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Modeling Phosphorus Retention and Release in Riparian Wetlands Restored on Historically Farmed Land

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Nature-based solutions are of interest in efforts to achieve reductions in phosphorus (P) loads to aquatic ecosystems. One potential solution of this kind is the restoration of riparian wetlands. Many candidate sites for riparian wetland restoration were formerly used for agriculture and therefore may contain legacy soil P from past fertilizer and/or manure applications. Here, we combined 2-year field studies of 3 restored riparian wetlands on formerly farmed land in the Lake Champlain Basin, Vermont, United States, with implementation of a novel *wetlandP* model to estimate net P retention. In the field, we measured variable inorganic P deposition ranging up to approximately 1 g P m^{-2} yr^{-1} and collected data on P stocks and fluxes required for modeling. At 2 sites, observed water quality dynamics during flood events were indicative of internal dissolved inorganic P (DIP) release from soils during low oxygen conditions. We calibrated and verified the *wetlandP* model using field data and used it to examine numerous scenarios. Our simulations indicated variable net total P (TP) retention, driven by a trade-off between particulate P trapping and DIP release, with most plausible scenarios (95 out of 108) indicating that the study wetlands serve as net TP sinks. Our net TP retention estimates (range = -0.06 to 0.45 g P m^{-2} yr^{-1}, mean P retention efficiency = 35%) are comparable to prior literature and help clarify key drivers. Our modeling results also show that release of legacy soil P as DIP can be sizable in some cases (range in net DIP retention = -0.11 to 0.02 g P m^{-2} yr^{-1}), especially for wetlands receiving river/stream water with low DIP concentration. We present a conceptual framework to help guide prioritization of riparian wetland restoration by ecological engineers and designers when water quality improvement via P retention is a goal.

Keywords Phosphorus; Wetland; Riparian; Floodplain; Restoration; Farmland

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Study/project photograph One of the study sites, showing a restored riparian wetland ecosystem on formerly farmed land. (Photograph by Adrian Wiegman.)

1. Introduction

In the United States and elsewhere, federal, state, and non-profit organizations are interested in pursuing the restoration of riparian and floodplain wetlands to help meet a broad range of conservation goals (Meli et al. 2014; Thorslund et al. 2017; Matsuzaki et al. 2019). Historically, many riparian wetlands were drained and converted to agricultural use because of their rich deep soils and proximity to water. This, along with other modifications such as drainage ditches, dam building, and levee construction, has led to extensive riparian wetland loss across much of the developed world (Hook 1993). In more recent decades, there have been efforts to reverse this trajectory and restore wetlands on land previously converted and farmed. This includes programs such as the Wetland Reserve Program in the United States, which has restored nearly 1.2 million ha (2.9 million acres) of wetlands as of 2023 (USDA NRCS 2023).

One conservation goal that can motivate riparian wetland restoration is the improvement of water quality through retention of phosphorus (P) in wetlands and consequent decreased loading to downstream aquatic ecosystems (Land et al. 2016; Singh et al. 2019). The Vermont portion of the Lake Champlain Basin (VT-LCB), located in the Northeastern United States, provides a characteristic example of this. In the VT-LCB a 34% reduction in P loading to Lake Champlain is required by a US EPA total maximum daily load (TMDL) (US EPA 2016). Restoration of wetlands in the VT-LCB has attracted interest from several stakeholders, and >4,000 potential wetland restoration sites in Vermont have been ranked using a GIS-based prioritization model to inform future efforts (VT DEC 2018; Arrowwood Environmental 2017). In the VT-LCB, more than 90% of the >4,000 potential wetland restoration sites were farmed as recently as 2016 (Wiegman et al. 2022). How such former agricultural land use affects P retention in restored riparian wetlands remains unclear.

Generalizations about P retention in wetlands are difficult to make, especially for restored riparian wetlands (Fisher and Acreman 2004; Hoffman et al. 2009; Land et al. 2016; Walton et al. 2020; Ury et al. 2023). Measuring or modeling P retention in riparian wetlands is complicated by unconstrained

Highlight

The restored riparian wetlands studied are generally net total phosphorus sinks on the landscape, but can be sizable sources of dissolved phosphorus in some cases.

hydrology and spatiotemporal variability in water and soil that affect P dynamics (Underwood et al. 2017; Wiegman et al. 2022). A recent meta-analysis of P retention in restored and constructed wetlands excluded riparian and floodplain wetlands because inflow and outflow are not easily defined (Ury et al. 2023). Some studies of restored riparian wetlands report net total P (TP) retention driven by particulate P deposition (Kronvang et al. 2009; Noe et al. 2019), while others have found potential for P export (Hoffmann et al. 2009; Jones et al. 2015) or decreasing TP retention over time (Mitsch et al. 2012).

Phosphorus retention in wetlands occurs due to settling and trapping of P associated with particles, accretion of P-containing organic matter, and chemical reactions with iron (Fe), aluminum (Al), calcium (Ca), and magnesium (Mg) (Reddy et al. 1999). Wetlands can sometimes release dissolved inorganic P (DIP) downstream due to solubilization of legacy P in soils, exhaustion of soil P sorption capacity, and the decomposition of organic matter (Reddy et al. 1999; Jones et al. 2015; Wiegman et al. 2022; Kizuka et al. 2023). DIP encompasses various orthophosphates and polyphosphates, which represent the most mobile, reactive, and biologically available forms of P in the environment (Ruttenberg 2014) (see Text S1 for a note on terminology).

Soils, vegetation, and prior land use can be key drivers of DIP dynamics in riparian wetlands (Kiedrzyńska et al. 2008; VanZomeren et al. 2020; Wiegman et al. 2022). Phosphorus saturation ratio (PSR) and soil P storage capacity (SPSC) are 2 related soil metrics that indicate the tendency of a soil to store or release DIP. Both metrics are derived from P, Al, and Fe extracted by an acid (e.g., ammonium oxalate or Mehlich-3 solution) (Nair et al. 2015). PSR is the molar ratio of P to the sum of Al and Fe, while SPSC indicates the mass of P (mg P kg^{-1}) that a soil is likely to store or release based on the difference between a soil's PSR and a locally estimated threshold PSR (Nair et al. 2015). Negative SPSC values indicate potential for P loss, while positive SPSC values indicate potential for P gain (Nair et al. 2015). Wiegman et al. (2022) found that SPSC was a particularly strong predictor of potential soil DIP release in riparian soils within the VT-LCB.

