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WETLAND RESTORATION

Phosphorus Fluxes in a Restored Carolina Bay Wetland Following Eight Years of Restoration

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Abstract

Restoring wetlands on agricultural land can release soil phosphorus (P) to surface waters. Phosphorus is a limiting nutrient in many freshwater systems, thus restricting its release will improve surface water quality by preventing algal blooms. A P balance was used to examine how P was cycling in a Carolina Bay wetland eight years after restoration from prior-drained agricultural land. The change in soil P was evaluated between archived samples taken at restoration (2005), and eight years after restoration (2013). Measured P fluxes included atmospheric deposition, plant uptake, and loss to surface water outflow. The soil total P pool at the time of restoration was 810 kg P ha^{-1}. No significant ($\alpha = 0.05$) decrease in the soil P pool was observed over the eight years. Atmospheric deposition contributed 1.0 kg P ha^{-1} yr^{-1}, plants incorporated 3.3 P ha^{-1} yr^{-1} into woody biomass and 0.4 kg P ha^{-1} yr^{-1} as forest floor litter, and 0.2 kg P ha^{-1} yr^{-1} was lost to surface waters draining the wetland. Because the loss of P to surface waters was small, and because runoff water concentrations of P declined through this period of study to concentrations below those likely to cause eutrophication (< 0.1 mg L^{-1}), we concluded that the wetland was not contributing to the degradation of surface water quality of nearby streams following restoration. Further, isolated wetlands such as that studied may be promising sites for future wetland mitigation projects due to limited impacts on surface water quality.

Keywords Wetland restoration · Water quality · Isolated wetlands · Converted wetlands · Agricultural land

Introduction

Over 50% of the original wetlands in the conterminous 48 states of the U.S. were drained primarily for food production between 1780 and 1980 (Dahl and Allord 1996). Since 1977, federal and state programs have been enacted to reverse the loss by restoring drained areas to their original wetland condition, frequently by plugging or filling drainage ditches. Wetland restoration is accomplished, in part, to improve water quality. However, in cases where wetlands are restored from agricultural land that is high in phosphorus (P) from years of fertilization, saturated and reduced soil conditions cause P to be released from the newly flooded wetland contributing to eutrophication of nearby surface waters. Between 1997 and 2001 there was an estimated annual net gain of 13,400 ha of wetlands nationally due to restoration of agricultural fields, while between 2001 and 2003 the annual net gain more than doubled from previous periods (USDA-NRCS 2013). Based on estimates from the North Carolina Division of Mitigation Services, approximately two-thirds of restored wetlands in North Carolina originated from drained and fertilized agricultural lands – equivalent to approximately 1,500 ha since 1999 (Smith 2011). Given that wetland restoration is increasing in the U.S., in part to improve water quality, it is critical to determine whether restoration will contribute to pollution of P-sensitive watersheds.

Phosphorus dissolution is the process in which P absorbed onto soil solids is released into the soil solution (Moorberg et al. 2015), making it susceptible to be lost from the soil to leaching, runoff, or release to ponded water (Aldous et al. 2005; Duff et al. 2009; Ardon et al. 2010). Phosphorus dissolution from wetlands restored from agriculture has been

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observed around the world. Recent studies of P dissolution have been done on peat soils in the Netherlands (Van Dijk et al. 2004), soils used for dairy production in Florida (Pant and Reddy 2003), restored lake fringe in Oregon (Aldous et al. 2007; Duff et al. 2009), and agricultural soils in the North Carolina Coastal Plain (Ardon et al. 2010), and a Carolina Bay complex in North Carolina (Bruland et al. 2003). These studies have shown that P dissolution is largely driven by iron (Fe) reduction processes (Reddy and DeLaune 2008), along with other mechanisms including ligand exchange (Earl et al. 1979; Lopez-Hernandez et al. 1986; Violante et al. 1991; Gerke 1992), P mineralization from drying and rewetting cycles (Song et al. 2007), changes in pH (Jackson 1964; Ponnamperuma 1972; Stumm and Morgan 1981), and increased P diffusion (Turner and Gilliam 1974a, b).

To ensure that wetland restoration and management practices do not contribute to pollution of nutrient sensitive streams, a better understanding of P fluxes within and out of wetlands restored from agricultural land is needed to identify potential management strategies that will reduce P loss. This study focused on a previously cultivated Carolina Bay wetland, known as Juniper Bay, that was restored as a wetland and then monitored for P in drainage waters in subsequent years (Vepraskas et al. 2010; Moorberg et al. 2015, 2017). The objectives of this study were to estimate a P budget for a wetland restored from an agricultural field by determining P fluxes for soil storage, atmospheric input, and loss from drainage water and estimating plant P uptake.

Materials and Methods

Juniper Bay is located in Robeson County, NC approximately 10 km south of Lumberton, NC (34°30′30″N 79°01′30″W). In 1999, the North Carolina Department of Transportation purchased this drained Carolina Bay wetland to mitigate the destruction of nearby wetlands caused by highway construction (Ewing 2003). The Bay is oval-shaped, oriented lengthwise along a northwest-southeast transect, and is virtually flat with an area of 291 ha. Soils in Juniper Bay include approximately 60% (186 ha) mineral soils (Leon sand; sandy, siliceous, thermic Aeric Alaquods; USDA Soil Taxonomy; Soil Survey Staff 1999) primarily at the edges (Fig. 1, SC and SS mapping units), with organic soils (Ponzer muck; loamy, mixed, dysic, thermic Terric Haplosaprists; USDA Soil Taxonomy; Soil Survey Staff 1999), occupying the remainder (105 ha) at the center (Fig. 1, OC and OS mapping units). This Bay was drained for agriculture beginning in 1971 by excavating a perimeter ditch around the edge of the Bay, and installing primary and secondary ditches within the Bay to facilitate drainage into a single surface water outlet on the southwestern edge of the wetland (Vepraskas et al. 2005). Juniper Bay was fertilized and limed annually to meet soil-test recommendations. It remained in crop production until 2001. Preliminary restoration efforts began in June 2003, and wetland hydrology was restored in 2005 by filling primary ditches and plugging tertiary ditches, leaving only the perimeter ditch intact. That perimeter ditch drains into one outlet on the southwest side of the Bay.

