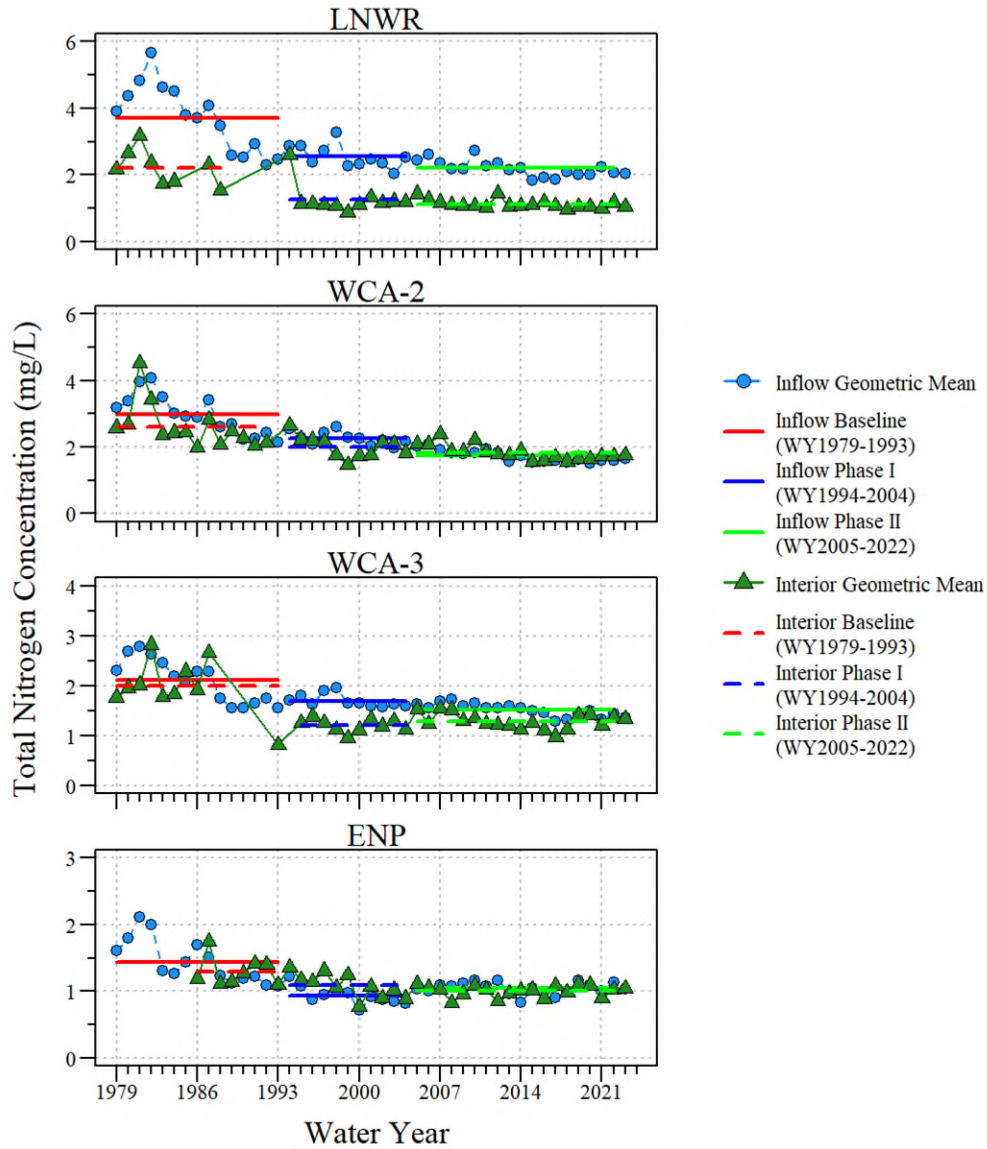
**Figure 3-20.** Annual geometric mean TN concentrations for all classifications for WY2023 at stations across the EPA.

Annual geometric mean TN concentrations have substantially declined since WY1979 for inflow and interior regions of the EPA as indicated by both **Table 3-11** and **Figure 3-21**. Annual inflow geometric mean concentrations across all areas show a significantly declining trend throughout the entire POR (**Table 3-11**). During the Phase II period, trends in annual inflow geometric mean concentrations were significantly declining for LNWR, WCA-2, and WCA-3. However, annual inflow geometric mean concentrations for ENP remained relatively level during this period (**Table 3-11**). Similar trends were observed for annual interior geometric mean concentrations for LNWR and WCA2 within the EPA with WCA-3 and ENP not having a significantly declining trend in the Phase II period (**Table 3-11**). The relatively low TN concentrations observed during WY2023 and the decreasing concentrations during the recent period (i.e., WY2005–present) may be the result of improved nutrient removal effectiveness of the Everglades STAs and effectiveness of EAA BMPs. As previously described (Payne et al. 2011, Julian et al. 2014b, 2015b), a strong relationship between interior station TN and total organic carbon within the EPA is present. This relationship indicates that the dominant source of the TN measured within the marsh is the organic material that naturally occurs in abundance in the wetland and enters the marsh from the oxidized sediments in the EPA. Additionally, relatively low observed NO<sub>x</sub> and dissolved inorganic nitrogen (un-ionized ammonia or NH<sub>3</sub>; see below) concentrations provide support to this conclusion, indicating that inorganic forms of nitrogen from anthropogenic sources to the EPA are relatively small and are not expected to pose a significant risk to the water quality and marsh condition within the EPA (**Figure 3-21**).