In riparian wetlands, P retention is also linked to local hydrology, which influences P loading rate, sedimentation, contact between water and soil/sediments, P uptake by vegetation, and decomposition (Hoffmann et al. 2009). The complicated hydrology of riparian wetlands (e.g., absence of well-defined inflows and outflows, networks of barriers/berms/roads and crevasses/ditches/culverts, variable depth/velocity across the floodplain) commonly precludes the ability to measure influent and effluent P loads in the field (Ury et al. 2023). In such cases, a combination of field measurements and modeling is required to estimate net P retention. Therefore, the overarching aim of this study was to provide a systematic assessment of P dynamics, including particulate P deposition, internal P cycling, and DIP release, in selected restored riparian wetlands on former agricultural land within the VT-LCB using a combination of field studies and modeling. Our specific objectives were to:

1. Design and parameterize a process-based model (*wetlandP*) to simulate hydrology and P dynamics in restored riparian wetlands.
2. Quantify P storages and P dynamics in the field for 15 sampling plots across 3 restored riparian wetlands, including multiple years and flood inundation events.
3. Run and assess model simulations, accounting for both particulate P and DIP, for selected restored riparian wetlands under various scenarios to clarify key drivers of net TP retention.

Based on previous observations in the literature (Land et al. 2016; Walton et al. 2020; Jones et al. 2015; Hoffmann et al. 2009) and our concurrent work in the study area (Wiegman et al. 2022), we hypothesized that the riparian wetlands studied would be characterized by a trade-off between particulate P capture and DIP export, with net TP retention dependent on site factors including soil P status.

2. Materials and Methods

Our methods are described briefly in the following sections. Additional details needed to replicate our procedures are given in the Supplementary Material (see Text S1 for modeling, Text S2 for the field study).

2.1 Model Development

Our modeling objective was to simulate time series of different P stocks at the plot-scale (i.e., establish the plot as the control volume) and then use the model to simulate a range of scenarios involving wetland-scale changes (i.e., external forcings) to estimate the magnitude of perturbation necessary to influence P retention and exports at the plot scale. We coded and parameterized a model, *wetlandP*, in R (R Core Team 2021) using the 'deSolve' package (Soetaert et al. 2010) to simulate the hydrology and P dynamics of selected restored riparian wetlands. The *wetlandP* model is derived from existing process-based models (Wang and Mitsch 2000; Hantush et al. 2013; Marois and Mitsch 2016; Wiegman et al. 2018) but is novel in its incorporation of soil P metrics commonly used in legacy P assessment (e.g., soil P saturation ratio [Nair et al. 2015]), representation of hydrology and dissolved P fluxes, and use of a relatively small number of local parameters for soil, water quality, and hydroclimate (see Text S1).

The *wetlandP* model simulates P fluxes and transformations in an aboveground surface water layer, as well as in a single active soil layer (Fig. 1). Changes in soil particulate inorganic P can occur due to sedimentation during flooding and DIP sorption/desorption (Reddy et al. 1999). Soil organic P changes occur due to sedimentation as well as mineralization, which is the conversion of organic P to inorganic P during the decomposition of organic matter (Reddy et al. 1999). Change in plant shoot and root P represents the balance of plant growth (and associated P assimilation) and mortality (i.e., transfer of shoot P to litter), and was modeled as a function of air temperature, plant species, and nutrient availability (Wang and Mitsch 2000). Change in DIP is a function of import and export in surface water, mineralization, sorption/desorption, plant uptake, and diffusive flux between the soil porewater and surface water (Hantush et al. 2013; Kalin et al. 2013).

Water inflows and outflows are forced by an input table containing a daily resolution time series of hydroclimatic variables: water height (H_w = water level above the soil surface), wetted area (A_w), intercepted precipitation, and evapotranspiration (see Box S1). A preprocessing subroutine takes the input table containing the forcing variables and calculates water volume (V_w) change and net lateral flow (Q_{net}). Subsequently, surface inflow and outflow are calculated from V_w, Q_{net}, and hydraulic residence time (HRT). HRT can either be set as a constant or modeled dynamically as a function of water height.

During model simulations, the model calculates P inflows (P_{in}) to a pool (state variable) for a given time step (t) as the length of the time step (dt = 1 day) multiplied by the inflow rate (Q_{in}) and the inflow concentration for a given P pool. Similarly, the model calculates P outflows (P_{out}) from a pool as dt multiplied by the compartment (aboveground or belowground) outflow rate (Q_{out}) and the concentration of P in the compartment for a given P pool. After each simulation, a postprocessing subroutine calculates net mass balances of TP and DIP for each pool from a time series of model outputs (Equation 1):

$$\Delta P = \sum_{t=1}^{n} (P_{\mathrm{in},t} - P_{\mathrm{out},t}) \qquad (1)$$

where ΔP is the net P balance (i.e., retention) of a given P pool (e.g., DIP) calculated as the difference between P inflows and P outflows, summed for all *n* time steps. Total P is calculated as the sum of DIP, LOP, ROP, and PIP (see definitions in Fig. 1). Net P balances are divided by the simulation length (in years) to produce average annual P retention estimates (g P m^{-2} yr^{-1}).

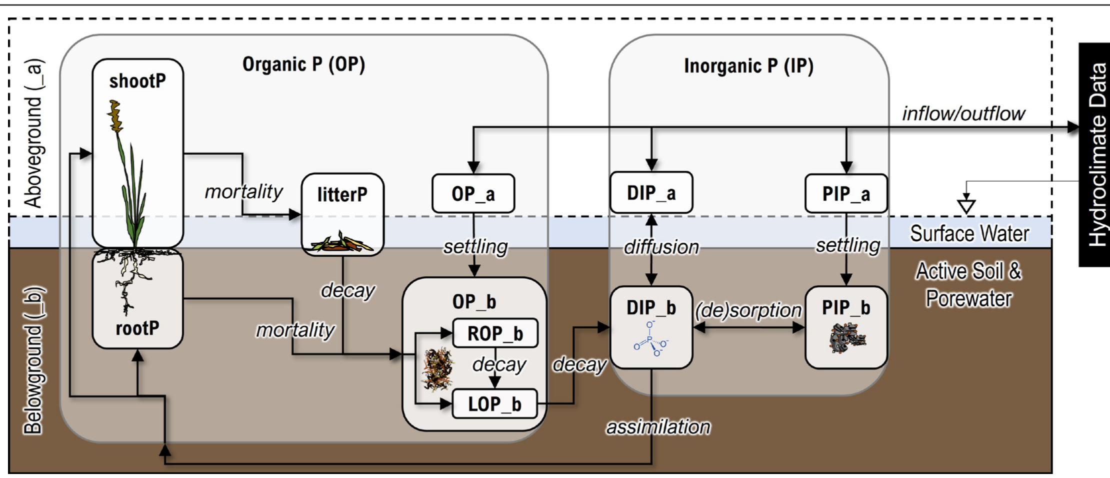

Fig. 1 Conceptual diagram of *wetlandP* model domain, compartments, state variables, and processes. Flows of phosphorus (P) are represented by lines with arrows and the associated process for each flow is labeled in italics. State variables are represented in boxes with bold text. ROP = Refractory OP, LOP = Labile OP, DIP = Dissolved IP, PIP = Particulate IP.