A P-balance was developed to better understand the nature and relationships of P fluxes in and out of Juniper Bay following restoration. The budget includes P inputs that are "new" sources of P going into the soil, P outputs that are losses of P from the soil, and an error term that is the remaining difference between inputs and outputs and/or error or fluxes that have gone unrecognized. We hypothesized that atmospheric deposition (P_{ATM}) is a major mechanism adding new P into the Bay, but groundwater inflow (P_{GI}) could also be contributing P. Major ways for P to be removed from the soils in Juniper Bay include plant uptake (P_{PL}), groundwater outflow (P_{GO}) and surface water outflow (P_{SO}). Previous work by Pati (2006) showed that the perimeter ditch would intercept most groundwater inflow into the Bay, thus transforming P_{GI} into P_{SO} which would drain out of the Bay through the outflow structure. In addition, Pati (2006) showed that the groundwater outflow component would be intercepted by the perimeter ditch as well, as long as the water levels in the ditch were managed to stay below a critical elevation. Such management of the perimeter ditch is currently practiced so that groundwater outflow from the Bay should be small. Huffman et al. (2007) estimated the net flow of ground and surface water into the Bay from the surrounding landscape was equivalent to 125 mm during the wet months of 2004, with inflows entering the perimeter ditch on the NW, NE, and SE sides of the Bay, and groundwater outflows exiting on the SW side of the Bay. The impact of the perimeter ditch is such that the terms for P_{GO}, P_{SO}, and P_{GI} were combined with the assumption that contributions from groundwater inflow are minimal, and that surface water outflow is primarily from drainage from Juniper Bay. This assumption was tested and validated, as described in the supplementary material (SI Table 1 and SI Table 2; SI Fig. 1, SI Fig. 2, SI Fig. 3, and SI Fig. 4).

The P balance for Juniper Bay can be written with the defined inputs and outputs as:

$$\Delta P_{soil} = P_{ATM} - P_{PL} - [P_{GO} + P_{SO} - P_{GI}] \quad (1)$$

For simplicity, we combined the components P_{GO}, P_{SO} and P_{GI} into one term called $P_{OUTFLOW}$ which was measured collectively at the outflow structure. The modified P balance used for this study is:

$$\Delta P_{soil} = P_{ATM} - P_{PL} - P_{OUTFLOW} \pm E \quad (2)$$

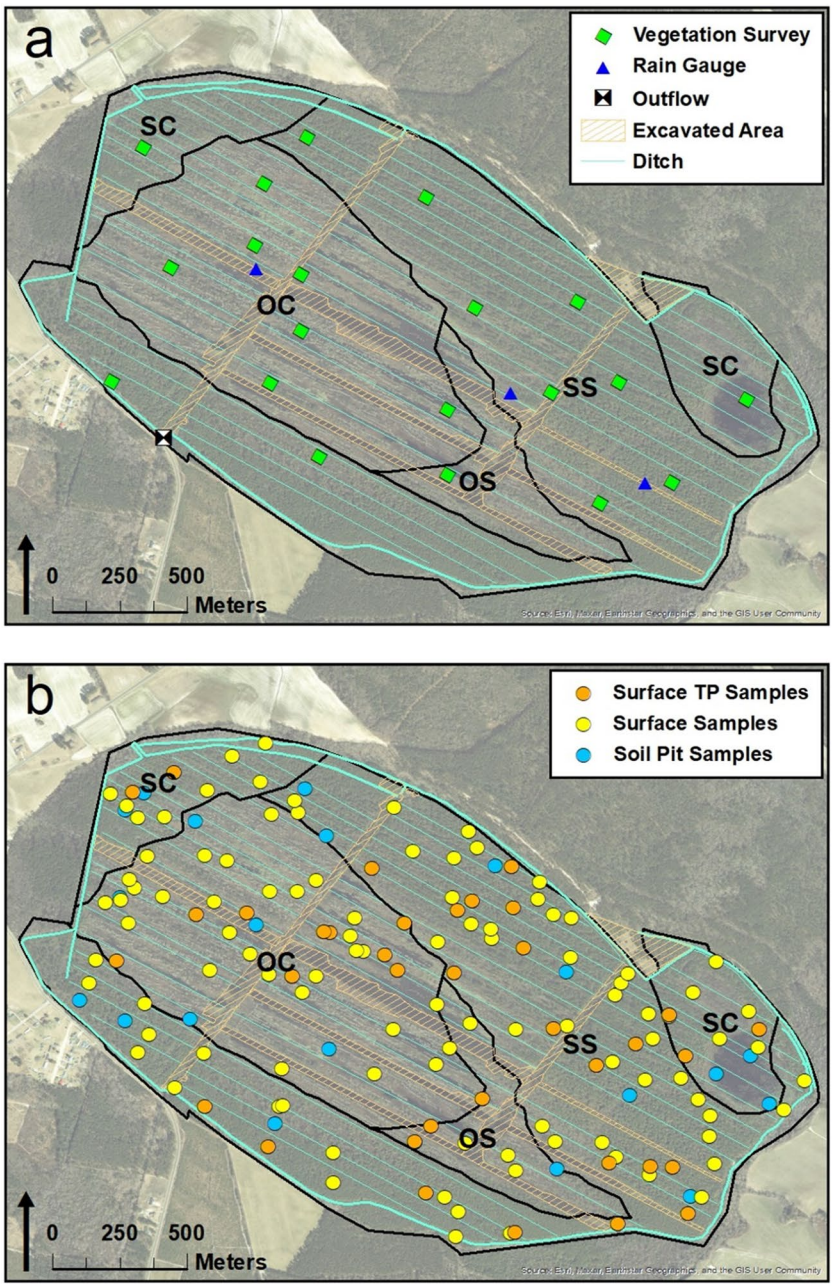

Fig. 1 Maps of Juniper Bay depicting locations of **a**) previous and existing drainage ditches, perimeter ditch outflow, vegetation survey locations (not drawn to scale), rain gauge location, and mineral and organic soil distribution; and **b**) resampled surface soil locations for extractable and total P, and resampled soil pit locations. The mapping units depict four soil conditions, including sands over clayey subsoil (SC), sands over sandy subsoil (SS), organic soil over clayey subsoil (OC), and organic soil over sandy subsoil (OS). The scale of both maps is 1:12,000

The volume of soil considered for Juniper Bay has a horizontal area defined by the perimeter ditch (Fig. 1B) with the soil depth starting at the soil surface and extending to a depth of 100 cm. The 100 cm depth was selected because previous work showed that the P increases from agricultural applications in Juniper Bay were not observed below 100 cm (Ewing 2003).

