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Watershed restoration in the Florida Everglades: Agricultural water management and long-term trends in nutrient outcomes in the Everglades Agricultural Area

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ABSTRACT

Water quality degradation from agricultural runoff remains a pressing problem worldwide. A major challenge for restoring water quality is the need for long-term evaluation of governance and management interventions. In agricultural contexts, the primary interventions are best management practices designed to minimize nutrient losses by reducing fertilizer application, soil erosion, and drainage. Most studies are undertaken over short time scales or a few farms, which makes it difficult to connect management to water quality outcomes at larger scales. This paper addresses these gaps by examining 22 years of water quality trends at monthly time scales across the entirety of the 166 small, artificial drainage basins in the Everglades Agricultural Area, the sugarcane-growing region of Florida, USA. The Everglades Forever Act mandated the adoption of best management practices to reduce phosphorus loads but devolved implementation to farms collectively rather than requiring individual compliance. We examined the effect of biophysical and management drivers on long-term trends for two outcomes: a ratio of pumping-to-rainfall, which measures drainage decisions, and total phosphorus load per acre. We analyzed the magnitude and consistency of observed trends using Theil-Sen and Mann-Kendall analysis respectively across wet and dry seasons. Statistically significant downward trends were more common for decreases in magnitude than in consistency for both variables, indicating important management shifts have occurred but that some have not been continually improved over time. However, we also found statistically significant upward trends in a small number of basins for both variables. These results suggest that devolving management to farms has led to a widespread shift in management but that incentives for ongoing improvement would be valuable. Findings on biophysical and management drivers were limited, indicating that more fine-grain data may be needed to better detect their effects.

1. Introduction

Declines in water quality due to agricultural nonpoint source pollution (NPS) remain a pressing challenge worldwide, despite numerous efforts to mitigate the problem (Rissman and Carpenter, 2015; Patterson, 2017; Breitburg et al., 2018; Crawford et al., 2019). One of the major challenges facing managers, policymakers, and scientists is the need to evaluate the effectiveness of various governance interventions in improving water quality outcomes (Reimer et al., 2014; Boardman et al., 2017; Yoder et al., 2019). Many of the biogeochemical pathways of nutrient loss are now relatively well understood, and multiple interventions have been found to reduce losses of both nitrogen (N) and phosphorous (P) at the field scale and for small catchments (Royer et al., 2006; McDowell et al., 2009; Melland et al., 2016; Christianson et al., 2018). Yet, how to effectively promote the adoption of these measures, often called best management practices (BMPs), remains the key challenge to mitigating nutrient losses (Stuart et al., 2015).

Although a wide range of agri-environmental policies target agricultural NPS pollution worldwide, relatively few studies have systematically evaluated their effectiveness for improving water quality (Dowd et al., 2008). This gap in research is likely due to several factors. First, monitoring data is often lacking or available only at coarse spatial and temporal scales, making it difficult to assess whether BMPs have been effective (ibid.). Routine monitoring at farm scales can be prohibitively expensive (Horan and Shortle, 2001). Second, BMP adoption

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remains voluntary in most cases. This complicates determining which BMPs have been implemented, where, and to what degree, and in turn makes it difficult to assess the impacts of the BMPs at large spatio-temporal scales (e.g., Shortle and Abler, 2001; Ribaudo, 2015). Third, there is often political opposition to mandating BMP adoption, since it restricts farmer's management (Shortle et al., 2012; Emery and Franks, 2012; Barnes et al., 2013; Taylor and Van Grieken, 2015; de Loë et al., 2015; Yoder and Roy Chowdhury, 2018). However, even in instances where policies mandate BMPs, such as under the European Union's Common Agricultural Policy's nitrate vulnerability zones, farmers do not necessarily comply fully, and may instead search for loopholes or undertake only legally required minimums (Barnes et al., 2013). These challenges demonstrate the importance of examining the water quality outcomes connected to broader agri-environmental policies.

In this paper, we examine the outcomes of water quality regulations in the Florida Everglades through time and across its principal agricultural region. Water draining from farms in the Everglades Agricultural Area (EAA) into the Everglades ecosystem historically carried elevated levels of total phosphorus (TP), and began disrupting trophic dynamics and driving species invasions in the early 1980s (Sklar et al., 2005). In response, federal litigation led to a consent decree, codified in the 1994 Everglades Forever Act, which set a collective TP load limit (i.e., a nutrient cap) that EAA farms could generate annually (United States v. SFWMD, 1991; FAC 40-E63). The TP cap resulted from negotiations between federal, state, and agricultural representatives, which devolved some decision-making responsibilities to EAA farms instead of mandating individual compliance. This approach provided farmers with some management flexibility to select among a range of BMPs, in contrast to more traditional command-and-control regulations (e.g., Sabatier et al., 2005). Most importantly, the nutrient cap allowed EAA farmers collective involvement in the decision-making process towards meeting their legal obligation.

This collective approach both afforded (and introduced) the possibility of tradeoffs between high and low TP reductions by different farms for a net collective regional reduction (Yoder and Roy Chowdhury, 2018). These policies and state monitoring over two decades has demonstrated that EAA farms have reduced their annual TP loads by 55 % on average since 1994 (Taylor et al., 2019), and that a majority of farms contributed to these improvements overall (Yoder, 2019). However, these findings provide only a snapshot of these improvements. Longitudinal analysis is essential to understand how farms across the EAA have dealt with the management challenge while addressing the variable rainfall regimes characterizing south Florida's subtropical climate. Intense rainfall events create difficult tradeoffs between pumping out water to prevent crop damage and yield loss due to flooding, vs. floodwater retention for greater nutrient uptake by plants prior to releasing the excess water into drainage canals (Bottcher & Izuno 1994).

We expand on prior BMP research in the EAA that analyzed a representative sub-set of EAA farms over 10 years, focusing on indicator variables important at the farm scale (Izuno et al., 1999; Lang et al., 2010; Daroub et al., 2007, 2009, 2011). We do so by examining long-term (22 years of) water quality trends in drainage basins across the entire EAA, addressing both the magnitude and consistency of water quality improvements in wet and dry seasons. We refer to EAA farming operations making decisions as “basin managers” since this is the language used in the EAA regulatory context and that basins (as artificial irrigation and drainage areas) are the units at which our dependent variables are monitored by the state. We address two overarching research questions:

1 What are the temporal trends in EAA water quality outcomes?
2 How do underlying management and biophysical drivers affect those outcomes?

In addition to being one of the largest ecosystem restoration efforts in the world, the Everglades also represents a promising improvement in mitigating agricultural NPS pollution. The availability of monitoring data at daily time intervals at hundreds of monitoring sites in the EAA presents a critical contribution to understand what water management trends reveal about flexible implementation under a collective pollution cap. South Florida, in particular, is valuable to study because the strong seasonal differences in precipitation can provide insights into other areas, such as the U.S. Midwest, where climate change is intensifying rainfall events, which may make nutrient retention more challenging (Stuart et al., 2015; Bowling et al. 2018).

2. Governing and managing agricultural NPS pollution

2.1. Devolving implementation to farmers to facilitate BMP implementation

A major tension in governing agricultural NPS pollution is whether greater restrictions, especially through regulations, are necessary to improve water quality. The general trend in environmental governance, especially for complex problems that are difficult to monitor or enforce, such as NPS pollution, has been away from command-and-control towards more participatory or collaborative approaches (Sabatier et al., 2005). Environmental laws protecting water quality have generally been effective when dealing with point sources of pollutants for these reasons, since the source of pollution is evident (Houck, 2003). However, efforts to govern agricultural NPS pollution have relied primarily on voluntary measures, which have been insufficient to improve water quality (Shortle and Abler, 2001; Ribaudo, 2015). One challenge in regulating agricultural NPS pollution is the challenge of connecting farm management to downstream consequences, since there is limited farm-scale monitoring. This leads to ambiguity in determining whether farm management leads to declining water quality and becomes a serious impediment to legitimizing the need among farmers to change management practices (Duncan, 2016).