2.2 Field Study

Additional details on P stocks, P fluxes, and water measurements in the field study are given in Supplementary Material Text S2.

2.2.1 *Study Areas*

We studied 3 historically farmed wetlands in the Vermont portion of the Lake Champlain Basin (VT-LCB) (Fig. 2a – Fig. 2e). The study sites occupy 2 types of riparian

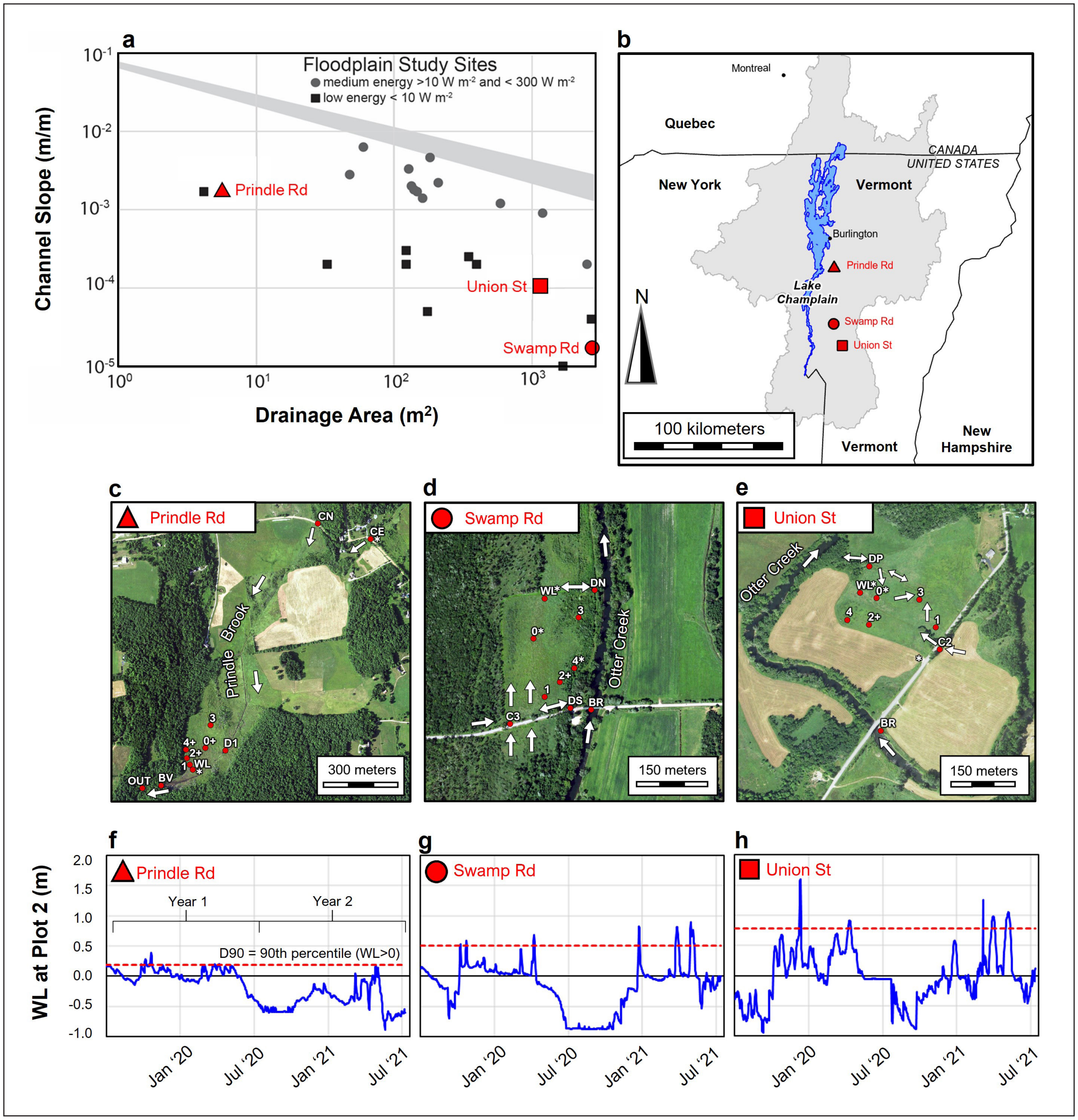

Fig. 2 Summary of study sites, showing: (a) Comparison to other Vermont floodplain monitoring sites (data from Diehl et al. 2022, red shapes represent the 3 sites from this study, black squares represent low-energy sites, and grey circles represent medium-energy sites); (b) Location within the Lake Champlain Basin and Vermont (source: U.S. Geological Survey, National Hydrography Dataset); (c – e) 2016 areal imagery with red dots showing sampling plot locations and white arrows showing water flow paths (source: Farm Service Agency, National Areal Imagery Program, 2016) ; (f – h) hydrographs of water level (WL, meters, blue line) relative to the soil surface at plot 2 (median site elevation) from HOBO water level loggers, dashed red line shows the 90% percentile depth for periods when WL >0.

settings (headwaters, large river) that are typical in the region for potential wetland restoration candidates with a history of farming and span the ranges for channel slope and drainage area of low-energy floodplain study sites in the VT-LCB (Fig. 2a). Each study site was farmed as hay or corn/hay rotation for decades before farming ceased between 2004 and 2006, with native trees planted in the higher elevation zones. Hydrological restoration actions varied between the sites (see USDA NRCS 2011, 2021 for wetland restoration guidelines).

The "Prindle Road" site is a headwater depression wetland that receives inflow from 2 first order streams conveyed through culverts (Fig. 2c) from an upstream area (2.5 km^2) containing a mix of pasture, natural vegetation, and low-density residential development. The historical stream channel was straightened and incised due to agricultural activity. Beavers have created a dam at the outlet, raising peak water levels by up to ~2 m above the thalweg of the incised stream.

The 2 other sites are adjacent to Otter Creek about 25 km apart along an undammed, well-connected floodplain complex that was intensively ditched, logged, and farmed during the 20th century. The "Union Street" site is located on the flashier upstream end of the floodplain complex. In 2018, an ~1 m tall earthen ditch plug armored with rip rap (USDA NRCS 2011, 2021) and an artificial berm meander scar were constructed at the Union Street site. The "Swamp Road" site is located on the downstream end of the floodplain and no ditch plugs or beaver dams were present during the time of study. Nearly every year floodwaters inundate the Swamp Road wetlands for weeks at a time.

Within each site we delineated a sampling zone in an area that had uniform prior land use and perennial emergent vegetation based on available areal imagery in Google Earth Pro (ranging from 1992 to 2018). We distributed 5 circular plots (5 m radius) along an elevation gradient at each site for destructive sampling. These sampling plots were placed randomly within equal area elevation percentile bins based on 0.7 m resolution LiDAR elevation data (VCGI 2018) and were labeled from zero to 4, in order of lowest to highest elevation (Fig. 2c – Fig. 2e). We also performed additional water sampling at the river upstream of each site, at ditches, and at other likely water flow paths (Fig. 2c – Fig. 2e).