**Table 3-11.** Kendall's  $\tau$  annual geometric mean TN concentration trend analysis results for each region's inflow and interior classification within the EPA for the entire POR (WY1979–WY2023) and Phase II to current water year (WY2005–WY2023).

Class	Area	POR (WY1979–WY2023)			Phase II & Current Water Year (WY2005–WY2023)		
		Kendall's $\tau$	$p$ -value	Sen's Slope Estimate <sup>a</sup>	Kendall's $\tau$	$p$ -value	Sen's Slope Estimate <sup>a</sup>
Inflow	LNWR	-0.70	<0.01	-0.04	-0.46	<0.01	-0.02
	WCA-2	-0.80	<0.01	-0.04	-0.51	<0.01	-0.03
	WCA-3	-0.68	<0.01	-0.02	-0.68	<0.01	-0.02
	ENP	-0.40	<0.01	-0.01	-0.02	0.95	0.00
Interior	LNWR	-0.53	<0.01	-0.02	-0.42	<0.05	-0.01
	WCA-2	-0.58	<0.01	-0.02	-0.50	<0.01	-0.02
	WCA-3	-0.30	<0.01	-0.01	-0.25	0.14	-0.01
	ENP	-0.41	<0.01	-0.01	0.02	0.95	0.00

a. Expressed as mg/L per water year.



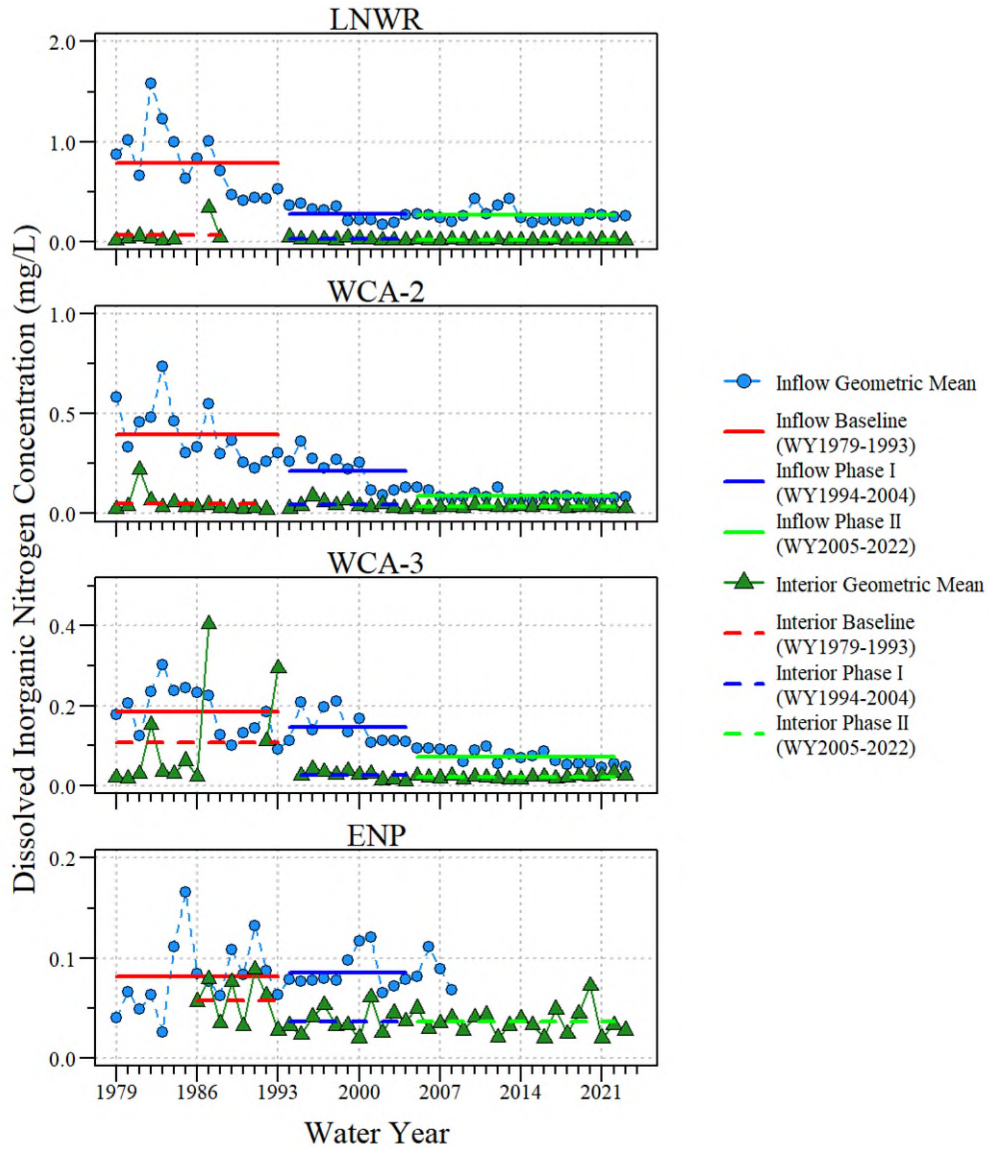
**Figure 3-21.** Annual geometric mean TN concentrations for inflow and interior areas of LNWR, WCA-2, WCA-3, and ENP from WY1979 through WY2023. Horizontal lines indicate the mean annual geometric mean TN concentrations for the Baseline (WY1979–WY1993), Phase I (WY1994–WY2004), and Phase II (WY2005–WY2022) periods.