One alternative to command-and-control regulation has been to devolve responsibility to lower levels of government or to an intermediary organization (Marshall, 2008; Berkes, 2010). Devolution has helped to increase the credibility of information being shared on interventions and normalizes new practices by involving intermediaries that are trusted by farmers (Marshall, 2004; Dedeurwaerdere et al., 2015; Del Corso et al., 2017). Devolution also helps to address concerns with procedural fairness, often expressed as practical needs or cultural preferences for flexibility in making farm management decisions, both of which are crucial to effective implementation of nutrient retention measures (Emery and Franks, 2012; Burton and Paragahawewa, 2011). Involving farmers in the rulemaking process through devolution can help to reduce resistance by allowing farmers to help design more effective approaches, identify barriers or problematic assumptions made by regulators, potentially encourage greater compliance through peer pressure to conform to new practices (Taylor and Van Grieken, 2015; de Loë et al., 2015; Yoder and Roy Chowdhury, 2018).

2.2. Approaches and challenges in monitoring NPS pollution

Monitoring NPS pollution flows is extremely challenging and is often prohibitively costly at the farm scale (Horan and Shortle, 2001). Nutrients such as N or P can leach below the surface into groundwater, move through sub-surface tile drainage, dissolved in surface runoff from rainfall or irrigation, or be transported as particulate matter from soil erosion (Robertson and Vitousek, 2009; Royer et al., 2006; Daroub et al., 2011). Measuring sub-surface nutrient losses to groundwater is very difficult, while monitoring tile outflows is also costly (Horan and Shortle, 2001). In the absence of monitoring data that can clearly connect the source to the outcome across scales, it becomes increasingly difficult to assign responsibility without inviting farmers' skepticism (Macgregor and Warren, 2006; Duncan, 2016). It also means that a small sub-set of farmers may be largely responsible for nutrient losses, while the blame is shared widely (e.g., McGuire et al., 2013). Given the ambiguity that exists in NPS pollution, devolving administration to engage farmers in rule-making processes may be more successful in changing practices than simply mandating changes alone (Barnes et al., 2013).

Monitoring and evaluation of water quality over decadal time scales has demonstrated worsening water quality from increases in N and P (Dubrovsky and Hamilton, 2010; Crawford et al., 2019). Major challenges remain to better assess "critical source areas," which contribute disproportionately higher levels of nutrients (White et al., 2009), the time lags between BMP implementation and water quality responses (Meals et al., 2010), and differences in water quality outcomes at different spatial scales (Melland et al., 2018). Most research on BMPs has been conducted at the field or farm scales, and for short time periods (Sharpley et al., 2009). While the broad trends and pathways of how excessive nutrients enter waterways are understood (Carpenter et al., 1998), the mismatch between watershed-scale monitoring and farm-scale management has not reduced the ambiguities over which farms contribute the most or least to degraded water quality (Barnes et al., 2013; Ribaudo, 2015). Since there are few cases of regulating NPS pollution, we know relatively little about the efficacy of different policies or BMP implementation for water quality improvements (Dowd et al., 2008). Most studies rely on modeling of possible effects rather than empirical outcomes (ibid.). Monitoring is not done in all cases (Patterson, 2017) and may only be undertaken at coarse time scales, such as monthly (Dowd et al., 2008). In the few cases where both monitoring and policies are analyzed, linking water quality outcomes to farm management was critical to demonstrating the importance of widespread participation to achieve larger-scale improvements (Del Corso et al., 2017) and long-term trends (Patterson, 2017).

2.3. Effectiveness of BMPs and site-specific factors

In contrast to the limited monitoring and evaluation of NPS policies, there is an extensive literature examining the efficacy of BMPs on water quality at the field and farm scale over short time frames, typically 1–4 years (Liu et al., 2017). However, the extent to which BMPs can deliver improved water quality at larger scales and over longer time frames remains a critical gap in the literature (Liu et al., 2017; Hanrahan et al., 2018; Melland et al., 2018; Falcone et al., 2018). While there are a wide variety of BMPs based on good science, many fail to perform as anticipated (Easton et al., 2008). In part, this is due to a range of complicated, site-specific biophysical factors, including soils, water table, location in the watershed, and intensity of rainfall, which complicate predictions (Ribaudo, 2015; Sharpley et al., 2015). For example, research has shown that the BMP of maintaining edge-of-field vegetative buffers reduced surface drainage volumes by an average 45 %, but varied in effectiveness from 0 to 100%. The same BMP trapped 75 % of sediments on average, but ranged in effectiveness from 2 to 100% (Arora et al., 2010).

Management factors play a substantial role in the implementation and effectiveness of BMPs. The amount, timing, and type of application of P and N through fertilizers can shape large differences in nutrient loss at the farm scale based on the use-efficiency of different crops (Sharpley et al., 2015) and the types of tillage practices that farmers use (Hanrahan et al., 2018). Research indicates that only one-third of U.S. farmers use BMPs for fertilizer amounts, timing, and incorporating fertilizers into soils (Ribaudo et al., 2011). Liu et al. (2017) argue that studies often based models on an assumption that farmers effectively maintain BMPs, while in practice BMPs lose effectiveness over time. Even while studies generally show that BMPs are effective in reducing phosphorus, the legacy effects of nutrients mean it can take anywhere from 4 to 20 years to reliably detect improvements in water quality at meso-scale ($1-100\ km^2$) watersheds (Melland et al., 2018). Because of the variability at the farm scale and time lags, longitudinal research on trends at multiple scales is critical.

Prior research on BMPs in the EAA focused on both biophysical and managerial factors and identified several variables driving water quality outcomes at the farm scale (Izuno et al., 1999; Lang et al., 2010; Daroub et al., 2007, 2009, 2011). These include: TP loads per acre, referred to as unit area load (UAL); ratio of pumping-to-rainfall (P:R), and soil type, which is tightly correlated with soil depth in this region. Lang et al. (2010) examined 16 variables across crop choices, water management, and farm-specific variables for 10 EAA farms from 1992 – 2002. They found that P:R was the most statistically significant, direct driver of UAL. High intensity rainfall events, common during the rainy season in south Florida, require pumping to lower water tables and avoid flood damage to crops. Earlier work established that such events result in P concentration spikes in drainage water, especially if they occur following fertilizer applications (Izuno et al., 1999).

Daroub et al. (2009) conducted trend analysis on the same 10 EAA farms for the same time period (1992–2002) and found a decreasing monotonic trend of P:R for three farms, an increasing trend for one farm, and no significant trends for the remaining six farms. However, seven farms had statistically significant decreasing trends for UAL, indicating differences in outcomes between water management (e.g., reducing drainage volumes and velocities) and controlling P by reducing soil erosion, fertilizer application, or removing aquatic vegetation in canals. Crop choices and management were also linked to TP loads. Higher percentages of land planted in sugarcane or under flood-fallow were correlated with lower TP loads (Daroub et al., 2011). Non-sugarcane crops, particularly vegetables and sod, require more fertilizer and intensive water management due to greater flood sensitivity (Lang et al., 2010). Other research has shown that sugarcane can function as a net assimilator of phosphorus in the EAA (Izuno et al., 1999; Glaz, 1995).

Lastly, soil depth also affects TP loads through water retention and adsorption processes. Because of soil subsidence, higher water tables in state-managed canals compared to internal farm canals (called "canal head difference") is statistically correlated with higher per-acre TP loads (Izuno et al., 1999). Deeper soils can enhance the ability of farms to retain water, reducing the need for pumped drainage that transports both soluble and particulate P (Lang et al., 2010). However, shallower soils permit greater interaction of drainage water with limestone bedrock, where soluble P is adsorbed and becomes unavailable for plant growth (Daroub et al., 2011). Research examining basin managers' perspectives on implementing BMPs found that the flexibility to tailoring BMPs to the farm was important to generating participation (Yoder and Roy Chowdhury, 2018). In particular, basin managers felt that flexibility with rainfall-detention BMPs, which affect P:R by allowing basin managers to select different thresholds of 0.5, 1.0, or 1.5 in. of rainfall detention prior to pumping, was a valuable and effective tool in reducing UAL. Basin managers typically selected 0.5-inch BMPs if they cultivated vegetables, which need drier soil conditions than sugarcane, or if they had shallow soil depth and need faster drainage (ibid.).