2.2.2 Flood Monitoring

We deployed high-resolution (15-min) sensors at each site to monitor surface water levels and dissolved oxygen concentrations above the soil-water interface. Our water sampling protocol consisted of a combination of the following approaches: (1) passive siphon sampling triggered during the first flush of water at each sampling plot, (2) auto-sampling at the median elevation plot over the first 24+ hours after flooding, and (3) grab sampling in the river and at wetland sampling plots during the rising and falling limb of the floods. Field measurements of water temperature and dissolved oxygen (DO, mg L^{-1}) were recorded each time a grab sample was taken using a Professional Plus Multiparameter Instrument (YSI, Inc.). Water samples were analyzed for total suspended solids (TSS), mineral (i.e., inorganic) suspended solids (ISS), organic suspended solids (OSS) (Roy et al. 2016), dissolved inorganic P (DIP) (D'Angelo et al. 2001; Ringuet et al. 2011), and total P (TP) (Patton and Kryskalla 2003; Murphy and Riley 1962).

2.2.3 Biomass, Litter, and Accretion

We quantified stocks of dry matter and P in aboveground and belowground biomass of herbaceous plants, woody biomass, and litterfall, as well as litter mass decay and litter net P mineralization (see Text S2). We estimated litter mass decay and net P mineralization rates using a one-year in situ litterbag decomposition experiment with litter samples collected in mid-October 2019. We estimated accretion rates in the riparian wetlands using ceramic tiles (30.5 cm x 30.5 cm) (McMillan and Noe 2017; Callaway et al. 2013), 3 per plot, incubated over a wet season (October 2019 – July 2020). We processed and analyzed herbaceous biomass, litterfall, and litter bag samples for LOI and HCl-TP using methods described below for soil. Samples of accreted material were analyzed for LOI, total P, inorganic P, and organic P, following the 3-pool parallel P fractionation method described below for soils (Richardson and Reddy 2013).

2.2.4 Soil Analyses

In July 2019, duplicate soil samples at 0 cm – 5 cm and 5 cm – 10 cm depths (excluding surface litter) were collected from each sampling plot using 7-cm diameter polycarbonate coring tubes, then sealed in polyethylene bags and stored at 4 °C until processing at the lab. At the lab, subsamples were kept moist at 4 °C, air dried (25 °C), or oven dried at 60 °C, then stored for different protocols. Dried subsamples were sieved (<2 mm) and homogenized, and a subsample of sieved soil was ground with a mortar and pestle. Dry soil subsamples were stored in watertight containers, in the dark, until analysis.

Within one week of sample collection, we initiated a sequential P fractionation (SF) on moist soils that separates 5 operational fractions of soil inorganic P (P_i) and organic P (P_o) (Fig. S12) (Reddy et al. 1998; Richardson and Reddy 2013; Roy et al. 2017). We summed all 5 fractions to estimate total P (SF-TP), all 3 P_i fractions to estimate inorganic P (SF-P_i), and 0.1 M NaOH-P_o + residual

P to estimate organic P (SF-P_o). With oven-dry ground soils, we conducted a 3-pool parallel fractionation using 1 M HCl extractions for ashed and non-ashed soils to estimate organic content via loss on ignition (LOI, at 550 °C for 4 hours), along with total P (HCl-TP), inorganic P (1 M HCl-P_i), and organic P (1 M HCl-P_o) (Levy and Schlesinger 1999; Richardson and Reddy 2013; Wiegman et al. 2022).

We determined oxalate-extractable Al, Fe, and P (Ox-Al, Ox-Fe, Ox-P, respectively) by extracting air-dried ground soils with acid ammonium oxalate for 3 hours in the dark (Courchesne and Turmel 2008). We calculated the PSR and SPSC with data from the oxalate extraction. We used a PSR threshold of 0.23 determined previously for Vermont riparian soils to calculate SPSC (Wiegman et al. 2022).

2.2.5 Data Analysis

Stocks of P in soil, biomass, litter, and accreted material were calculated by multiplying the mass stock by P content of the material for a given P pool. We calculated summary statistics (mean, standard deviation) at the plot level by averaging plot replicates, and at the site level by averaging means for all 5 wetland sampling plots. Because the plots were distributed randomly along an elevation gradient, the site level averages represent spatially weighted averages of the sampling zone. We investigated monotonic relationships amongst soil, biomass, and litter attributes using Spearman rho correlations of plot level mean values (Kassambara 2023).

For water, P stocks were calculated as the product of P concentration (e.g., mg P L^{-1}) and water volume. We classified the water quality samples based on 3 categories: the site, the sample origin (inflow/outflow, river/stream, or wetland), and the flood phase: filling (rising water level) or draining (falling water level). We determined differences among water quality categories using the Kruskal-Wallis test with post hoc Dunn-Bonferroni multiple comparisons (Dinno 2022). We also fit simple linear regressions amongst water quality parameters. We used $\alpha = 0.05$ in all statistical tests. All calculations and statistics were performed using R (R Core Team 2021).

2.3 Model Calibration and Scenarios

We simulated TP dynamics at sampling plots 0, 2, and 4 of each site for the 2-year study period. In general, year 1 data were used for model calibration, while year 2 data were used to verify model performance. We did not attempt to optimize model performance for any single metric at an individual study plot because doing so can generate biases due to overfitting. Rather, we qualitatively assessed the model performance across all 3 sites (9 study plots) simultaneously, using the following criteria to guide calibration and verification:

- Biomass, inorganic sediment deposition, and litter P were all clustered near the 1:1 line for modeled versus measured results, and modeled results had approximately the same mean and variance as observations (when data were aggregated across all sites and plots). Accretion P was excluded from verification (year 2 data unavailable).
- Surface water DIP and total P stocks (g P m^{-2}) were within the same order of magnitude observed in the field during floods, and modeled results had approximately the same mean and variance as aggregated observations.
- Stocks of soil organic matter were relatively stable (not increasing or decreasing by more than 1% – 2% per year).

Following calibration and verification of the model, we performed a sensitivity analysis to elucidate hydrologic versus biogeochemical controls on net TP retention across the riparian wetlands (Table 1). We ran scenarios over the 2-year monitoring period using our calibrated model. Additional details on data preprocessing, model calibration, verification, and sensitivity analysis are given in the Supplementary Material (see Text S1).

3. Results

3.1 Field Study

We provide extensive documentation of the field study results in the Supplementary Material (Text S3). Here, we focus on results essential to *wetlandP* model development and simulations: (1) water level and water quality observations during the study period, and (2) selected measurements for key P stocks and fluxes.