## Dissolved Inorganic Nitrogen Concentrations

Nitrogen loading from canals is primarily in the form of dissolved organic nitrogen as discussed above and summarized by McCormick and O'Dell (1996). In areas of high phosphorus loading (see the *Total Phosphorus Loads* subsection), studies have observed a shift from phosphorus to nitrogen growth limitations of biota (Payne et al. 2000). Furthermore, following the addition of phosphorus, nitrogen availability is a significant regulator in vegetative dynamics including cattail expansion (McCormick and O'Dell 1996). In pristine areas of the Everglades, high rates of  $N_2$  fixation by cyanobacteria and soil microbes have been observed. Therefore, dominant sources of nitrogen to the Everglades ecosystem are namely from  $N_2$  fixation and surface water inputs of dominantly dissolved organic nitrogen. However, a fraction of dissolved inorganic nitrogen (DIN) enters the Everglades through stormwater run-off and nitrogen cycled by animals, especially fish and birds.

DIN, expressed as the sum of  $NO_x + NH_3$ , is highly bioavailable and typically rapidly consumed by biota. Inorganic nitrogen storage in wetlands is facilitated by the immobilization of  $NH_3$  in the biomass of plants and microbes or  $NO_x$  is enzymatically converted to ammonium ion ( $NH_4^+$ ) via nitrate ( $NO_3$ ) reduction from dissimilarity or assimilatory  $NO_3$  reduction to  $NH_4^+$  and, due to the pH conditions in the Everglades ecosystem,  $NH_4^+$  is rapidly converted to  $NH_3$ .

Generally, DIN concentrations observed in the EPA are very low and constitute a small fraction of the overall nitrogen in the system. Concurrent with periods of high TN, DIN concentrations were also elevated (**Figure 3-22**). Annual geometric mean inflow concentrations were highest in LNWR with an annual geometric mean concentration of 1.6 mg/L during WY1982 but have since declined to 0.26 mg/L for WY2023. This temporal decline in DIN concentration has occurred throughout the system with geometric mean concentrations ranging from 0.77 mg/L during the Baseline period to 0.26 mg/L in WY2023 for LNWR inflows, 0.39 to 0.08 mg/L for WCA-2 inflows, and 0.17 to 0.05 mg/L for WCA-3 inflows (**Table 3-12** and **Figure 3-22**). Since WY2009,  $NH_3$  data has not been collected for ENP inflows or WCA-3 outflow regions, however  $NO_x$  samples were collected concurrent with either total Kjeldahl nitrogen or TN. Similar trends are apparent in interior portions with concentrations being several orders of magnitude less than observed inflow concentrations.



**Figure 3-22.** Annual geometric mean DIN concentrations for inflow and interior areas of LNWR, WCA-2, WCA-3, and ENP for WY1979–WY2023. Horizontal lines indicate the mean annual geometric mean DIN concentrations for the Baseline (WY1979–WY1993), Phase I (WY1994–WY2004), and Phase II (WY2005–WY2022) periods.

**Table 3-12.** Summary statistics of DIN concentrations in mg/L for the Baseline (WY1979–WY1993), Phase I (WY1994–WY2004), Phase II (WY2005–WY2022), and WY2023 periods.