3. Study area: EAA water management, regulations, and BMPs

The EAA encompasses 280,000 ha of farmland, where sugarcane is the dominant crop, making up 80 % of crops by acreage, while vegetables, sod, and rice comprise the remaining major crops (Izuno et al., 1999). Water management is driven by several interconnected dynamics: seasonal rainfall, elevation, and hydric soils. Agriculture depends on six large canals operated by a state agency, the South Florida Water Management District (SFWMD), to provide irrigation supply and flood control because the flat elevation limits the velocity of natural drainage. Farms irrigate and drain by raising and lowering the water table using large hydraulic pumps, which withdraw water from and discharge into state-owned canals. Internally, farms route water among one or multiple farms, depending on whether additional pumps are privately owned or owned jointly through a drainage district. Because of historical inundation, hydric soils are the predominant soil type (Ingebritsen et al., 1999). These have oxidized from exposure to oxygen, leading to upwards of seven feet of subsidence in some locations (Sklar et al., 2005). This means farms with less soil depth experience greater flood risks to their crops' root zones, since there is substantial sub-surface seepage between fields or farms, depending on differences in the water table (Yoder, 2019). Soils are generally deeper closer to the northern end of the EAA, which received seasonal inflows from Lake Okeechobee, and shallower farther south (Lang et al., 2010).

EAA drainage waters enriched with phosphorus (P) caused species invasions and disrupted the Everglades' nutrient-poor food webs beginning in the 1980s (Sklar et al., 2005). The declining water quality eventually led to a lawsuit, negotiated settlement, and state legislation regulating acceptable P loads from EAA water draining into the Everglades (United States v. SFWMD, 1991). A major component of the regulations was that EAA farms would implement a set of BMPs to help reduce P loads draining from their farm to achieve collective reductions set at 25 % below a pre-regulatory baseline P load. EAA farms have maintained their joint compliance every year, averaging yearly P loads 55 % lower than their baseline level (Taylor et al., 2019). Regulatory permits require BMPs to address fertilizer application, sediment controls, and water management (Adorisio et al., 2006). Examples of BMPs include more soil testing, banding instead of broadcasting fertilizer, vegetated buffers along ditches and canals, and reducing the volume and slowing the velocity of drainage through reduced pumping (Daroub et al., 2011).

4. Data and methods

We obtained GIS and water management data from the SFWMD (South Florida Water Management District (SFWMD, 2016a) for all of the EAA farms for May 1994 – April 2016. Basins are the unit of analysis for which water management data are available, reflecting the area's interconnected canal infrastructure. In the EAA, up to several dozen farms may share a hydraulic pump, which serves both as the inflow and outflow location. These pumps also serve as the monitoring station for collecting SFWMD water management data. Basins and pump locations are depicted in Fig. 1. Thus, all water quality monitoring and metrics are associated with basins. We used SFWMD Everglades Works of the District permits to identify the primary farm operator for each basin, based on greater than 50 % acreage holdings within each basin. GIS data included EAA basin polygons, canal networks, and the locations of water quality monitoring stations.

Basin-level water management data included daily phosphorus concentrations (mg/L), daily pumped water volume discharges (millions of gallons) from each basin's hydraulic pump(s), and daily rainfall (inches), which we converted to monthly values. Following prior research by Daroub et al. (2009) and Lang et al. (2010), we derived two dependent variables from these monthly data: unit area phosphorus load (UAL kg P ha^{-1}) and the unitless ratio of water volume discharges pumped from basins (m^3) to rainfall volumes (m^3) (P:R). UAL was calculated by dividing water volume discharges and phosphorus loads from basins by basin areas, respectively. Monthly rainfall volumes were calculated by multiplying basin area and monthly rainfall totals at the water quality stations associated with each basin. Data aggregations and transformations were conducted using MATLAB and Excel.

For rainfall, we added 17 weather station sites via the SFWMD's DBHYDRO dataset (South Florida Water Management District (SFWMD, 2016b) to supplement the data collected at each monitoring station. To better approximate actual rainfall amounts observed at each monitoring station, the rainfall data was interpolated via a simple kriging approach in ArcGIS Desktop 10.4. The Average Nearest Neighbor tool was used to calculate the mean distance between rainfall stations (in feet). The mean distance was then rounded up to the nearest 100 feet and this value was used as the lag distance in the Simple Kriging tool, to ensure that areas with sparsely located monitoring stations would be assigned estimated rainfall values. The monthly interpolated rainfall raster datasets were set to a spatial resolution of 500 feet (152.4 m). Estimated rainfall values were then extracted from the pixel co-located with each water quality monitoring station.

In addition, we derived two additional metrics for farm management and soil type. In Works of the District permits, each permittee selected different levels of water detention (0.5, 1.0, or 1.5 inch) during rainfall events to reduce the amount of discharge, and potential sediment and nutrient transport, mechanically drained by hydraulic pumps. This rainfall detention BMP was included as a key hypothesized management driver of UAL and P:R outcomes. Soils within basins were derived from NRCS SSURGO data (version November 2015) downloaded from the Florida Geographic Data Library, which indicated that a total of 52 unique soil types are located within the EAA. Two georeferenced soil type maps (Cox et al., 1988; South Florida Water Management District (SFWMD, 2006) and soil type descriptions for the EAA (Snyder, 2004) were used to condense the 52 soil types into 10 generalized categories. Soil depth ranges were included as an attribute in the SSURGO data. The percent area associated with each soil type in each EAA basin was calculated. For each basin, the soil with the largest relative (percent) share, i.e. dominant soil type, was identified.

4.1. Data analysis

A total of 166 out of the 187 basins present in the EAA were included in the analysis. We excluded 13 basins due to the absence of water quality data or the presence of non-agricultural land uses, while 8 adjacently located basins were merged because they shared the same downstream water quality monitoring station. The monthly water quality, discharge, and rainfall data were coded by wet season (June – October) and dry season (November – May) (Lang et al., 2010). We used the Theil-Sen (TS) median slope statistic to quantify the rate of change in annual median UAL and P:R values by wet and dry season. This procedure calculates the slope for each pairwise combination of samples in time. The median of these slopes is then used to characterize the trend. The median slope was chosen since it is a robust statistic that is resistant to the impacts of outliers and thus effective in characterizing trends in small time series (Eastman et al., 2009). Furthermore, past research on water quality trends in the EAA has also utilized median slopes in this manner (Daroub et al., 2011). Median slopes were calculated using the package 'mblm' version 0.12.1 in R.

We then used Mann-Kendall (MK) trend analysis to quantify the consistency of changes in UAL and P:R within both the wet and dry seasons. The MK test is a non-parametric test, which makes no assumption regarding the underlying probability distribution of the variable and is less affected by missing values or outliers (Bandyopadhyay et al., 2011). The MK test was conducted in ArcGIS Pro 2.2 via the Create Space-Time Cube from Defined Locations tool. The tool was first used to aggregate the monthly data into annual wet and dry season medians, then the MK statistic was calculated in the form of a z-score within each basin. A p-value associated with each MK result was also calculated and a 90 % confidence level or greater ($p < = 0.10$) was considered significant. The MK z-scores were then mapped and categorized by BMP, Primary Basin Manager, and Dominant Soil Type to identify and compare relationships between key water quality driver variables. We derived our hypotheses (Table 1) based on prior literature on the biophysical and management contexts of water quality outcomes. Our choices of hypothesized biophysical drivers are based on findings by Izuno et al. (1999), Daroub et al. (2009, 2011), and Lang et al. (2010). Our examination of management drivers is more exploratory but informed by prior findings reported in Yoder and Roy Chowdhury (2018) and Yoder (2019).