3.1.1 Water Level and Water Quality

We monitored 3 flood inundation events at each riparian wetland site during the monitoring period, with events varying in terms of season, driver (rain, snowmelt, or both), peak water level, water temperature, and DO dynamics (Table S2). At the 2 Otter Creek sites, wetland water during the draining phase tended to have lower TSS concentration, lower DO, and similar or greater TP and DIP concentrations compared to river water during filling (Fig. S13 and Sig. S14). In contrast, at Prindle Road, river and wetland water were both characterized by relatively low TSS (typically ≤5 mg L^{-1}) and similar DO, TP, and DIP (Fig. S13 and Fig. S14).

Observed relationships between TSS, TP, DIP, and DO indicate similarities and differences across systems. In the river samples of Otter Creek, TP was strongly positively correlated with TSS, and TP was not significantly correlated with DIP (Fig. 3a and Fig. 3b). On the contrary, TP in inflow samples at Prindle Road was

Table 1 Scenarios used in the *wetlandP* model sensitivity analysis

Parameter	Scenarios	Justification
Hydraulic residence time (HRT)	Dynamic HRT based on power model (1), constant HRT of 10 days (2) and 100 days (3) to reflect potential changes in discharge through the wetlands	Our field data indicated that the wetlands were subject to a wide range of HRT.[a] Varying HRT between 10 and 100 days enabled comparison of sensitivity to HRT across sampling sites. Only power HRT simulations were used for system level P balances reported here.
Particle settling	100% particle retention (1), Stokes' Law (2)[b]	Riparian wetlands and floodplains have a wide range of hydrologic conditions that affect sediment deposition. The power model for HRT and flooding depth dictates that flow rates (and particulate P loads) are greatest when water levels are highest. When Stokes' Law and the power model for HRT are applied together, some sediment may bypass the system before being deposited, especially at the highest flooding depths. Both sets of assumptions were used to estimate system level P balances.
Influent water quality	Median values for field observed inflow concentrations of TSS, TP, and DIP held constant (1), doubled (2), and halved (3)	These scenarios enable investigation of the effects of reduction or improvement in upstream water quality. Siphon data were used for calibration and verification. Only stream (i.e., river/inflow) data were used to estimate system level P balances.
Wetland water level	Observed wetland water levels held constant (1) and increased by a factor of 1.2x (2)	Informed by estimated 20% increase in average flows during the next several decades due to wetter climate in the Lake Champlain Basin.[c] In addition, the 2-year study period was relatively dry, resulting in smaller than average observed floods, thus higher water levels may be more reflective of typical years.[d]

Notes:
[a] See Supplementary Material Text S1.
[b] Marois and Mitsch (2016)
[c] Guilbert et al. (2014)
[d] See Discussion section 4.1 and Supplementary Material Text S4.

more closely correlated with DIP than with TSS (Fig. 3a and Fig. 3b). At all sites, TSS was a poor predictor of TP in wetland water samples (Fig. 3a). At Otter Creek, TP concentrations in wetlands were strongly positively correlated with DIP (Fig. 3b), and DIP commonly exceeded 0.05 mg P L^{-1}. Conversely, DIP in samples at wetland plots of Prindle Road rarely rose above 0.05 mg P L^{-1}, and TP was poorly correlated with DIP (Fig. 3b). There was a significant negative exponential relationship between DIP and DO across wetland, river, and inflow samples at each site (Fig. 3c). The correlation of DIP with DO in wetland surface water was strongest at Swamp Road—the site most prone to DO depletion (Fig. S15)—and weakest at Prindle Road—the site least prone to DO depletion (Fig. 3c).