Region	Class	Water Year Period	Sample Size	Geometric Mean	Geometric Standard Deviation	Median	Minimum	Maximum
LNWR	Inflow	1979–1993	1,669	0.77	3.53	0.91	<0.02	12.56
		1994–2004	2,269	0.27	3.66	0.31	<0.02	57.20
		2005–2022	1,905	0.25	3.04	0.27	<0.02	8.88
		2023	66	0.26	3.46	0.23	<0.02	3.25
	Interior	1979–1993	385	0.03	3.94	0.02	<0.02	2.54
		1994–2004	1,067	0.02	2.27	0.02	<0.02	27.30
		2005–2022	2,103	0.02	1.99	0.02	<0.02	2.91
		2023	123	0.02	1.81	0.01	<0.02	0.21
	Outflow	1979–1993	601	0.15	5.75	0.15	<0.02	6.86
		1994–2004	695	0.09	4.67	0.08	<0.02	3.70
		2005–2022	749	0.04	2.59	0.03	<0.02	4.41
		2023	25	0.03	2.04	0.03	<0.02	0.27
	Rim	1979–1993	96	0.17	5.65	0.18	<0.02	4.84
		1994–2004	450	0.22	4.15	0.23	<0.02	6.30
		2005–2022	92	0.25	3.71	0.25	<0.02	4.82
		2023	0	--- <sup>a</sup>	---	---	---	---
WCA-2	Inflow	1979–1993	837	0.39	4.59	0.52	<0.02	6.97
		1994–2004	1,010	0.20	4.00	0.25	<0.02	3.70
		2005–2022	1,615	0.08	2.51	0.08	<0.02	4.41
		2023	104	0.08	2.33	0.08	<0.02	0.80
	Interior	1979–1993	1,515	0.05	4.75	0.03	<0.02	11.84
		1994–2004	1,166	0.04	2.65	0.04	<0.02	6.23
		2005–2022	1,312	0.03	2.17	0.03	<0.02	2.30
		2023	37	0.03	1.82	0.03	<0.02	0.20
	Outflow	1979–1993	1,382	0.07	4.27	0.06	<0.02	4.46
		1994–2004	1,023	0.05	3.38	0.04	<0.02	2.51
		2005–2022	1,331	0.05	2.20	0.04	<0.02	1.73
		2023	59	0.04	1.85	0.03	<0.02	0.24
WCA-3	Inflow	1979–1993	2,201	0.17	4.47	0.20	<0.02	5.69
		1994–2004	2,618	0.14	4.14	0.19	<0.02	4.82
		2005–2022	4,154	0.07	2.65	0.06	<0.02	3.27
		2023	275	0.05	2.79	0.04	<0.02	0.68
	Interior	1979–1993	605	0.03	3.68	0.02	<0.02	2.05
		1994–2004	1,459	0.02	2.40	0.02	<0.02	1.40
		2005–2022	1,668	0.02	2.04	0.02	<0.02	1.10
		2023	71	0.03	1.81	0.02	<0.02	0.13
	Outflow	1979–1993	1,734	0.06	2.96	0.06	<0.02	2.48
		1994–2004	1,684	0.06	2.57	0.06	<0.02	2.00
		2005–2022	522	0.07	2.50	0.07	<0.02	1.14
		2023	0	---	---	---	---	---
ENP	Inflow	1979–1993	2,107	0.08	3.23	0.07	<0.02	2.48
		1994–2004	1,913	0.08	2.71	0.09	<0.02	2.00
		2005–2022	621	0.09	2.62	0.10	<0.02	0.93
		2023	0	---	---	---	---	---
	Interior	1979–1993	567	0.05	4.37	0.04	<0.02	79.34
		1994–2004	938	0.03	3.00	0.03	<0.02	2.67
		2005–2022	1,007	0.03	2.53	0.03	<0.02	1.79
		2023	75	0.03	2.21	0.03	<0.02	0.21

a. No sample collected.