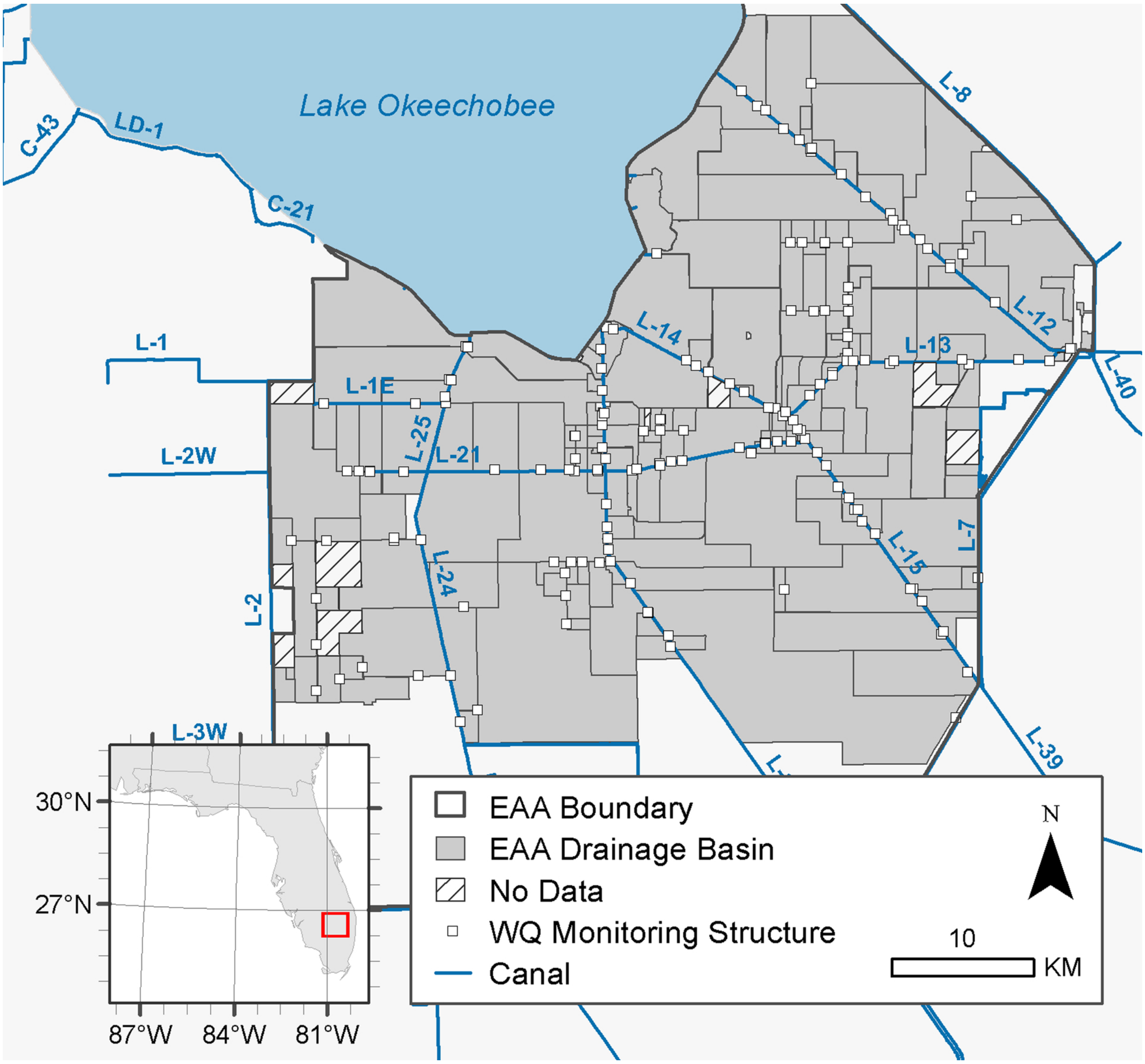

Fig. 1. Map of the study area, the Everglades Agricultural Area, with individual drainage basins and water monitoring sites, which are located at hydraulic pumps used for irrigation and drainage by farm operators.

5. Results: long-term trends in water management

5.1. Summary statistics of water management data

This study set out to identify the longitudinal water quality trends that have occurred over the past two decades and how the underlying management and biophysical drivers have shaped those long-term outcomes. We begin by reporting summary statistics on both sets of drivers. Table 2 reveals large variations in rainfall and EAA farms' water management as captured in UAL and P:R for both wet and dry seasons. The wide range of minimum and maximum values is consistent with the precipitation extremes across south Florida's wet and dry seasons. Mean precipitation is nearly three times higher in the wet seasons than in the dry season, while the standard deviation, median, and maximum are also higher in the wet season. These data also reveal the challenges of outliers, as the maximum P:R value is an order of magnitude higher in the dry season when compared to wet season. An exploratory analysis of the monthly data, reveals that the outlier is produced by an usually small interpolated rainfall amount in just one month (February 2001), while drainage volumes remained similar to preceding and succeeding months (see Supplementary Materials). Precipitation declines slightly over the 22-year study period, with no clear increase or decrease in annual ranges (Fig. 2). Heat map raster charts illustrate the seasonal variation in rainfall, including the occasional outlier months of intense dry-season rainfall (Fig. 3).

As expected, the mean, median, and standard deviations in UAL are lower in the dry season than in the wet season. However, the maximum values for UAL are not largely different from wet to dry seasons (Table 1). The heat maps illustrate these general trends, while also revealing outlier dry season months with higher UAL (Fig. 3). We expected that P:R would be lower in the dry season than in the wet season. However, mean P:R is more than five times larger in the dry season, suggesting greater pumping relative to rainfall during those months (Table 1). Given the high maximum value in the dry season, it is possible that outliers, such as hurricanes during August and September, may account for some of the unexpected results (Fig. 2).

We found that Terra Ceia (> 1.3 m) and Pahokee ($0.91 - 1.3$ m) are the predominant soil types found in EAA basins, representing 76 and 66 basins respectively, while Torry (> 1.3 m) is the main type in 10 basins (Fig. 4c). The remaining soil types appear dominant in no more than 3 basins each. We expected to see shallower soils farther south, and the data are generally consistent with this expectation, despite some

Table 1
Water management variables and their hypothesized relationships with UAL and P:R trends.

Explanatory Variable	Description	Hypothesized Relationship	
Biophysical Drivers		UAL	P:R
Seasons			
–Wet Season	Monthly rainfall (May-Oct). More rainfall, especially intense events can require more pumped drainage to avoid crop damages from flooding.	+	+
–Dry Season	Monthly rainfall (Nov-Apr). Less rainfall reduces risk of crop flooding, decreasing flows and pumped drainage.	–	–
Soil Types	Majority soil type within a basin. Greater soil depth limits potential for adsorption with limestone bedrock to minimize phosphorus loss but allow greater on-farm water storage		
–Torry	> 1.3 m deep.	+	–
–Terra Ceia	> 1.3 m deep	+	–
–Pahokee	0.91 – 1.3 meters deep	–	+
Managerial Drivers		UAL	P:R
Best Management Practices	Basin managers are required to implement best management practices under the regulations to restore water quality.		
–0.5-inch BMP	Lower rainfall threshold to guide pumping decisions. Can indicate the presence of vegetable crops or shallower soils that require quicker drainage. Unknown whether this driver will reveal statistically significant relationships.	–/+	–/+
–1.0-inch BMP	Higher rainfall threshold to guide pumping decisions. Can indicate the presence of sugarcane, which is more flood tolerant and requires less pumping.	–	–
Basin Manager	Different companies are the sole or majority manager for each basin and may affect the consistency or variability of water management. Prior research indicates differences in the potential coefficients but that all managers have reduced their overall phosphorus loads based on public records.		
–Florida Crystals		–	–
–US Sugar		–	–
–Growers Coop		–	–

Table 2
Summary statistics table for UAL, P:R, and rainfall for wet and dry seasons in the EAA from 1994-2016.