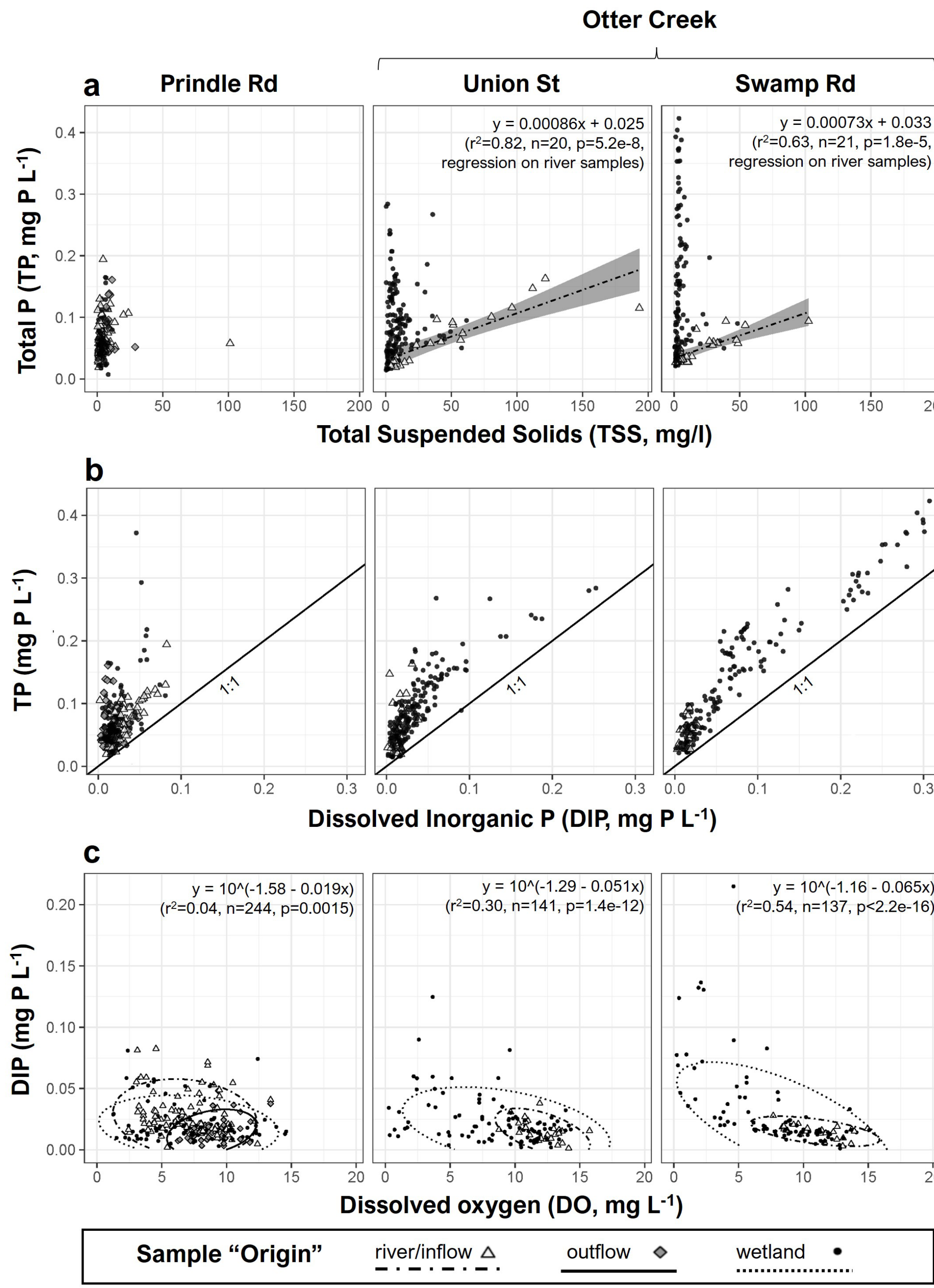

Fig. 3 Scatter plots of selected surface water properties: (a) TP vs. TSS, (b) TP vs. DIP, (c) DIP vs. DO by site and origin; Prindle Road (left), Union Street (center), Swamp Road (right). Panel "a" shows a linear regression line with confidence interval (grey shaded area) on river samples for Otter Creek sites. Panel "b" shows the 1:1 line of TP and DIP. Panel "c" shows ellipses for sample origin shown along with statistics for linear regression on DIP vs. DO for all data from each site.

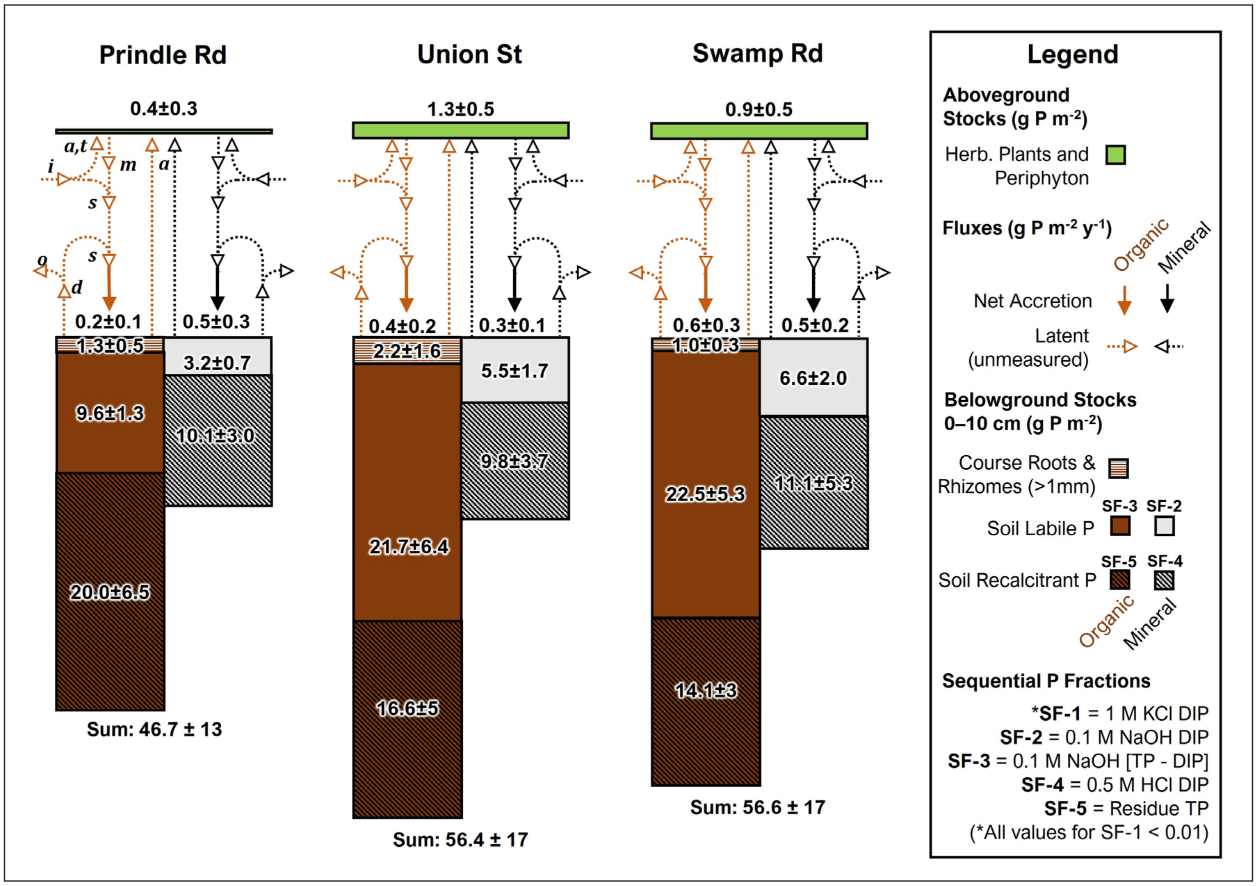

Fig. 4 Site-wide average (mean ± 1 standard deviation) of P forms and amounts (g P m^{-2} for stocks, g P m^{-2} yr^{-1} for accretion). Rectangles represent measured P stocks (height is proportional to size of P stock). Latent (unmeasured) P fluxes are shown as dashed lines and unfilled arrows: i = inflow, o = outflow, m = mortality/litterfall, d = desorption/diffusion/bioturbation, a = assimilation, t = trapping/filtering, s = settling/precipitation/adsorption. The solid downward arrows indicate the net accretion P flux to the soil surface. Black and grey lines/boxes represent mineral matter. Brown lines/boxes represent organic matter. Green boxes represent herbaceous plants and periphyton.

3.1.2 *Wetland P Stocks and Fluxes*

We summarize field measurements of various P stocks and fluxes for all 3 sites in Fig. 4 and provide detailed results in the Supplementary Material (Text S3).

Total P accretion ranged from 0.22 g P m^{-2} to 1.7 g P m^{-2} and was positively correlated with flooding depth (Table S3 and Table S4, Spearman rho = 0.74, p = 0.0027). Both inorganic and organic P accretion had similar magnitudes, 0.08 g P m^{-2} – 1.0 g P m^{-2} and 0.15 g P m^{-2} – 0.9 g P m^{-2}, respectively, and were both positively correlated with flooding depth (respectively, rho = 0.68 and = 0.61, p <0.01). Inorganic P accretion was highly correlated with, and of similar magnitude to, total P accretion minus macrophyte litter P (Table S3 and Table S4, rho = 0.93, p <1e-6). We consider inorganic P accretion to be our most reliable measurement to estimate P inputs from deposition of riverine/stream sediment during flood inundation events, although biological turnover contributed to some of the accreted inorganic P observed.