The change in the soil P pool was determined by measuring total phosphorus (TP_{soil}) in archived soil samples collected prior to restoration in 2005 and comparing those values with TP_{soil} found in samples extracted from the same locations eight years after restoration of wetland hydrology in 2013. The TP_{soil} concentrations for time-zero (2005) were determined from two groups of archived soil samples. Prior to restoration, soils were sampled in 2000 from 48 soil pit locations to a depth of 100 cm (Fig. 1B) by Ewing et al. (2012), and in 2004 on a grid of 700 locations across the Bay. Five samples were collected at each of the 700 grid sampling locations using a 2.2 cm diameter soil push probe at each sampling location to a depth of 30 cm. The samples were separated into 0–15 cm and 15–30 cm depth increments, then composited for subsequent analysis. It was assumed that P concentrations in the subsoil (30–100 cm) had not changed between 2000 and restoration in 2005 based

on the observations made by Ewing et al. (2012) which noted that subsoil P concentrations (4–7 g m^{-3}) were comparable to those found in nearby reference Carolina Bay wetlands (1–3 g m^{-3}). Since Juniper Bay remained artificially drained from 2000 to 2004, soil P should have remained immobile prior to restoration.

For each archived soil sample location, GPS coordinates were recorded in 2005 at the time of sampling, thus allowing new samples to be collected from the same locations. For the 0–15 cm and 15–30 cm depths 138 locations of the 700 total were selected for re-sampling and analysis using an area-weighted, stratified random sampling scheme (Fig. 1B). The samples were collected as described for the original 2004 sampling. The samples were separated into four strata based on the soil mapping units developed for restoration of Juniper Bay during the North Carolina Department of Transportation soil survey. All archived surface and subsoil sites were located using GPS receivers with a wide area augmentation system correction and 2–5 m accuracy.

The study by Ewing et al. (2012) included 48 soil pit locations. These locations consisted of 24 pairs of pits—one at the crest (middle) of the fieldlet, and one adjacent to the ditch. They observed large amounts of disturbance in the ditch pits due to maintenance and dredging of the drainage ditches during agricultural production; therefore, only the crest pits were used in this study. Also, Ewing et al. (2012) studied five pits from soils with histic epipedons at the transition from mineral to organic soils. Because these histic soils represented a small area of the Bay, they were also omitted from this study. The remaining 19 soil pits used in this study are shown in Fig. 1B. The 2013 soil samples were extracted using a 7.6 cm diameter soil bucket auger for all three depths.

All soil samples were submitted to the North Carolina Department of Agriculture Soil Testing Service for analysis of extractable P by the Mehlich-3 method (Mehlich 1984) with P concentrations determined using Inductively Coupled Plasma Mass Spectroscopy. Soil TP analysis was performed on 25% of the re-sampled 0–15 cm and 15–30 cm depth soil samples that were selected at random within each stratum (Fig. 1B). Soil TP analysis was also performed on four representative horizons from each pit location (Fig. 1B). Soil TP was determined by performing a nitric-perchloric acid digestion (Carter 1993) which was analyzed by the North Carolina State University Environmental and Agricultural Testing Service (Raleigh, North Carolina, USA) where P concentrations were determined using Inductively Coupled Plasma Mass Spectroscopy. The TP_{soil} determined on a mass per mass basis were re-expressed as mass P per volume of soil for inclusion into the P-balance. Bulk densities reported by Ewing et al. (2012) for the pre-restoration samples at all depths were used to convert the mass of soil to its equivalent volume. Bulk density was determined again for the 2013 samples for the 0–15 cm and 15–30 cm depths using the core method (Grossman and Reinsch 2002). Samples were collected in triplicate at each of the 19 soil pit locations and averaged across the four soil mapping units for the 0–15 cm and 15–30 cm depths. Bulk densities for the 30–100 cm depths were assumed to be the same as the pre-restoration values as reported by Ewing et al. (2012).

Atmospheric deposition of P was monitored from May 2012 through June 2013 at three locations within the wetland (Fig. 1A), as described by Kreiser (2003). Samplers were installed adjacent to existing rain gauges using a bulk rain water collection apparatus (SI Fig. 5) modeled after Likens et al. (1967) and Johnson and Swank (1973). Samples were collected every two to four weeks and acidified for preservation. Samples were submitted to the North Carolina State University Environmental and Agricultural Testing Service (Raleigh, North Carolina, USA) for determination of dissolved reactive P (DRP) and dissolved total P (DTP). Dissolved reactive P was measured colorimetrically using a multichannel QuikChem 8000 (Lachat Instruments, Milwaukee, WI, USA) using the method of Prokopy and Wendt (1994). Dissolved total P was analyzed colorimetrically simultaneously on the same instrument and the method described by (Liao 2001). The average concentration of DTP for this time period, along with historic rainfall data collected on site were used to estimate P_{atm} from 2005 to 2013 over the entire 291 ha area of the wetland.