Season	Statistics	UAL (kg P ha^{-1})	P:R (m^3:m^3)	Rainfall (mm)
	Mean	0.2	0.8	171.9
	Median	0.1	0.5	170.3
Wet	Standard Deviation	0.5	1.8	79.9
	Minimum	0.0	0.0	1.4
	Maximum	17.8	107.0	531.5
	Mean	0.1	4.6	58.5
	Median	0.0	0.1	43.4
Dry	Standard Deviation	0.3	211.7	54.1
	Minimum	0.0	0.0	0.0
	Maximum	15.7	25,971.6	281.6

variation (Fig. 4C). The location of the 0.5-inch rainfall detention BMP is also consistent with the location of shallower, Pahokee soils, though not uniformly matched. However, it is more consistently present in the north-to-south gradient, which is consistent with an expectation of shallower soils further from Lake Okeechobee (Fig. 4B). The distribution of different managers presents a more nuanced picture. Only farms within the Growers Coop cultivate vegetables in addition to sugarcane. Grower Coop-managed basins are distributed across both deeper and shallower soil types. However, the presence of deeper soils and 0.5-inch BMPs could indicate areas where vegetables are more likely to influence UAL and P:R (Fig. 4).

5.2. Magnitude of water management trends

We examined TS median slopes to gauge the magnitude of upward or downward trends for UAL and P:R to examine the outcomes of basin managers' responses to seasonal rainfall, with the risk of crop flooding mainly concentrated during the wet season. During the wet season, we found that nearly one-half of the basins (80) experienced statistically significant decreases in the magnitude (TS) of UAL (Fig. 5). Of these 80 basins, 63 had weakly downward trends with decreases in magnitude of -0.01. Another 17 had slightly larger decreases in magnitude of -0.01 to -0.04. However, in contrast to our expectations, we found that 23 basins had statistically significant increases in the magnitude (TS) of UAL during the wet season, while 63 basins had no UAL trends. However, most of the upward trends in UAL were weak, with increases in magnitudes of 0.005. The highest magnitude increases were 0.02, present in only three basins. While these are weakly upward trends, it is clear that there is a split in UAL trends across basins and that not all basin managers are contributing to staying within the shared pollution cap.

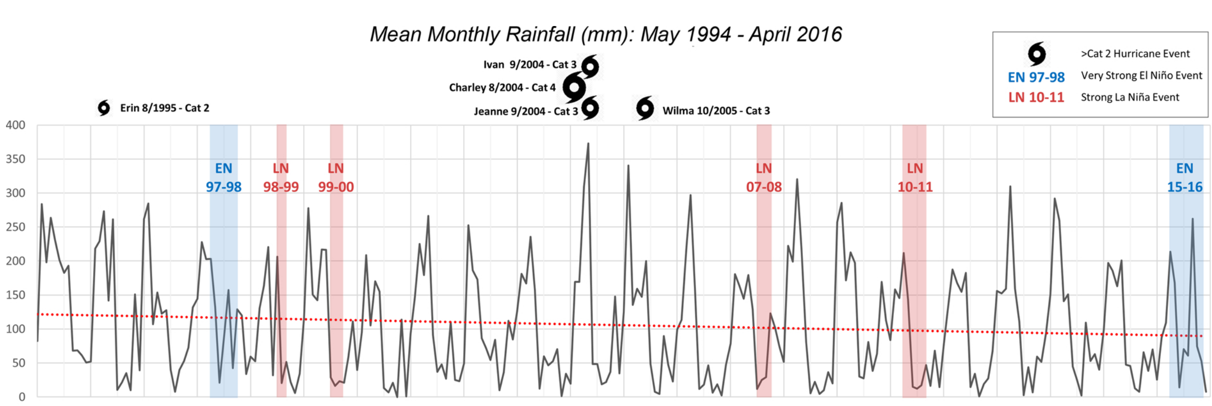

Fig. 2. Mean monthly rainfall (mm) and its long-term trend in the Everglades Agricultural Area from 1994–2016.

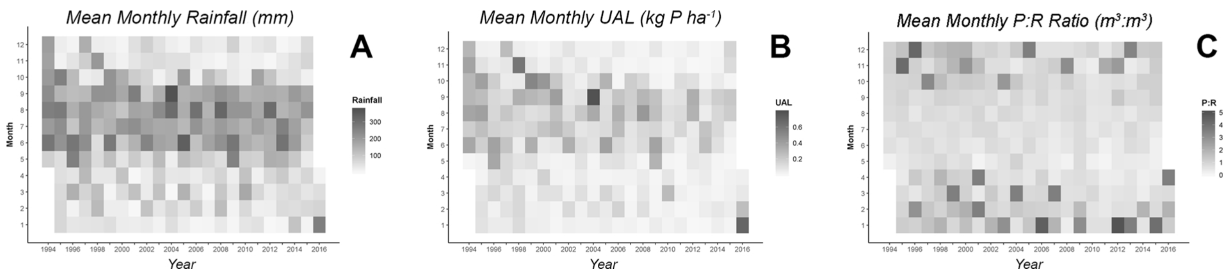

Fig. 3. Heat map raster charts of (a) mean monthly rainfall (mm), (b) unit area load (kg P/ha), and (c) pumping-to-rainfall ratio (unitless).

Overall, the majority of basins demonstrated weak to strong downward trends showing reduced amounts of phosphorus enrichment in EAA drainage waters.

UAL outcomes during the dry season displayed notable downward trends, with more than one-third of basins showing downward trends (61) with statistically significant decreases in magnitude (TS). In contrast, there were only three basins with statistically significant changes of increasing magnitudes in UAL. However, a majority of basins (102) had no statistically significant upward or downwards trends. Overall, these results tend to support our hypothesis that there would be better water management outcomes in the dry season because the risk of flooded crops is at its lowest. When considering the number of upward trends, this outcome is clearly supported, since there are far more upward trends during the wet season. However, the lack of upward or downward trends in a majority of basins during the dry season is unexpected, especially when there are more basins with statistically significant downward trends in the wet season (80) than in the dry season (61).

We found high variability among basins with both increasing and decreasing P:R trends. During the wet season, a plurality of basins had statistically significant decreasing magnitudes (72), while a large minority of basins had statistically significant increases in the magnitude of P:R (40). There remaining 54 basins had no statistically significant trends (Fig. 5). We did not anticipate such a large number of basins would have increases in the magnitude of their P:R outcomes, which indicates that water management has worsened across many basins. Despite this, nearly half of the basins have achieved improvements in their drainage in response to rainfall during the most challenging months of the year.

In the dry season, P:R trends are similar to UAL dry-season trends. One-third of basins experienced statistically significant decreases (56) in the magnitude of P:R outcomes, while 10 basins experienced statistically significant increases. The remaining 100 basins did not have statistically significant trends. Similar to our hypothesis for UAL trends, the P:R outcomes partially support our expectations. However, the presence of upward trends and the number of basins with no trends are surprising.

5.3. Consistency of water management trends

We analyzed the consistency of water management outcomes by testing for monotonicity of trends (M-K statistic) in UAL and P:R for each basin, to see whether basin managers were making sustained improvements under the shared pollution cap. We anticipated that given the long-term improvements in water quality documented by state monitoring (see Taylor et al., 2019) that nearly all basins would display sustained downward trends.

However, we found that a majority of basins had no statistically significant trends (91) for UAL during the wet season. Of the basins with statistically significant trends, a large portion had downward UAL trends (62), while 13 basins had statistically significant upward UAL trends (Fig. 6). The results for the dry season were more unexpected, since lower rainfall would imply a lower risk of crop flooding, and concomitantly, lower pressure for flood risk management via pumping. A large majority of basins (124) had no statistically significant UAL trends. The next largest group of basins had statistically significant downward UAL trends (37), while only five basins had upward trends.

Similarly, a majority of basins (101) had no statistically significant trend in P:R outcomes during the wet season, followed by 44 with downward trends and 21 still showing upward trends. There were 129 basins with no M-K trend in P:R during the dry season. Of the remainder, 33 basins had statistically significant downward trends while only 4 had upward trends. In the dry season, when there is little risk of flooding, it is surprising that so few basins had downward P:R trends. This indicates that reduced pumping in response to rainfall has not steadily improved over time, except among a minority of basins in both wet and dry seasons.