The combined stocks of organic and inorganic P in surface soil (0 cm – 10 cm) were an order of magnitude greater on average at each study site than all other P stocks together (i.e., belowground biomass, herbaceous vegetation, litter, and accreted sediment) (Fig. 4, Table S3). Soil organic P was the dominant form of P overall, comprising >60% of all P stocks (excluding woody biomass) across all 3 sites. At both Otter Creek sites, soil organic P was mostly labile (i.e., extracted by 0.1 M NaOH in the sequential P fractionation), while at Prindle Road soil organic P was mostly recalcitrant (Fig. 4). Similarly, a greater proportion of soil inorganic P was in labile forms (i.e., extracted by 1 M KCl or 0.1 M NaOH in the sequential P fractionation) at the Otter Creek sites compared to Prindle Road, with similar amounts of recalcitrant soil inorganic P across all 3 sites. Compared to both Otter Creek sites, Prindle Road soils had greater content of clay, aluminum, calcium, and recalcitrant P (SF-5) (Table S3). SPSC (0 cm – 10 cm) was lowest at Swamp Road (-37 mg P kg^{-1} to 273 mg P kg^{-1}) compared to Prindle Road (129 mg P kg^{-1} to 362 mg P kg^{-1}) and Union Street (286 mg P kg^{-1} to 497 mg P kg^{-1}) and was negatively correlated to flooding depth (Table S3 and Table S4, Spearman rho = -0.58, p = 0.026). Across all 3 sites, intact core experiments indicated ranges in the plot-average 7-day DIP release rate from soil to overlying water of 0.002 g P m^{-2} d^{-1} to 0.015 g P m^{-2} d^{-1} and 0.004 g P m^{-2} d^{-1} to 0.031 g P m^{-2} d^{-1} for aerobic and anaerobic conditions, respectively, with average release rates being highest at Swamp Road and lowest at Union Street (Table S5).

3.2 Modeling Results

3.2.1 Model Calibration and Verification

Using field results described in Section 3.1 to parameterize and force the *wetlandP* model, all calibration and verification criteria described in Section 2.3 were met (Text S1).

3.2.2 Model Scenarios

Here we summarize model results across scenarios that represent a plausible range of conditions for the study wetlands. This includes all simulations with power model HRT and river/stream derived inflow concentrations (n = 108 total, see Table 1 and Table S6 for a full description of scenarios). For these simulations, net TP retention estimates ranged from -0.06 g P m^{-2} yr^{-1} to 0.45 g P m^{-2} yr^{-1}, with a mean (± 1 standard deviation) of 0.09±0.10 g P m^{-2} yr^{-1} (Fig. 5). Mean TP retention efficiency was 35±30% and varied depending on site, inflow concentration, and assumptions for particle settling (Fig. 6). In all cases, particulate P import was greater than export. However, the balance between DIP imports and exports tended to be negative (range = -0.11 g P m^{-2} yr^{-1} to 0.02 g P m^{-2} yr^{-1}, DIP retention efficiency = -43±69%) and varied depending on site and inflow concentration (Fig. 7). Relatively few simulations (13 of 108) resulted in net P export despite negative DIP retention on average, suggesting that the wetlands investigated in this study generally function as P sinks on the landscape (Table S5).

Wetland net TP balance and retention efficiency had differing degrees of sensitivity to river/stream concentrations, HRT, water level, and assumptions about particle trapping. Mean TP retention efficiency was ~30% greater on average when assuming 100% particle trapping (51±26%, mean ± 1 standard deviation) vs. Stokes' Law (20±25%, mean ± 1 standard deviation) (Fig. 6). Greater influent concentrations of TSS and TP led to enhanced net TP retention (Fig. 5c). For example, net TP balances ranged from -0.06 g P m^{-2} yr^{-1} to +0.035 g P m^{-2} yr^{-1} and +0.035 g P m^{-2} yr^{-1} to +0.24 g P m^{-2} yr^{-1} for 0.5x and 2x river/stream concentrations, respectively, across simulations that applied 1x observed water levels and assumed Stokes' Law for particle settling. TP and DIP retention efficiencies were positively impacted by increases in inflow concentrations (Fig. 6 and Fig. 7). A shorter fixed 10-day HRT—which increased discharge through the wetland—led to greater net TP retention compared to fixed 100-day HRT for all site-influent combinations (Fig. S17). While changing water levels and HRT also had a noticeable effect on TP and DIP balances for a given site, the model was more sensitive to plausible changes in influent water quality than it was to water level or HRT. For example, a base 2 order of magnitude increase in concentration (0.5x to 1x, or 1x to 2x) had a much greater effect on net TP retention than a base 10 increase in HRT (10 days to 100 days) or a 20% (1.2x) increase in water levels (Fig. S17).

4. Discussion

4.1 Context and Limitations

Our modeling results, which were informed by our 2-year field study, indicate that the 3 riparian wetlands we monitored likely serve as long term net TP sinks. In a review of wetland buffer zones, Walton et al. (2020) reported mean TP retention 0.7 ± 1.4 g P m^{-2} yr^{-1} (49 studies, riparian wetlands, fens, and floodplain wetlands) corresponding to retention efficiency of 21±72%. Simulated TP retention estimates at our sites, which ranged from -0.1 g P m^{-2} yr^{-1} to 0.5 g P m^{-2} yr^{-1} with a mean efficiency of 35±30%, were comparable but lower in magnitude.

Before this study, the few literature estimates available indicated that TP retention varied widely but that net release occurred on average for restored riparian wetlands on historically drained and farmed soils (Land et al. 2016). However, many of the relevant TP retention studies were conducted within 3 years post-farming (Ardón et al. 2010; Hoffmann et al. 2012), when release of agricultural legacy soil P is likely to be greatest (Wiegman et al. 2022). Our sites had not been farmed for over 10 years. Prior research in VT-LCB riparian zones indicated that DIP release from soils can decline exponentially with time since farming at a rate of 7% to 10.5% per year (Wiegman et al. 2022). Accordingly, potential for DIP release at our study sites was low to moderate relative to other regional sites. Therefore, the sites monitored here represent the dynamics of maturing wetlands rather than a worst-case scenario of when flooding first occurs on a very recently restored farm field with soils that are heavily saturated with P.