Phosphorus uptake and accumulation by trees was estimated for the entire area of Juniper Bay. The North Carolina Department of Environment and Natural Resources (now the North Carolina Department of Environmental Quality, NCDEQ) planted wetland tree saplings throughout Juniper Bay at the time of restoration (NCDEQ 2010). Between 2005 and 2010, NCDEQ (2010) established and maintained 19 vegetation plots for wetland mitigation purposes at Juniper Bay. Those 10 m by 10 m plots were located and expanded by 20 m on all sides to create 30 m × 30 m plots for this tree survey, as shown in Fig. 1A (plots not drawn to scale). Tree species, height, and diameter at breast height (DBH) were recorded for all trees greater than 10 cm DBH within each vegetation plot. Wood biomass was then estimated for each tree using allometric equations from Gonzalez-Benecke et al. (2011) for loblolly pine (*Pinus taeda* L.) and pond pine (*Pinus serotina* Michx.), and allometric equations from Schroeder et al. (1997) and Jenkins et al. (2003) for all other species. Biomass P content was estimated for all species using P concentrations presented by Bedford et al. (1999). The total plot woody biomass per-hectare (kg ha^{-1}) and woody biomass P (kg P ha^{-1}) was determined by summing all of the tree biomass and biomass P within each plot, and dividing by the plot area.

Plant litter (the fibric, Oi soil horizon) samples were collected in October of 2014 from eight randomly selected

vegetation plots. A 1 m by 1 m square PVC frame was laid in the center of each plot. Five large nails were driven into the ground until the nail head was at the litter surface at each of the four corners and one at the center. The litter was then collected for analysis. The heights of each nail above the soil surface were measured and averaged to determine the average depth of plant litter per square meter.

The litter was dried in an oven at 70 °C for four days, then weighed to determine total litter biomass. Subsamples were ground and analyzed by the Environmental and Agricultural Testing Laboratory at North Carolina State University (Raleigh, North Carolina, USA) for C and P analysis. Carbon was analyzed using a PerkinElmer model 2400 CHN elemental analyzer. Phosphorus was determined with a dry-ash method based on the method described by Jones and Case (1990). The percent P by weight (% w/w) for the subsamples was multiplied by the litter dry weight per square meter (g m^{-2}) to estimate the mass of P per square meter (g P m^{-2}) for each site. This value was then converted to kg P ha^{-1}, which represents the amount of P that has accumulated in the litter over the eight years since restoration.

The perimeter ditch surrounding Juniper Bay drains into a single surface water outflow structure at the edge of the Bay (Fig. 1A). Samples from the drainage outlet were taken four times daily from 2010 to 2013 using a Teledyne ISCO automatic water sampler (Teledyne ISCO, Lincoln, Nebraska, USA) and composited into one sample. These samples were collected approximately every 14 days. From 2005 to 2010, manually collected samples (1 L volume) were collected from the center of the channel at the outflow monthly using a bottle attached to a pole. All water samples were acidified for preservation and submitted to the North Carolina State University Environmental and Agricultural Testing Service (Raleigh, North Carolina, USA) for DRP and DTP analysis. Dissolved reactive P was measured colorimetrically using a multichannel QuikChem 8000 (Lachat Instruments, Milwaukee, WI, USA) using the method of Prokopy and Wendt (1994). Dissolved total P was analyzed colorimetrically simultaneously on the same instrument and the method described by (Liao 2001). Only DRP was measured on the grab samples (2005–2010), while both DRP and DTP were analyzed for the daily samples (2010–2013). Organic P (difference between DTP and DRP) was not determined from 2005–2010 and was assumed to contribute to the error term in the P balance as an un-accounted loss. A subset of samples was also analyzed for total P, but no significant difference between total P and DTP was observed. This indicated that particulate P was not present at this site in measurable amounts, so DTP was used for calculating $P_{outflow}$ instead of total P.

Discharge rates were measured from December 2010 through 2013. Surface outflow in the perimeter ditch was measured at the main outlet using dual compound weirs installed in 2001 as described by Vepraskas et al. (2005) Compound weirs consist of a V-notch cut into the center of the crest of a larger rectangular notch. Discharge is calculated using two different equations depending on if the discharge is contained in the V-notch or rectangular portion of the weir. During high flow events the discharge was calculated as follows (United States Bureau of Reclamation 2001):

$$Q = 3.9h_1^{1.72} - 1.5 + 3.3Lh_2^{1.5} \quad (3)$$

where Q is the discharge in ft^3/s, h_1 is the head above the point of the V-notch in ft, L is the combined length of the horizontal portions of the weir in ft, and h_2 is the head above the horizontal crest in ft. If flow was confined to only the V-notch portion of the weir, then the standard V-notch equation was used (United States Bureau of Reclamation 2001):

$$Q = 2.49 \times h_1^{2.48} \quad (4)$$

Using the same variables identified above. The stage height was determined using two pressure transducers (HOBO U20L-04 Water Level pressure transducers, Onset Computer Corp, Bourne, MA, USA), one installed below the weir and the other above. This height was used to determine which part of the weir the water was in, and thus determine which equation to use. The pressure transducer data was recorded using a Campbell Scientific CR-10X data logger (Campbell Scientific, Logan, UT, USA).

Surface water discharge was estimated prior to December 2010 using a monthly water balance. Rainfall was measured on site at three rainfall stations (SI Table 4). Evapotranspiration rates from the MODIS Land Subsets Oak Ridge National Laboratory Distributed Active Archive Center (ORNL DAAC 2018) were used from January 2005 through December 2013. The MODIS Land Subset ET is remotely sensed ET with a 500 m resolution that is determined from leaf area index (LAI) and radiation at the earth's surface. That dataset provides total ET over 8 days. To estimate monthly ET, the 8-day ET values were divided by eight to estimate the daily ET value on the day it was reported. Missing daily ET values were then interpolated using MatLab (MathWorks, Natick, MA, USA) and summed for each month to estimate total monthly ET from January 2005 to December 2009. Evapotranspiration from January 2010 to November 2010 was estimated using the Thornthwaite method (Thornthwaite 1948) and monthly mean temperature data from the Lumberton Regional Airport (NOAA NCDC 2013) approximately 11 km northwest of Juniper Bay. The total amount of P lost in the drainage water was calculated by multiplying P concentrations in the drainage water by the volume of drainage water leaving through the outflow. This calculation was performed on a daily basis from December 2010 to 2013, and on a monthly basis for 2005 through November 2010.