5.4. Role of management and biophysical drivers

Lastly, we looked at the management and biophysical drivers shaping water quality outcomes using ANOVA to compare Theil-Sen trends and Tukey HSD Post-Hoc comparison between individual basin managers, soil types, and BMPs. The ANOVA comparison of mean T-S trends revealed significant differences between managers for wet-

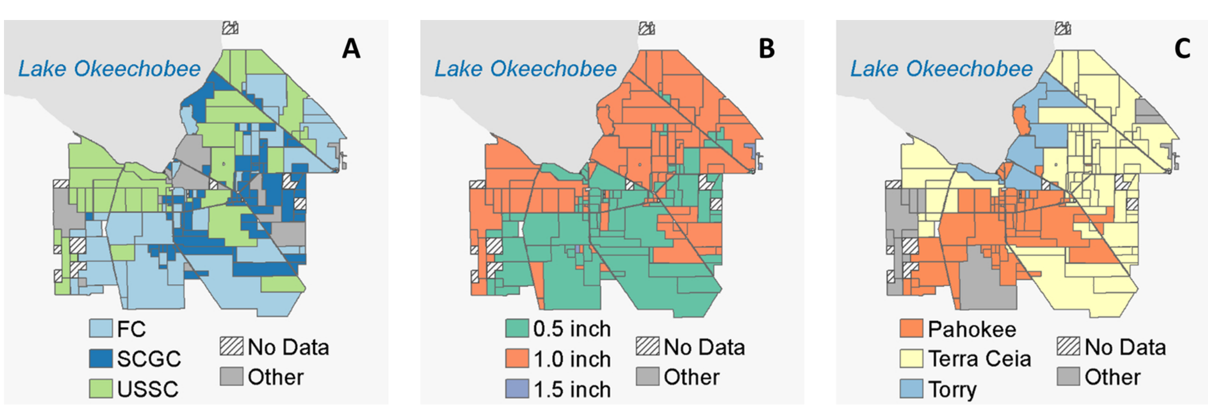

Fig. 4. Distribution of (a) majority basin manager, (b) rainfall detention best management practice levels, and (c) dominant soil types.

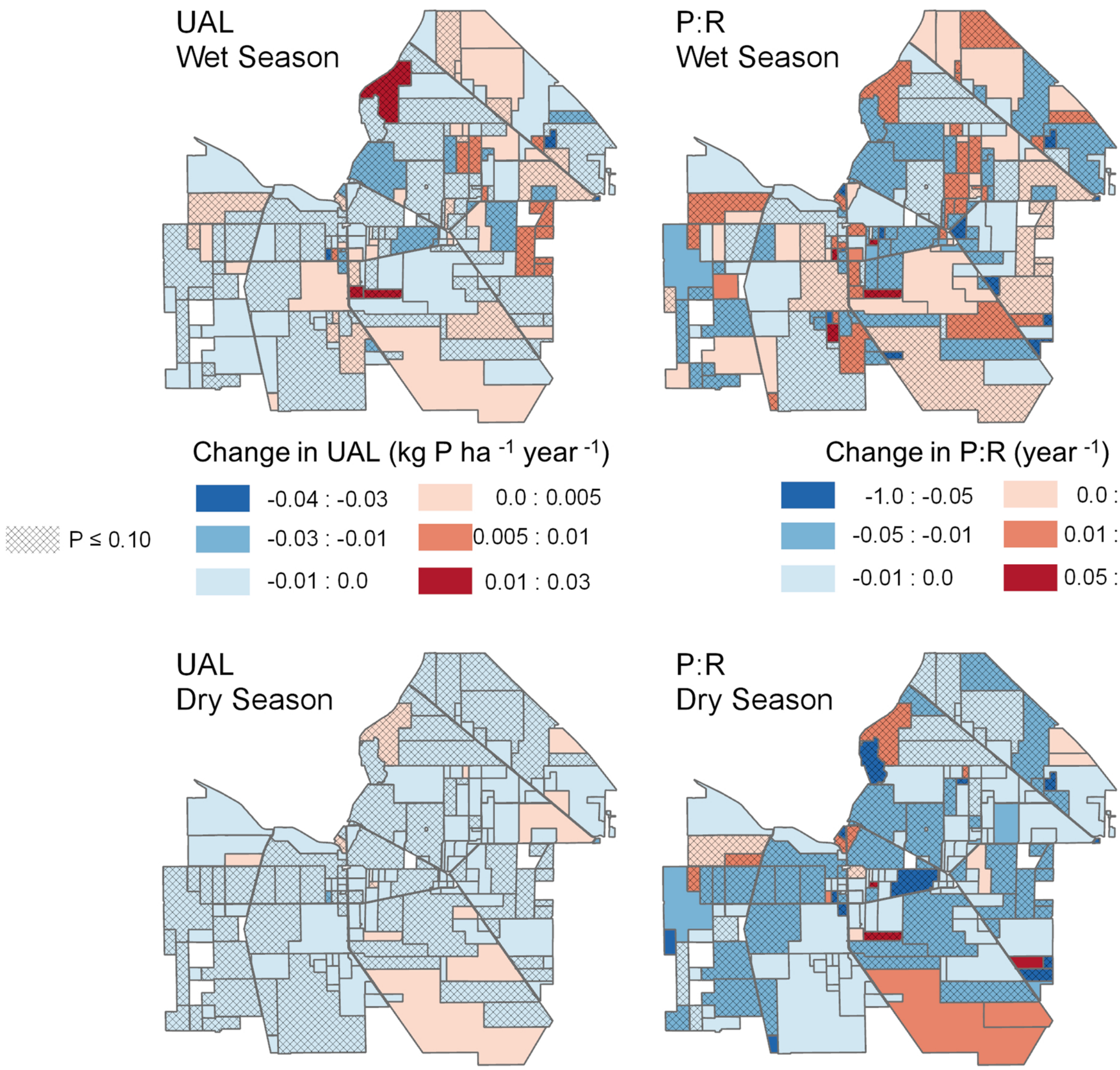

Fig. 5. Changes in the magnitude of water management trends (Theil-Sen median slopes). Crosshatches indicate basins with statistically significant trends at the p = 0.1 % level.

season P:R (strongly significant) and UAL (weakly significant) outcomes (Table 3). Tukey HSD Post-Hoc comparisons suggest that the Sugar Cane Growers Coop and Florida Crystals had significantly distinct magnitude and consistency in their water quality (both UAL and PR) trends during the wet season (Tables 4 and 6). For dry season UAL (MK), there is a small, statistically significant difference between US Sugar and Florida Crystals. In contrast, the ANOVA of mean Mann-Kendall z-scores shows that the primary basin manager is statistically significant in both wet and dry seasons for negative UAL and P:R trends (Table 5).

In contrast to our expectations, we did not find any statistically significant relationships between our three soil types and UAL or P:R outcomes (Tables 3 and 5). The different thresholds for rainfall detention BMPs were significantly related to the consistency of declining UAL trends during the dry season UAL and declining P–R trends in the wet season (Table 5). We did not find any statistically significant differences for the ANOVA results of mean Theil-Sen median slopes (Table 4). The Tukey HSD Post-Hoc comparison showed no statistically significant differences between 0.5-inch and 1.0-inch BMPs but suggested a statistically significant difference between the 1.0-inch and 1.5-inch BMPs. However, there were only two basins in our dataset with 1.5-inch BMPs. Both basins were ranches with pasture and no pumped drainage. In brief, the primary basin manager, particularly Florida Crystals, appears to have the main effect on UAL and P:R trends, while rainfall detention BMPs generally, but not specifically based on different thresholds, also had a small effect.