---

## LITERATURE CITED<sup>7</sup>

---

- Armstrong, D.A., D. Chippendale, A.W. Knight, and J.E. Colt. 1978. Interaction of ionized and un-ionized ammonia on short-term survival and growth of prawn larvae, *Macrobrachium rosenbergii*. *Biological Bulletin* 154:15-31.
- Axelrad, D.M., T. Lange, M. Gabriel, T.D. Atkeson, C.D. Pollman, W.H. Orem, D.J. Scheidt, P.I. Kalla, P.C. Federick, and C.C. Gilmour. 2008. Chapter 3B: Mercury and Sulfur Monitoring, Research and Environmental Assessment in South Florida. In: *2008 South Florida Environmental Report – Volume I*, South Florida Water Management District, West Palm Beach, FL.
- Axelrad, D.M., C.D. Pollman, B. Gu, and T. Lange. 2013. Chapter 3B: Mercury and Sulfur Environmental Assessment for the Everglades. In: *2013 South Florida Environmental Report – Volume I*, South Florida Water Management District, West Palm Beach, FL.
- Bancroft, G.T., W. Hoffman, R.J. Sawicki, and J.C. Ogden. 1992. The importance of the water conservation areas in the Everglades to the endangered wood stork (*Mycteria americana*). *Conservation Biology* 6(3):392-398.
- Bechtel, T., S. Hill, N. Iricanin, C. Mo, and S. Van Horn. 2000. Chapter 4: Status of Water Quality Criteria Compliance in the Everglades Protection Area and at Non-ECP Structures. In: *2000 Everglades Consolidated Report*, South Florida Water Management District, West Palm Beach, FL.
- Bellinger, B.J., and B.A.S. Van Mooy. 2012. Nonphosphorus lipids in periphyton reflect available nutrients in the Florida Everglades, USA. *Journal of Phycology* 48(2):303-311.
- Benoit, J.M., R.P. Mason, and C.C. Gilmour. 1999a. Estimation of mercury-sulfide speciation in sediment pore waters using octanol—water partitioning and implications for availability to methylating bacteria. *Environmental Toxicology and Chemistry* 18(10):2138-2141.
- Benoit, J.M., C.C. Gilmour, R.P. Mason, and A. Heyes. 1999b. Sulfide controls on mercury speciation and bioavailability to methylating bacteria in sediment pore waters. *Environmental Science & Technology* 33(6):951-957.
- Benoit, J.M., C.C. Gilmour, A. Heyes, R.P. Mason, and C.L. Miller. 2003. Geochemical and biological controls over methylmercury production and degradation in aquatic ecosystems. In: *ACS Symposium Series* 835:262-297.
- Bronaugh, D.B., and A. Werner. 2019. zyp: Zhang + Yue Pilon trends Package. R Package version 0.10 - 1.1. Available online at <https://CRAN.R-project.org/package=zyp>.
- Burns & McDonnell. 2003. *Everglades Protection Area Tributary Basins Long Term Plan for Achieving Water Quality Goals*. South Florida Water Management District, West Palm Beach, FL. October 27, 2003. Available online at [https://www.sfwmd.gov/sites/default/files/documents/waterquality\\_0.pdf](https://www.sfwmd.gov/sites/default/files/documents/waterquality_0.pdf).
- Castañeda-Moya, E., R.R. Twilley, V.H. Rivera-Monroy, K. Zhang, S.E. Davis, and M. Ross. 2010. Sediment and nutrient deposition associated with Hurricane Wilma in mangroves of the Florida coastal Everglades. *Estuaries and Coasts* 33:45-58.
- DeBusk, W.F., and K.R. Reddy. 2003. Nutrient and hydrology effects on soil respiration in a northern Everglades marsh. *Journal of Environmental Quality* 32:702-710.

---

<sup>7</sup> All *South Florida Environmental Reports* and *Everglades Consolidated Reports*, and the *Everglades Interim Report* are all available online at <https://www.sfwmd.gov/science-data/scientific-publications-sfer>.

- FDEP. 2004. *Data Quality Screening Protocol*. Florida Department of Environmental Protection, Tallahassee, FL. July 15, 2004. Available online at <https://floridadep.gov/sites/default/files/data-quality-screening-protocol.pdf>
- FDEP. 2013. *Technical Support Document: Derivation of Dissolved Oxygen Criteria to Protect Aquatic Life in Florida's Fresh and Marine Waters*. DEP-SAS-001/13, Florida Department of Environmental Protection, Tallahassee, FL. Available online at <https://floridadep.gov/sites/default/files/tsd-do-criteria-aquatic-life.pdf>.
- Frydenborg, R., and B. Frydenborg. 2015. *Effectiveness of Everglades Agricultural Area Best Management Practices for Total Nitrogen Reduction*. Florida Department of Agriculture and Consumer Services, Tallahassee, FL. Available online at [https://www.fdacs.gov/content/download/76316/file/NC2\\_FinalReport.pdf](https://www.fdacs.gov/content/download/76316/file/NC2_FinalReport.pdf).
- Gabriel, M., N. Howard, F. Matson, S. Atkins, and D. Rumbold. Appendix 5-7: Annual Permit Compliance Monitoring Report for Mercury in the STAs. In: *2008 South Florida Environmental Report – Volume I*, South Florida Water Management District, West Palm Beach, FL.
- Gilmour, C.C. 2011. *A Review of the Literature on the Impact of Sulfate on Methylmercury in Sediments and Soils*. Technical SP689, Florida Department of Environmental Protection, Tallahassee, FL. Available online at [https://hero.epa.gov/hero/index.cfm/reference/details/reference\\_id/4154251](https://hero.epa.gov/hero/index.cfm/reference/details/reference_id/4154251).
- Gilmour, C.C., E.A. Henry, and R. Mitchell. 1992. Sulfate stimulation of mercury methylation in freshwater sediments. *Environmental Science & Technology* 26(11):2281-2287.
- Hampson, B.L. 1977. Relationship between total ammonia and free ammonia in terrestrial and ocean waters. *International Council for the Exploration of the Sea Journal of Marine Science* 37(2):117-122.
- Julian II, P. 2015. Appendix 3A-6: Water Year 2010–2014 Annual Total Phosphorus Criteria Compliance Assessment. In: *2015 South Florida Environmental Report – Volume I*, South Florida Water Management District, West Palm Beach, FL
- Julian II, P., G. Payne, and S. Xue. 2013. Chapter 3A: Status of Water Quality in the Everglades Protection Area. In: *2013 South Florida Environmental Report – Volume I*, South Florida Water Management District, West Palm Beach, FL.
- Julian II, P., B. Gu, R. Frydenborg, T. Lange, A.L. Wright, and J.M. McCray. 2014a. Chapter 3B: Mercury and Sulfur Environmental Assessment for the Everglades. In: *2014 South Florida Environmental Report – Volume I*, South Florida Water Management District, West Palm Beach, FL.
- Julian II, P., G. Payne, and S. Xue. 2014b. Chapter 3A: Status of Water Quality in the Everglades Protection Area. In: *2014 South Florida Environmental Report – Volume I*, South Florida Water Management District, West Palm Beach, FL.
- Julian II, P., B. Gu, G. Redfield, K. Weaver, T. Lange, P. Frederick, J.M. McCray, A.L. Wright, F.E. Dierberg, T.A. DeBusk, M. Jerauld, W.F. DeBusk, H. Bae, and A. Ogram. 2015a. Chapter 3B: Mercury and Sulfur Environmental Assessment for the Everglades. In: *2015 South Florida Environmental Report - Volume I*, South Florida Water Management District, West Palm Beach, FL.
- Julian II, P., G. Payne, and S. Xue. 2015b. Chapter 3A: Status of Water Quality in the Everglades Protection Area. In: *2015 South Florida Environmental Report – Volume I*, South Florida Water Management District, West Palm Beach, FL.
- Julian II, P., G. Payne, and S. Xue. 2016. Chapter 3A: Status of Water Quality in the Everglades Protection Area. In: *2016 South Florida Environmental Report – Volume I*, South Florida Water Management District, West Palm Beach, FL.