The ΔP_{soil} was determined on an annual basis over the 8-year period since the wetland was restored (2005 to 2013). The error (E) term in Eq. 2, based on measured fluxes, was calculated as the remainder term between the soil ΔP_{soil} and flux ΔP.

Statistical analysis was performed on the soil total P data using PROC GLIMMIX in SAS 9.3 (SAS Institute Inc., Cary, NC, USA) with a gamma distribution. The LSMeans presented were back-transformed using an "ilink" command. Mehlich-3 extractable P data were analyzed with a natural log transformation using PROC MIXED in SAS 9.3. The LSMeans reported were back-transformed using a procedure described by Jørgensen and Pedersen (2013). Confidence intervals were corrected for multiple comparisons using a Tukey adjustment. A t-test was performed in SigmaPlot 12.5 (Systat Software, Inc., San Jose, CA, USA) to test for differences in woody biomass and biomass P content between the mineral and organic soils. Summary statistics were determined for P_{atm} using SigmaPlot 12.5.

Results

Total soil P concentrations in 2005 and 2013 are summarized in Table 1 and reported as kg P ha^{-1} basis in SI Table 3. No difference in soil TP was detected between 2005 and 2013 ($p = 0.42$) for the entire Bay to a depth of 1 m. There was also no significant difference between years at any depth for either the mineral or organic soils. Likewise, there was no significant change in Mehlich-3 extractable P between 2005 and 2013 (SI Table 4, SI Table 5). The organic soils at Juniper Bay had higher ($p < 0.0001$) concentrations of TP (0.131 kg m^{-3}, SE 0.013) than the mineral soils (0.072 kg m^{-3}, SE 0.005) across all three depths and both sampling years. The TP concentrations were highest at the surface for both mineral and organic soils in 2005 and 2013 ($p < 0.0001$). The sum of total soil P on a per-hectare basis to a depth of 1 m in 2005 was 1,094 kg P ha^{-1} for the organic soils which represent 105 ha (40% of the Bay) and 642 kg P ha^{-1} for the mineral soils which represent 186 ha (60% of the Bay). Thus, the total pool of total soil P in 2005, calculated as an area-weighted average, was 804 kg P ha^{-1}. In comparison the total pool in 2013 was 758 kg P ha^{-1} (1,008 kg P ha^{-1} and 592 kg P ha^{-1} for the organic and mineral soils, respectively), though again, the differences between total soil P concentrations were not significant.

Results from a separate study examining P fractions at Juniper Bay compared to two unfarmed reference Carolina Bays are presented in the Supplemental Information (SI Table 10). The history of agriculture and fertilization at Juniper Bay increased both inorganic and organic P relative to the reference Carolina Bays, though the largest increases were to inorganic P.

Rainfall P concentrations during the duration of the study averaged 0.11 mg DTP L^{-1} (SE 0.02). There was no significant difference in P among the three stations. The average concentration of rainfall DTP from 2005 to 2012 was assumed to be equal to the 0.11 mg DTP L^{-1} observed in this study. That concentration and daily rainfall data for Juniper Bay were used, along with the total area of the Bay, to estimate the P_{atm} following restoration in 2005 – a total of 8 kg ha^{-1} over eight years, or 1.0 kg ha^{-1} yr^{-1}. Monthly rainfall is summarized in SI Table 4. Estimated monthly P_{atm} is summarized on a per-hectare basis in Table 2.

There was no significant difference in estimated woody biomass or estimated woody biomass P between soil types (mineral versus organic). The average values across all plots were used for further calculations. The estimated average woody biomass was 29,300 kg ha^{-1} (SE 5,400), and the estimated average woody biomass P was 26.6 kg P ha^{-1} (SE 4.9). This translates

Table 1 Soil total P concentrations by soil type, depth, and year

Soil	Depth	2005						2013					
		LSMean		SE	Significance			LSMean		SE	Significance		
					a	b	c				a	b	c
	cm	kg TP m^{-3}						kg TP m^{-3}					
Mineral	15	0.122	±	0.013	A	A	A	0.132	±	0.015	A	A	A
Mineral	30	0.069	±	0.008	A	B	A	0.058	±	0.007	A	B	A
Mineral	100	0.051	±	0.007	A	B	A	0.044	±	0.006	A	B	A
Organic	15	0.257	±	0.039	A	A	B	0.207	±	0.032	A	A	B
Organic	30	0.118	±	0.018	A	B	B	0.160	±	0.025	A	B	B
Organic	100	0.076	±	0.017	A	B	A	0.065	±	0.015	A	C	A

[a]Comparison of total P concentration between years for a given soil type and depth in column

[b]Comparison of total P concentration between depths within a given year and soil type in column

[c]Comparison of total P concentration between soil types within a given year at a given depth in column

Means with the same letter not significantly different ($\alpha = 0.05$)

Table 2 Monthly and Annual atmospheric P deposition (P_{atm}, kg TP ha^{-1}) between January 2005 and December 2012

Month	2005[a]	2006	2007	2008	2009	2010	2011	2012
Jan	0.056	0.077	0.018	0.079	0.035	0.090	0.037	0.077
Feb	0.046	0.078	0.048	0.116	0.047	0.116	0.133	0.077
Mar	0.054	0.015	0.040	0.104	0.093	0.089	0.112	0.101
Apr	0.055	0.047	0.070	0.101	0.030	0.019	0.067	0.059
May	0.041	0.137	0.045	0.081	0.262	0.085	0.102	0.169
Jun	0.053	0.146	0.092	0.078	0.127	0.186	0.041	0.114
Jul	0.126	0.122	0.015	0.096	0.111	0.277	0.099	0.099
Aug	0.015	0.154	0.066	0.179	0.162	0.082	0.198	0.236
Sep	0.037	0.029	0.012	0.229	0.009	0.212	0.101	0.090
Oct	0.067	0.019	0.048	0.022	0.075	0.034	0.061	0.033
Nov	0.090	0.097	0.002	0.128	0.194	0.035	0.090	0.000
Dec	0.054	0.079	0.113	0.080	0.165	0.064	0.020	0.077
Total	**0.695**	**1.001**	**0.570**	**1.294**	**1.310**	**1.289**	**1.062**	**1.133**

[a]Rainfall data was acquired from a nearby weather station at the Lumberton, NC airport (NOAA NCDC 2013)

to 3.33 kg P ha^{-1} yr^{-1} over the eight years following restoration. The plant litter biomass and P contents are shown in Table 3. The average accumulation of P into the litter layer was 3.2 kg P ha^{-1} (SE 0.7), or 0.4 kg P ha^{-1} yr^{-1} (SE 0.1).