6. Discussion: effectiveness of devolution to generate long-term improvements in water quality

Our findings show that nearly half of basins made substantial changes in their water management that has resulted in greatly decreased TP enrichment in their drainage water. These improvements in water quality have been largely but not uniformly maintained, as indicated by TS and MK trends for UAL and P:R. The Theil-Sen slopes indicate that the magnitude of the decrease in UAL and P:R was a substantial shift in management, especially during the wet season. This contrasts with the consistency trends, which suggest that while more basins decreased UAL and P:R than increased them, there was also great variability in the strength of negative and positive trends, as well as in the significance of those trends, especially during the dry season. More surprising were the lack of detectable consistency trends in a majority of basins for UAL and P:R. If basin managers prioritized reducing the amount of pumped drainage, we would expect more basins with

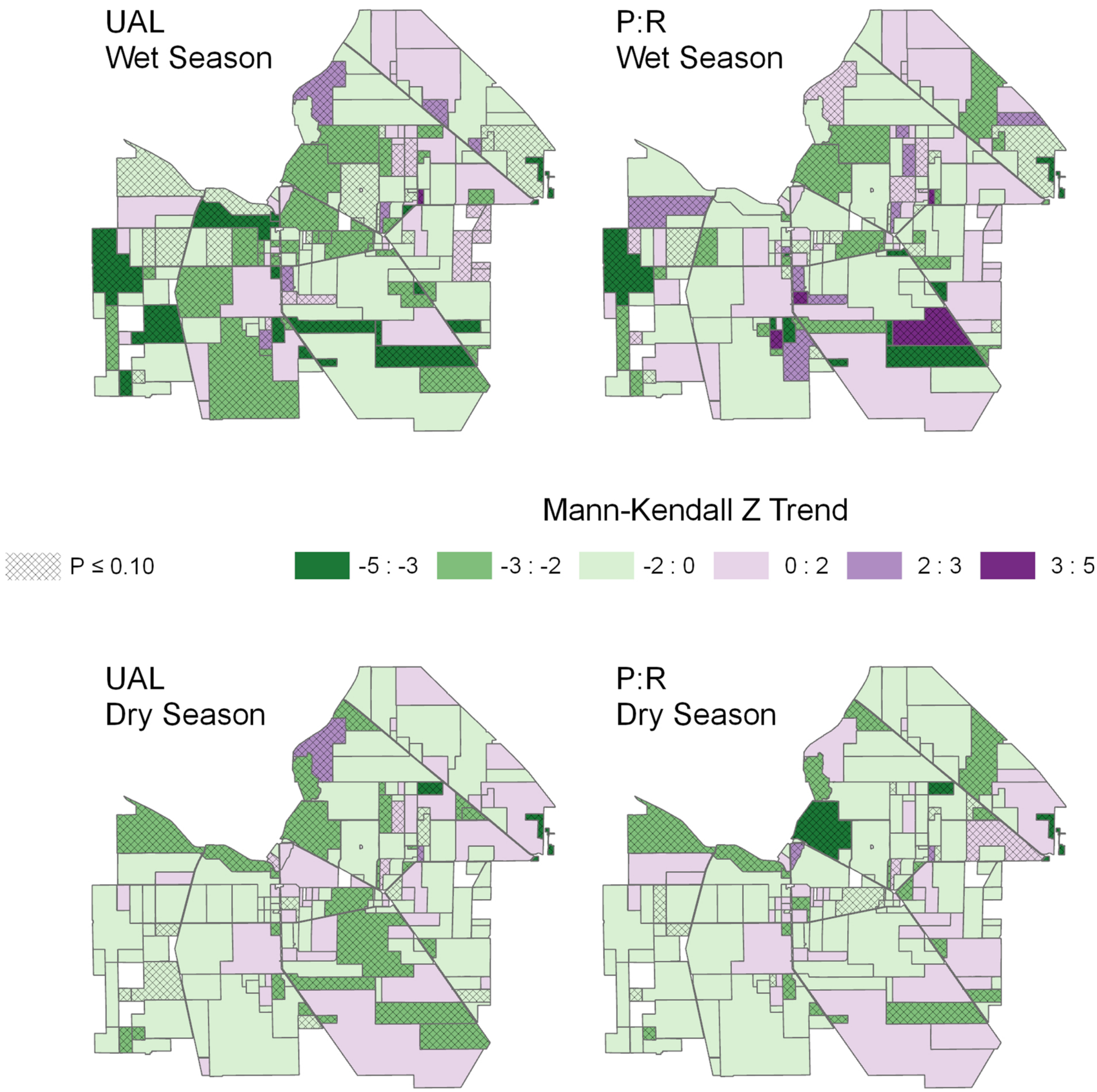

Fig. 6. Consistency of water management trends (Mann-Kendall z-scores). Crosshatches indicate basins with statistically significant trends at the p = 0.1 % level.

statistically significant negative MK trends during the dry season than in the wet seasons for both dependent variables, since the lower precipitation amounts should reduce concerns about crops being at risk of flooding.

One explanation for these differences is that the changes in management represent, in effect, a large change in practice which greatly reduced TP enrichment. An example of this would be the use of rainfall detention BMPs to reduce the timing and volume of water being pumped (Yoder and Roy Chowdhury, 2018). However, once this practice is implemented, it is possible that basin managers could not

Table 3
ANOVA comparison of means for managerial and biophysical factors shaping the magnitude (Theil-Sen statistic) of water quality and management trends. The abbreviations for Basin Managers represent Florida Crystals (FC), Sugar Cane Growers Cooperative (SCGC), and US Sugar Corporation (USSC).

	Season	Primary Basin Manager			Dominant Soil Type			Rainfall Detention BMP		
		FC	SCGC	USSC	Pahokee	Terra Ceia	Torry	0.5 Inch	1.0 Inch	1.5 Inch
UAL kg P ha^{-1} yr^{-1}	Wet	−0.0005	−0.0033	−0.0042	−0.0029	−0.0023	−0.003	−0.0032	−0.0025	−0.0026
	p-value		**0.055**			0.89			0.84	
	Dry	−0.0005	−0.0006	−0.0014	−0.0007	−0.0004	−0.0004	−0.0008	−0.0006	0
	p-value		0.19			0.57			0.79	
P:R Yr^{-1}	Wet	0.003	−0.013	−0.003	−0.004	−0.006	−0.02	−0.006	−0.006	−0.028
	p-value		**0.013**			0.22			0.55	
	Dry	−0.008	−0.022	−0.014	−0.009	−0.009	−0.008	−0.022	−0.01	0
	p-value		0.71			0.99			0.63	

Table 4
Tukey HSD post-hoc analysis of ANOVA comparison of means for Theil-Sen trends.

	Season	Basin Manager	Difference	Lower Bound	Upper Bound	Adjusted P-Value
UAL kg P ha^{-1} yr^{-1}	Wet	SCGC-FC	−0.00284	−0.00622	0.00054	0.118517
		USSC-FC	**−0.00378**	**−0.00774**	**0.00017**	**0.063914**
		USSC-SCGC	−0.00094	−0.00447	0.002578	0.801307
P:R yr^{-1}	Dry	**SCGC-FC**	**−0.0157**	**−0.02847**	**−0.00294**	**0.011458**
		USSC-FC	−0.00629	−0.02123	0.00864	0.579599
		USSC-SCGC	0.009411	−0.00389	0.022716	0.218465

Table 5
ANOVA comparison of means for managerial and biophysical factors shaping the consistency (Mann-Kendall statistic) of water quality and management trends. The abbreviations for Basin Managers represent Florida Crystals (FC), Sugar Cane Growers Cooperative (SCGC), and US Sugar Corporation (USSC).

	Season	Primary Basin Manager			Dominant Soil Type			Rainfall Detention BMP		
		FC	SCGC	USSC	Pahokee	Terra Ceia	Torry	0.5 Inch	1.0 Inch	1.5 Inch
UAL kg P ha^{-1} yr^{-1}	Wet	−0.32	−1.19	−0.99	−1.04	−0.82	−1.46	−1.08	−0.88	−2.87
	p-value		**0.012**			−0.42			0.18	
	Dry	−0.08	−1.03	−0.77	−0.75	−0.62	−0.70	−0.87	−0.57	−2.45
	p-value		**0.00013**			0.82			**0.039**	
P:R Yr^{-1}	Wet	0.33	−0.88	−0.50	−0.43	−0.51	−1.13	−0.56	−0.57	−3.72
	p-value		**0.00050**			0.45			**0.020**	
	Dry	−0.03	−0.90	−0.88	−0.54	−0.73	−1.12	−0.72	−0.66	−2.49
	p-value		**0.00014**			0.284			0.095	

Table 6
Tukey HSD post-hoc analysis of ANOVA comparison of means for Mann-Kendall trends.