- Julian II, P., G.G. Payne, and S.K. Xue. 2017. Chapter 3A: Water Quality in the Everglades Protection Area. In: *2017 South Florida Environmental Report – Volume I*, South Florida Water Management District, West Palm Beach, FL.
- Julian II, P., A.R. Freitag, G.G. Payne, S.K. Xue, and K. McClure. 2018. Chapter 3A: Water Quality in the Everglades Protection Area. In: *2018 South Florida Environmental Report – Volume I*, South Florida Water Management District, West Palm Beach, FL.
- Julian II, P., A.R. Freitag, G.G. Payne, and S.K. Xue. 2019. Chapter 3A: Water Quality in the Everglades Protection Area. In: *2019 South Florida Environmental Report – Volume I*, South Florida Water Management District, West Palm Beach, FL.
- Julian II, P., A.R. Gilhooly, G.G. Payne, and S.K. Xue. 2020. Chapter 3A: Water Quality in the Everglades Protection Area. In: *2020 South Florida Environmental Report – Volume I*, South Florida Water Management District, West Palm Beach, FL.
- Julian II, P., A.R. Gilhooly, G.G. Payne, and S.K. Xue. 2021. Chapter 3A: Water Quality in the Everglades Protection Area. In: *2021 South Florida Environmental Report – Volume I*, South Florida Water Management District, West Palm Beach, FL.
- Juston, J.M., R.H. Kadlec, W.F. DeBusk, M.J. Jerauld, and T.A. DeBusk. 2015. Persistence of legacy soil P and elevated background water P concentrations in Water Conservation Area 2A, a northern Everglades wetland: Legacy soil P in wetlands elevates water-P levels. *Water Resources Research* 51(12):9746-9762.
- Kadlec, R.H., and S.D. Wallace. 2009. *Treatment Wetlands, 2nd Edition*. CRC Press, Boca Raton, FL.
- Krest, J.M., and J.W. Harvey. 2003. Using natural distributions of short-lived radium isotopes to quantify groundwater discharge and recharge. *Limnology and Oceanography* 48(1):290-298.
- Kincaid, T.M., A.R. Olsen, and M.H. Weber. 2019. Spsurvey: Spatial Survey Design and Analysis. R package version 4.1.0. <https://CRAN.R-project.org/package=spsurvey>.
- Landing, W.M. 2014. Appendix 3B-2: Peer-Review Report on the Everglades Agricultural Area Regional Sulfur Mass Balance: Technical Webinar: November 20, 2013. In: *2015 South Florida Environmental Report – Volume I*, South Florida Water Management District, West Palm Beach, FL.
- McCormick, P.V., and M.B. O'Dell. 1996. Quantifying periphyton responses to phosphorus in the Florida Everglades: A synoptic-experimental approach. *Journal of the North American Benthological Society* 15:450-468.
- McCormick, P.V., and R.J. Stevenson. 1998. Periphyton as a tool for ecological assessment and management in the Florida Everglades. *Journal of Phycology* 34:726-733.
- McCormick, P., S. Newman, S. Miao, R. Reddy, D. Gawlik, C. Fitz, T. Fontaine, and D. Marley. 1999. Chapter 3: Ecological Needs of the Everglades. In: *Everglades Interim Report*, South Florida Water Management District, West Palm Beach, FL.
- McCormick, P.V., S. Newman, G. Payne, S.L. Miao, and T.D. Fontaine. 2000. Chapter 3: Ecological Effects of Phosphorus Enrichment in the Everglades. In: *2000 Everglades Consolidated Report*, South Florida Water Management District, West Palm Beach, FL.
- Miller, W.L. 1988. *Description and Evaluation of the Effects of Urban and Agricultural Development on the Surficial Aquifer System, Palm Beach County, Florida*. Report 88-4056, United States Geological Survey, Tallahassee, FL. Available online at <https://pubs.er.usgs.gov/publication/wri884056>.
- Mo, C., V. Ciuca, and J.P. Madden. 2021. *Settlement Agreement Report, Third Quarter, July – September 2020*. Prepared for the Everglades Technical Oversight Committee by South Florida Water Management District, West Palm Beach, FL. February 10, 2020; revised April 27, 2021. Available