Discharge rates from Juniper Bay were estimated prior to December 2010 based on a simple water balance as the difference between monthly rainfall (SI Table 4) and monthly ET (SI Table 5). Rainfall was compared to normal values for the AgACIS WETS table for the Lumberton Regional Airport which had an observation period of 1971–2000 (USDA-NRCS 2021). Less than normal rainfall was observed for 2005, 2006, 2007, 2011 and 2012, while 2008–2010 had normal rainfall. Evapotranspiration rates are summarized in SI Table 5. Discharge from the single outflow at Juniper Bay was measured directly starting in December 2010. Monthly discharge rates are summarized in SI Table 6. For months where the estimated discharge was negative (ET > rainfall) the discharge was assumed to be 0 mm.

The concentration of DRP over time at the Juniper Bay outflow is shown in Fig. 2. The concentration of DRP increased following restoration in 2005 and remained at elevated concentrations until 2010. Following 2010 the concentrations declined to pre-restoration levels. The P discharge, estimated on a monthly basis for 2005–2010 and a daily basis for 2011–2012, averaged 0.2 kg P ha^{-1} yr^{-1} during the eight years following restoration. Exports of P are summarized in SI Table 9.

The P balance for Juniper Bay is summarized in Fig. 3. The main flux of P entering the Bay during this study was from the atmosphere at 1.0 kg ha^{-1} yr^{-1}. Plant uptake into woody biomass was the largest P flux out of the soil and was estimated at 3.3 kg P ha^{-1} yr^{-1}. Phosphorus that had accumulated in the forest floor litter was estimated at 0.4 kg P ha^{-1} yr^{-1}. Phosphorus leaving the site through the drainage water was estimated at 0.2 kg P ha^{-1} yr^{-1}. This leaves an error term of -2.9 kg P ha^{-1} yr^{-1}.

Table 3 Plant litter biomass and P content

Plot	Dominant Trees Present	Average Litter Depth (cm)	Litter Dry Weight	P Concentration	Litter P
		cm	g	% (w/w)	kg P ha^{-1}
1	Pine	4.50	850.6	0.08	7.1
8	Bald Cypress	2.60	337.7	0.09	3.0
9	Pine, Oak, Sweetgum	4.82	870.4	0.05	4.6
12	Bald Cypress, Pine, Willow	2.76	274.4	0.05	1.4
14	Pine, Bay	1.38	259.5	0.07	1.8
15	Bay, Bald Cypress	2.58	290.5	0.10	3.0
16	Oak	3.90	488.8	0.07	3.6
19	Bald Cypress	0.76	82.5	0.11	0.9
	Average litter P				3.2 ± 0.7