	Season		Difference	Lower Bound	Upper Bound	Adjusted P-Value
UAL kg P ha^{-1} yr		*Basin Manager*				
	Wet	**SCGC-FC**	**−0.87716**	**−1.56891**	**−0.18541**	**0.00878**
		USSC-FC	−0.66988	−1.47916	0.139404	0.12597
		USSC-SCGC	0.207279	−0.51381	0.928369	0.775255
	Dry	**SCGC-FC**	**−0.95007**	**−1.46763**	**−0.43251**	**7.52E-05**
		USSC-FC	**−0.68899**	**−1.29449**	**−0.08349**	**0.021359**
		USSC-SCGC	0.261083	−0.27843	0.800599	0.48773
		Rainfall Detention BMP				
	Dry	1.0 BMP-0.5 BMP	0.299696	−0.14594	0.745334	0.252543
		1.5 BMP-0.5 BMP	−1.57499	−3.61494	0.464951	0.164269
		1.5 BMP-1.0 BMP	−1.87469	−3.91166	0.162285	**0.078219**
P:R Yr^{-1}		*Basin Manager*				
	Wet	**SCGC-FC**	**−1.21202**	**−1.93014**	**−0.49389**	**0.000296**
		USSC-FC	**−0.82881**	**−1.66895**	**0.011325**	**0.054054**
		USSC-SCGC	0.383203	−0.36538	1.131787	0.448107
	Dry	**SCGC-FC**	**−0.87064**	**−1.37126**	**−0.37001**	**0.000186**
		USSC-FC	**−0.84689**	**−1.43258**	**−0.26121**	**0.00229**
		USSC-SCGC	0.023744	−0.49812	−0.545604	0.993626
		Rainfall Detention BMP				
	Wet	1.0 BMP-0.5 BMP	0.141694	−0.46489	0.748279	0.845331
		1.5 BMP-0.5 BMP	**−3.15876**	**−5.93545**	**−0.38206**	**0.021366**
		1.5 BMP-1.0 BMP	**−3.30045**	**−6.0731**	**−0.5278**	**0.015037**
	Dry	1.0 BMP-0.5 BMP	0.061503	−0.37464	0.497642	0.940541
		1.5 BMP-0.5 BMP	**−1.77654**	**−3.77301**	**0.219925**	**0.092005**
		1.5 BMP-1.0 BMP	**−1.83804**	**−3.8316**	**0.155516**	**0.077519**

continually improve or fine-tune the practice further. Alternatively, it is possible that because of initial changes and success in reducing TP loads, basin managers became less vigilant in maintaining those improvements. Either could potentially explain why there were more changes in the magnitude of UAL and P:R outcomes than in the consistency of trends over time. Based on these two outcomes, we argue that while devolving management to basin managers has had a positive overall effect for improving water quality, including additional incentives or restrictions to encourage ongoing improvements could make an important difference if similar approaches are considered in other contexts. Despite this shortcoming, the TS trends indicate that there was a widespread change in management that has made a substantial difference in water quality. Another important factor, specific to the EAA, is the flood-tolerance and P-assimilation capacity of sugarcane (Glaz, 1995). Because it is the largest crop by area, it contributes both qualitatively and quantitively to reducing UAL at this regional scale. The assimilative capacity of other crops for N and P in other areas may be very distinct from the EAA context.

Other findings also indicate that management, rather than precipitation or soil type, has been the most substantial driver of UAL and P:R outcomes. We expected to find a difference between the 0.5-inch and 1.0-inch BMP thresholds as an indication that precipitation was a key limitation on water management, especially across wet and dry seasons. This does not appear to be the case. Even though dry season UAL and wet season P:R showed negative trends significantly related to BMPs, there was no post-hoc difference between the 0.5-inch and 1.0-

inch thresholds. Additionally, BMPs were not statistically significantly related to Theil-Sen trends in UAL or P:R. Basin managers have multiple ways to reduce UAL beyond BMPs related to pumping thresholds. These include dredging canals, more rigorous soil erosion control measures, and banding instead of broadcasting fertilizer, among other measures (e.g., Adorisio et al., 2006). One plausible interpretation for our results is that the overall improvements in water quality suggest that management efforts matter cumulatively, where rainfall detention BMPs represent one component in the overall shift in practices but have not necessarily functioned as a tool that has been or can be fine-tuned over time. One could also argue that more fine-grain analysis is needed to look at pumping differences in response to rainfall events at daily or weekly intervals, with different time lags tested. Monthly data, which we utilized in this analysis, may be too coarse to reveal the precise correlation between some of these BMPs and water quality improvements.

Lastly, identifying how best to deploy monitoring efforts to inform farm management to improve water quality remains a key challenge. Given that in other cases where BMPs have been mandated but have not generated substantial changes in farm management (e.g., Barnes et al., 2013), tying monitoring data to relevant farm management decisions may offer one way to encourage farm managers to continually improve. The number of positive trends for our dependent variables that were statistically significant indicates that some additional use for monitoring data as a feedback or accountability tool is warranted. In addition, more timely data drawing on trend analysis, in addition to annual snap shops that have been done by the SFWMD, would provide a fuller picture on a range of management challenges. This strikes us as particularly important given the trends in governance for including a greater range of stakeholders in decision-making processes. Our prior research has demonstrated the importance of peer communication and pressure in the EAA as central to the shift in management that is evident in the trend data we presented here (Yoder and Roy Chowdhury, 2018).

While this study demonstrated improvement in agricultural water management by focusing on long-term trends, we faced some constraints in the types of data that were available. While there were ample monitoring sites, we were limited by the permit records to a single five-year period for information on basin-level managers (2012–2017). Earlier records did not list managers, which means we could neither account for changes in rental agreements nor for consolidation within the industry. Similarly, the location of crops was not available at the basin scale. Crop choices were listed with overall acreages for entire permits; the main companies often consolidated many basins together within a single permit. Moreover, the variables collected in the dataset by Lang et al. (2010) and related research were unavailable in publicly accessible data.

7. Conclusions

This study shows that devolving responsibility for BMP adoption under a mandatory nutrient limit can deliver improved water quality at larger spatial scales and over longer time frames. This advances our understanding of the efficacy of using BMPs at larger scales to mitigate nutrient pollution (e.g., Liu et al., 2017; Hanrahan et al., 2018; Melland et al., 2018; Falcone et al., 2018). In addition, it reveals both strengths and weaknesses of flexibility in governance arrangements. Critics of devolving implementation to farmers could reasonably point to the statistically significant increasing trends in magnitude and consistency for P:R as strong indications that this flexibility leads to some suboptimal outcomes. However, given the dynamism of the climate in south Florida, and the broader challenges of measuring NPS pollution effectively across scales, a flexible approach has generated long-term improvements in water quality. The variability of rainfall remains an important challenge, as does the difficulty of monitoring NPS pollution in areas lacking the intensive monitoring system that exists in the EAA. While we did not address climate change in this study, changes in the intensity of rainfall events and prolonged dry spells, such as in the U.S. Midwest, indicate that these monitoring needs are likely to increase in importance in the future. Agri-environmental policies seeking to improve water quality would benefit from greater attention to combining flexible management approaches, such as a range of BMP options, with frequent and extensive monitoring to provide both feedback for managers and accountability toward long-term environmental goals.

Declaration of Competing Interest

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

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Appendix A. Supplementary data

Supplementary material related to this article can be found, in the online version, at doi:https://doi.org/10.1016/j.agee.2020.107070.

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