online at

[https://www.sfwmd.gov/sites/default/files/documents/2020\\_3rd\\_quarter\\_rept\\_revision\\_w\\_SRS\\_result\\_final.pdf](https://www.sfwmd.gov/sites/default/files/documents/2020_3rd_quarter_rept_revision_w_SRS_result_final.pdf).

- Neil, L.L., R. Fotedar, and C.C. Shelly. 2005. Effects of acute and chronic toxicity of unionized ammonia on mud crab, *Scylla serrata* (Forsskål, 1755) larvae. *Aquaculture Research* 36:927-932.
- Noe, G.B., D.L. Childers, and R.D. Jones. 2001. Phosphorus biogeochemistry and the impact of phosphorus enrichment: Why is the Everglades so unique? *Ecosystems* 4:603-624.
- Orem, W., C. Gilmour, D. Axelrad, D. Krabbenhoft, D. Scheidt, P. Kalla, P. McCormick, M. Gabriel, and G. Aiken. 2011. Sulfur in the South Florida ecosystem: Distribution, sources, biogeochemistry, impacts, and management for restoration. *Critical Reviews in Environmental Science and Technology* 41(Supplement 1):249-288.
- Payne, G., and S. Xue. 2012. Chapter 3: Status of Water Quality in the Everglades Protection Area. In: *2011 South Florida Environmental Report – Volume I*, South Florida Water Management District, West Palm Beach, FL.
- Payne, G., K. Weaver, T. Bennett, and F. Nearhoof. 2000. *Everglades Phosphorus Criterion Technical Support Document, Part I: Water Conservation Area 2*. Florida Department of Environmental Protection, Tallahassee, FL.
- Payne, G., K. Weaver, and S. Xue. 2007. Chapter 3C: Status of Phosphorus and Nitrogen in the Everglades Protection Area. In: *2007 South Florida Environmental Report – Volume I*, South Florida Water Management District, West Palm Beach, FL.
- Payne, G., K. Weaver, and S. Xue. 2008. Chapter 3C: Status of Phosphorus and Nitrogen in the Everglades Protection Area. In: *2008 South Florida Environmental Report – Volume I*, South Florida Water Management District, West Palm Beach, FL.
- Payne, G., S. Xue, K. Hallas, and K. Weaver. 2010. Chapter 3: Status of Water Quality in the Everglades Protection Area. In: *2010 South Florida Environmental Report – Volume I*, South Florida Water Management District, West Palm Beach, FL.
- Payne, G., S. Xue, K. Hallas, and K. Weaver. 2011. Chapter 3: Status of Water Quality in the Everglades Protection Area. In: *2011 South Florida Environmental Report – Volume I*, South Florida Water Management District, West Palm Beach, FL.
- Pfeuffer, R.J. 1985. *Pesticide Residue Monitoring in Sediment and Surface Water within the South Florida Water Management District*. Technical Publication 85-2 and DRE 214, South Florida Water Management District, West Palm Beach, FL.
- Pfeuffer, R.J., and G.M. Rand. 2004. South Florida Ambient Pesticide Monitoring Program. *Ecotoxicology* 13:195-205.
- Pietro, K., R. Bearzotti, M. Chimney, G. Germain, N. Iricanin, T. Piccone, and K. Samfilippo. 2006. Chapter 4: STA Performance, Compliance and Optimization. In: *2006 South Florida Environmental Report – Volume I*, South Florida Water Management District, West Palm Beach, FL.
- Redfield, G. 2002. Atmospheric Deposition Phosphorus: Concepts, Constraints and Published Deposition Rates for Ecosystem Management. *The Scientific World Journal* 2:1843-1873.
- Reddy, K.R., S. Newman, S. Grunwald, R. Corstanje, and R. Rivero. 2005. *Everglades Soil Mapping Final Report*. South Florida Water Management District, West Palm Beach, FL.
- Saunders, D.L., and J. Kalff. 2001. Nitrogen retention in wetlands, lakes and rivers. *Hydrobiologia* 443:205-212.