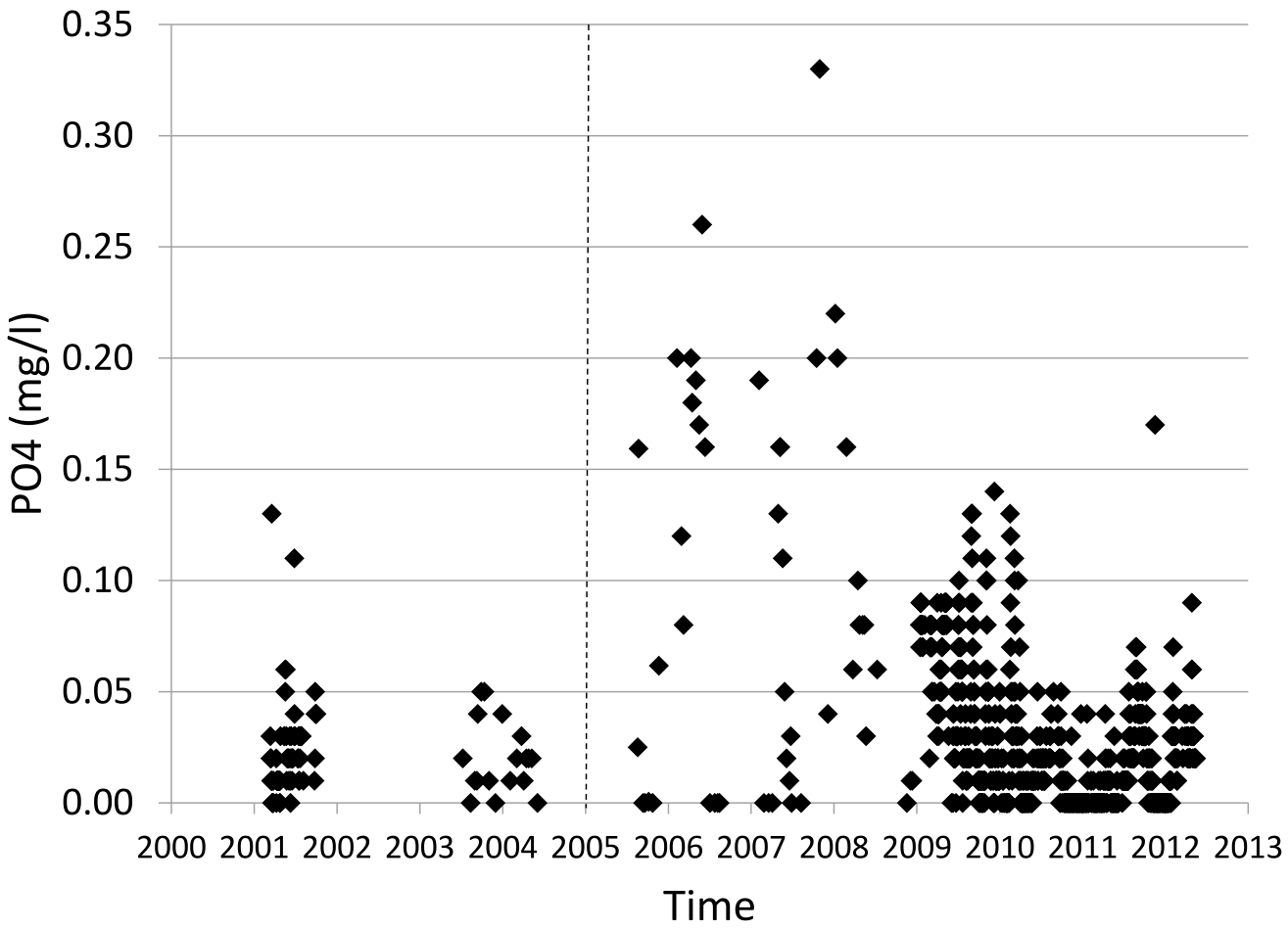

Fig. 2 Concentration of dissolved reactive P at the Juniper Bay outflow over time. The wetland was restored in 2005 (dashed line), after which an increase in dissolved reactive P at the outflow was observed through approximately 2010. The dissolved reactive P concentrations declined to pre-restoration levels thereafter. Eutrophication would be expected at 0.10 mg dissolved reactive P per liter in freshwaters, assuming N is not limiting. Outflow concentrations only exceeded that concentration once after 2010

Discussion

The primary focus of this study was to determine if Juniper Bay has been, is currently, or will be a source of P for downstream surface waters following restoration of its prior-drained agricultural land. The increase in soil P over 30 years of agricultural fertilization did lead to soil P concentrations significantly higher than un-farmed reference Carolina Bays (Ewing et al. 2012) and a total P pool of 804 kg P ha^{-1} at the time of restoration in 2005. A separate study in 2010 presented in the SI showed those increases occurred in both the mineral and organic P fractions of both the mineral and organic soils with the largest proportional increases occurring in the inorganic P. For this P balance study comparing pre- and post-restoration soil samples there was a nominal decrease in total soil P concentrations. However, those differences were not significant so we must assume that there was no change in the soil P pool following restoration.

A P balance was estimated in order to better understand P fluxes at Juniper Bay, and to guide future management. The main flux of P into the Bay was P_{atm}. Atmospheric deposition of P was small, at 1 kg ha^{-1} yr^{-1}, which is similar to the median annual deposition of TP in North America of 3.2 kg ha^{-1} reported by Tipping et al. (2014). The concentration of DTP in the rainwater was also approximately the same concentration as in the drainage water. Because the runoff ratio (ratio of runoff to total rainfall) of Juniper Bay is very small, very little of that rain (and P) reached the outflow structure. The flux of P out of Juniper Bay in the drainage water was under 0.2 kg P ha^{-1} yr^{-1} during the eight years following restoration. The largest P loss was due to plant uptake. The average annual incorporation of P into woody biomass was 3.3 kg P ha^{-1} yr^{-1} in addition to 0.4 kg P ha^{-1} yr^{-1} accumulating in the litter. This flux of P from the soil and into woody biomass and plant litter should slow any potential release of P to drainage waters.

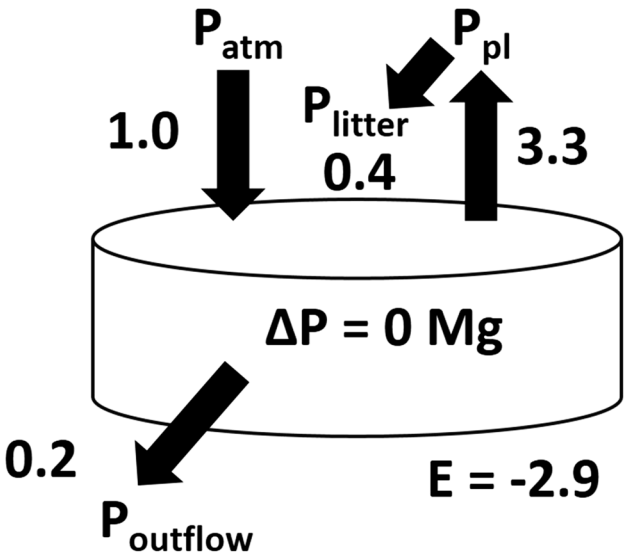

Fig. 3 Summary of the P balance for Juniper Bay from 2005–2013. The fluxes are reported as kg P ha^{-1} yr^{-1} and include change in total P (ΔP_{soil}), atmospheric deposition (P_{atm}), plant uptake (P_{pl}), plant litter (P_{litter}), surface water outflow ($P_{outflow}$), and error (E)

Our estimated error term, E (-2.9 kg P ha^{-1} yr^{-1}) was calculated assuming ΔP_{soil} is 0 kg ha^{-1} yr^{-1} due to finding no significant difference between soil total P between 2005 and 2013. The magnitude of the soil P pool, and the similarly large error in total soil P are an order of magnitude higher than the fluxes reported. This is the most likely contributing factor towards the calculated error in this P balance. A more intensive soil sampling approach may reduce those error terms in total soil P. There are other potential sources of error as well. One is having only DRP and not DTP measurements of the drainage water prior to 2010. The organic P missed may account for some P flux out of the wetland. However, the measured flux of P in the drainage water was still very small, smaller than the soil P pool by three to four orders of magnitude. We note that we did not measure P concentrations of woody tissue in this study, nor did we conduct repeated sampling of litterfall. Such changes would likely have improved our accuracy of P lost to plant uptake or litter accumulation. Further, plant uptake is likely larger than was predicted due to P taken up by small trees (< 10 cm DBH), shrubs, and herbaceous plants, which were not measured in this study.

The agreement of measured runoff volumes at the outflow with runoff predicted with a water balance suggest that losses or gains of water from the surrounding landscape are relatively minimal (see supplemental information). This is further evidenced by the small hydrologic gradients in four transects around the Bay (SI Table 1), and the resulting relatively slow porewater velocities (SI Table 2).

While post-restoration concentrations of P in the Juniper Bay drainage water did depict a small release of P out of Juniper Bay, concentrations have since declined to pre-restoration levels (≤ 0.1 mg P L^{-1}) within five years of restoration (Fig. 2). Phosphorus concentrations above 0.1 mg L^{-1} would be expected to contribute to eutrophication in freshwater systems (Correll 1998). The concentration of P in the drainage water was also equal to P concentrations found in the rainwater. In addition, the P that has left Juniper Bay through the outflow since restoration accounts for approximately 0.2% of the total pool of P at the site in 2005. Because of the low concentrations of P in the drainage water, and the low magnitude of P losses to the drainage water relative to the total P pool, P export from Juniper Bay to surface waters is not expected to be a major concern in the future. Bruland et al. (2003) determined that in a restored Carolina Bay wetland complex the export of soluble reactive P and total P from the restored wetland was less than that of an actively farmed wetland, thus concluding that wetland restoration after just two years resulted in a net improvement to water quality. Concentrations of P following restoration did eventually decrease to concentrations observed following agricultural production before wetland restoration. Plant uptake may also reduce the amount of plant available P – the P fraction most easily exported. However, P loss due to plant uptake alone will likely take decades to centuries to reduce soil P concentrations down to natural concentrations.

The results of this study indicate that while most of the residual soil P that was left over from agricultural production is still in the Juniper Bay soils, it is not moving off site. This indicates that Carolina Bays, like Juniper Bay, may make excellent potential sites for wetland restoration. However, Carolina Bays that are drained by streams may be exceptions. Such wetlands would be expected to have more P leaving the site through surface outflow because of higher hydraulic gradients caused by the dissecting streams. The ideal areas for wetland restoration are closed depressions that have precipitation as the main water source and evapotranspiration as the main water loss.

In weighing the potential risks and benefits of restoring Carolina Bay wetlands that have been used for production agriculture, it is important to consider the conversions between land uses for these wetlands within the region. In a concurrent study, separate from that presented here, Sullivan et al. (2017) inventoried Carolina Bays in Bladen County, North Carolina a representative "Bay-dense" region within the southeastern Coastal Plain close to Robeson County, North Carolina where the present study was conducted. They documented land-use change from 1972 through 2010 using decadal Landsat imagery. They found that during that time period, 43% of the Bays and 91% of the Bay area were associated with land-use change between 1972 and 2010. In 1972, Bays were predominantly forested (79% by count),

with remaining Bays converted to agriculture or urban use prior to 1972. Land-use changes were predominantly from forest to agriculture (46%) and agriculture to forest (37%). Conversion to forest remained low from 1984 to 1991 and from 2000 to 2010, with a net loss in agricultural land use of 2,085 and 1,457 ha, respectively. A surge in conversion from forest to agriculture occurred between 1991 and 2000, which was surprising given the 1990 US Army Corps of Engineers/Environmental Protection Agency agreement targeting no net wetland loss. From 1972 through 2010, there was an estimated net gain of 744 ha of Bay forest relative to Bay agriculture. Sullivan et al. (2019) conducted a follow-up study assessing the risk of P export to nearby or intersecting streams. They found that 1,360 Carolina Bays in Bladen County, North Carolina representing 43% of the Bays and 80% of the total Bay area had streams that either intersected the Bays or came within 15 m of the edge of a the Bays. These wetlands posed a risk of P export depending on the land use or changes in land use of each Bay. Isolated wetlands without a nearby or intersecting stream were determined to pose little risk of P export.

Summary & Conclusions

The objective of this study was to create a P balance for a Carolina Bay restored from agricultural land. Juniper Bay was restored by filling in primary ditches, plugging secondary ditches, and maintaining the perimeter ditch surrounding the site. This resulted in a low hydraulic gradient for drainage water. The soil P pool to a depth of 1 m was determined to be 804 kg P ha^{-1} in 2005 and 742 kg P ha^{-1} in 2013, though this difference was not significant. Phosphorus fluxes into and out of Juniper Bay included a gain in P from atmospheric deposition, and losses of P to surface water outflow and plant uptake. The error term depicts an unaccounted loss of 2.9 kg ha^{-1} yr^{-1}. This is miniscule compared to the size of the P pool existing in the soil at Juniper bay and is smaller than all standard errors reported for total soil P (SI Table 3). Phosphorus loss to surface waters was minimal both in magnitude (0.2 kg P ha^{-1} yr^{-1}) and in current concentrations (approximately 0.1 mg P/L). The concentration of P exiting the Bay is approximately the same as was observed in rainwater at Juniper Bay, and is not expected to contribute to eutrophication of downstream surface waters.

The key takeaway from this study is Carolina Bay wetlands similar to Juniper Bay are promising sites for wetland restorations due to the low risk of P contributions following restoration from agricultural production. Carolina Bays are broad and nearly level, which results in very low hydraulic gradients following restoration back to wetlands. This likely reduces movement of P to nearby streams or drainageways. However, internal drainage ditches must be filled and/or plugged during the restoration process. Carolina Bays that are deeply dissected by streams or ditches may have sufficient hydraulic gradients to facility P transport, so restoration of such sites must be done with care.

Supplementary Information The online version contains supplementary material available at https://doi.org/10.1007/s13157-023-01725-z.

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Authors' Contributions CJM lead efforts for sample collection, sample analysis, data analysis, data visualization, and writing. MJV was the principle investigator for both sources of funding, supervised the research activities of CJM, and was a major contributor to writing the manuscript. JGW assisted with sample collection for the archived soil samples. DDR assisted soil total P analysis. All authors provided revisions to manuscript drafts and read and approved the final draft.

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Data Availability The datasets generated during and/or analyzed during the current study are available from the corresponding author on reasonable request.

Code Availability The code used during the current study is available via Moorberg (2014).

Declarations

Ethics Approval Not applicable.

Consent to Participate Not applicable.

Consent for Publication Not applicable.

Conflicts of Interest The authors have no conflicts of interest.

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