- SFWMD. 2012. *Restoration Strategies Regional Water Quality Plan*. South Florida Water Management District, West Palm Beach, FL.
- Sotolongo Lopez, M., L. Hudson, and Shi Kui Xue. 2023. Chapter 3: Water Quality in the Everglades Protection Area. In: *2023 South Florida Environmental Report – Volume I*, South Florida Water Management District, West Palm Beach, FL.
- Thurston, R., and R.C. Russo. 1981. Ammonia toxicity to fishes. Effect of pH on the toxicity of the unionized ammonia species. *Environmental Science and Technology* 15(7):837-840.
- USACE and SFWMD. 1999. *Central and Southern Florida Project Comprehensive Review Study Final Integrated Feasibility Report and Programmatic Environmental Impact Statement*. United States Army Corps of Engineers, Jacksonville, FL; and South Florida Water Management District, West Palm Beach, FL.
- USEPA. 1997. *Guidelines for Preparation of the Comprehensive State Water Quality Assessments*. EPA-841-B-97-003A and 002B, United States Environmental Protection Agency, Washington, DC.
- USEPA. 2002. *Consolidated Assessment and Listing Methodology, First Edition*. United States Environmental Protection Agency, Washington, DC.
- USEPA. 2013. *Aquatic Life Ambient Water Quality Criteria for Ammonia – Freshwater*. EPA-822-R-13-001, United States Environmental Protection Agency, Washington, DC.
- Weaver, K. 2001. Appendix 4-4: Evaluation of Chronic Toxicity Based Guidelines for Pesticides and Priority Pollutants in the Florida Everglades. In: *2001 Everglades Consolidated Report*, South Florida Water Management District, West Palm Beach, FL.
- Weaver, K. 2004. *Everglades Marsh Dissolved Oxygen Site-Specific Alternative Criterion Technical Support Document*. Florida Department of Environmental Protection, Tallahassee, FL.
- Weaver, K., and G. Payne. 2004. Chapter 2A: Status of Water Quality in the Everglades Protection Area. In: *2004 South Florida Environmental Report – Volume I*, South Florida Water Management District, West Palm Beach, FL.
- Weaver, K., and G. Payne. 2005. Chapter 2A: Status of Water Quality in the Everglades Protection Area. In: *2005 South Florida Environmental Report – Volume I*, South Florida Water Management District, West Palm Beach, FL.
- Weaver, K., and G. Payne. 2006. Chapter 2A: Status of Water Quality in the Everglades Protection Area. In: *2006 South Florida Environmental Report – Volume I*, South Florida Water Management District, West Palm Beach, FL.
- Weaver, K., T. Bennett, G. Payne, G. Germain, S. Hill, and N. Iricanin. 2001. Chapter 4: Status of Water Quality Criteria Compliance in the Everglades Protection Area. In: *2001 Everglades Consolidated Report*, South Florida Water Management District, West Palm Beach, FL.
- Weaver, K., T. Bennett, G. Payne, T. Bechtel, S. Hill, and N. Iricanin. 2002. Chapter 2A: Status of Water Quality in the Everglades Protection Area. In: *2002 Everglades Consolidated Report*, South Florida Water Management District, West Palm Beach, FL.
- Weaver, K., G. Payne, and T. Bennett. 2003. Chapter 2A: Status of Water Quality in the Everglades Protection Area. In: *2003 Everglades Consolidated Report*, South Florida Water Management District, West Palm Beach, FL.
- Weaver, K., G. Payne, and S.K. Xue. 2007. Chapter 3: Status of Water Quality in the Everglades Protection Area. In: *2007 South Florida Environmental Report – Volume I*, South Florida Water Management District, West Palm Beach, FL.

- Weaver, K., G. Payne, and S.K. Xue. 2008. Chapter 3: Status of Water Quality in the Everglades Protection Area. In: *2008 South Florida Environmental Report – Volume I*, South Florida Water Management District, West Palm Beach, FL.
- Weaver, K., S.K. Xue, and G. Payne. 2009. Chapter 3: Status of Water Quality in the Everglades Protection Area. In: *2009 South Florida Environmental Report – Volume I*, South Florida Water Management District, West Palm Beach, FL.
- Wiener, J.G., D.P. Krabbenhoft, G.H. Heniz, and A.M. Scheuhammer. 2003. Ecotoxicology of Mercury. Pages 409-463 in: D.J. Hoffman, B. Rattner, G.A. Burton, and J. Cairns (eds.), *Handbook of Ecotoxicology*, CRC Press, Boca Raton, FL.