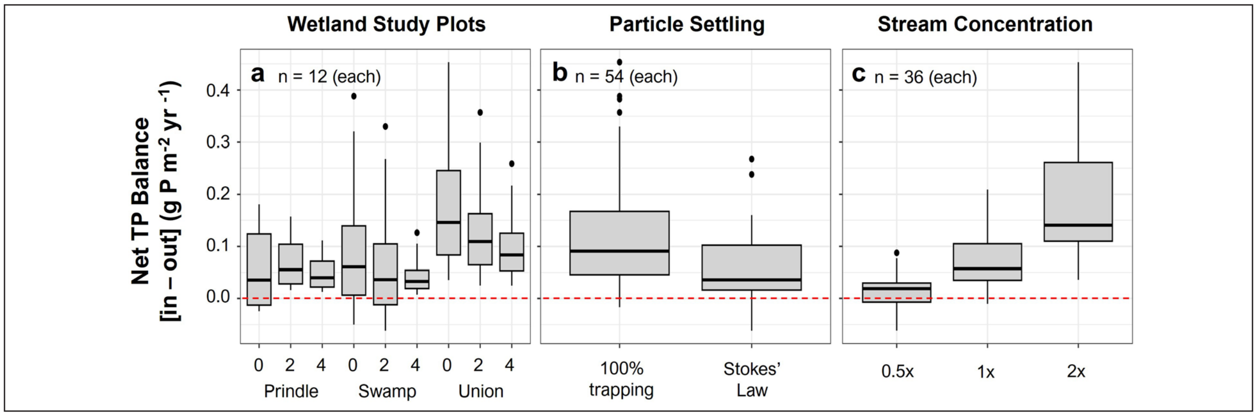

Fig. 5 Box and whisker plots of *wetlandP* simulated net total phosphorus (TP) balance for all simulations with power model HRT and stream derived inflow concentrations (108 total simulations divided among groups in each subplot, see Table 1). Results are grouped by (a) study wetland and sampling plot (low elevation = 0, median elevation = 2, high elevation = 4), (b) assumptions for particle settling, and (c) the factor by which stream (inflow) concentrations are varied for TP and TSS.

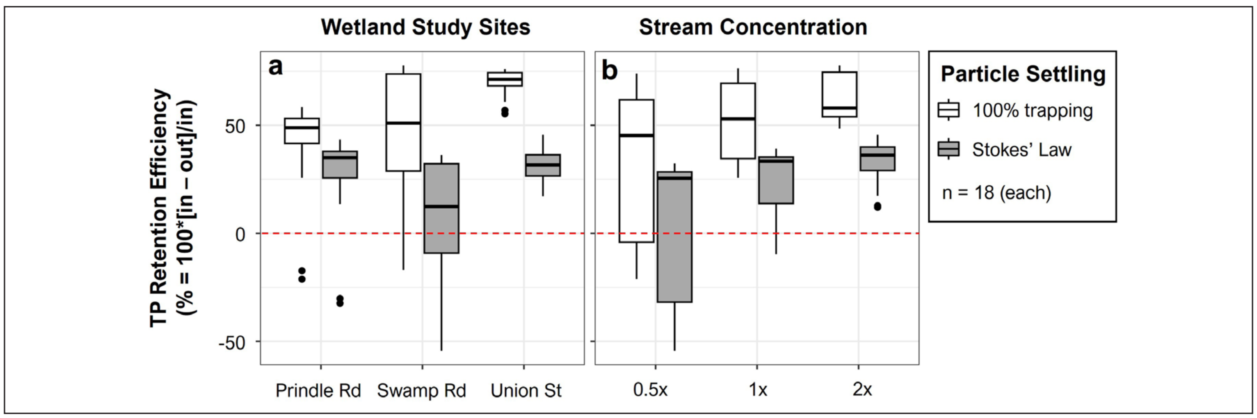

Fig. 6 Box and whisker plots of *wetlandP* simulated net TP retention efficiency (%) grouped by site (a) and stream concentration factor (b) and assumptions for particle settling (108 total simulations, 18 per group, see Table 1). White boxes indicate simulations where 100% particle trapping is assumed, and grey boxes indicate simulations where sedimentation is via Stokes' Law.

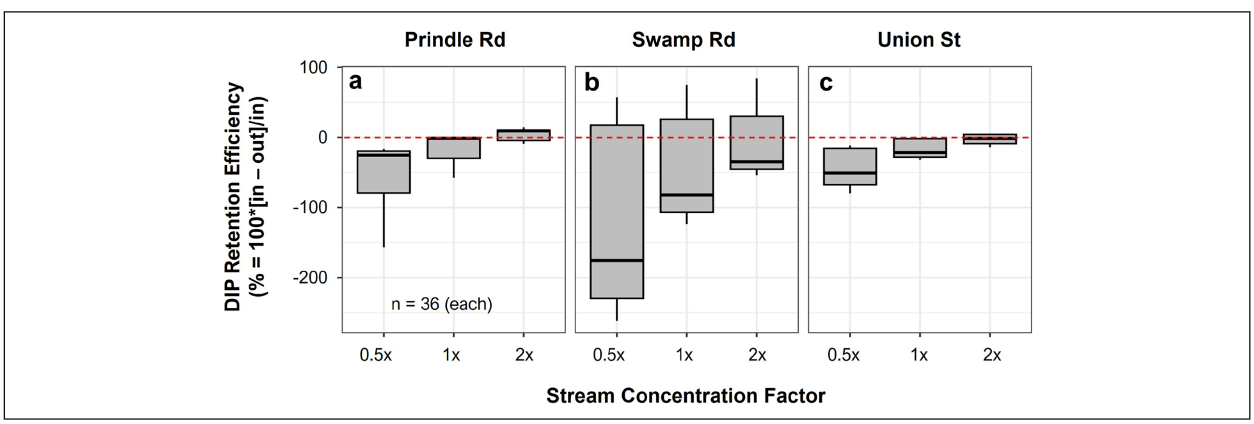

Fig. 7 Box and whisker plots of *wetlandP* simulated DIP retention efficiency, grouped by site and stream concentration factor (108 total simulations, 36 per group, see Table 1).

Potential for P capture was also relatively low at our sites compared to other sites in the region. Influent concentrations of TP and DIP have a strong impact on whether net TP balance tends towards capture or release (Fig. 5). The inflow TP concentrations we observed at our study sites (<0.1 mg P L^{-1}) were low relative to high-flow events in other agriculturally impacted rivers/streams of the VT-LCB, where TP concentrations are often near or above 0.2 mg P L^{-1} and can sometimes exceed 1 mg P L^{-1} (Underwood et al. 2017; Vaughan et al. 2018; Myers 2023). Our study areas were on the lower end of energy settings within the VT-LCB (Fig. 2a) (Diehl et al. 2022). So, it is not surprising that the particulate P deposition at the 3 sites that we monitored was on the lower end (up to approximately 1 g P m^{-2} yr^{-1}) of the spectrum reported by Diehl et al. (2022) for a broader suite of floodplains in the VT-LCB (1 g P m^{-2} yr^{-1} to >100 g P m^{-2} yr^{-1}).

Furthermore, sediment and P concentrations can vary with river/stream discharge and resulting flood pulse size, which our model did not account for. For example, Underwood et al. (2017) showed that TSS, particulate P, and dissolved P vary with river discharge in Vermont, with high discharge typically accompanied by relatively high concentrations. Discharges did not exceed that of a 2-year recurrence interval at any of the 3 sites during the monitoring period (Text S3). This suggests net TP retention could be substantially greater for large flood events (e.g., 10-year recurrence interval) than what we estimated in this study.

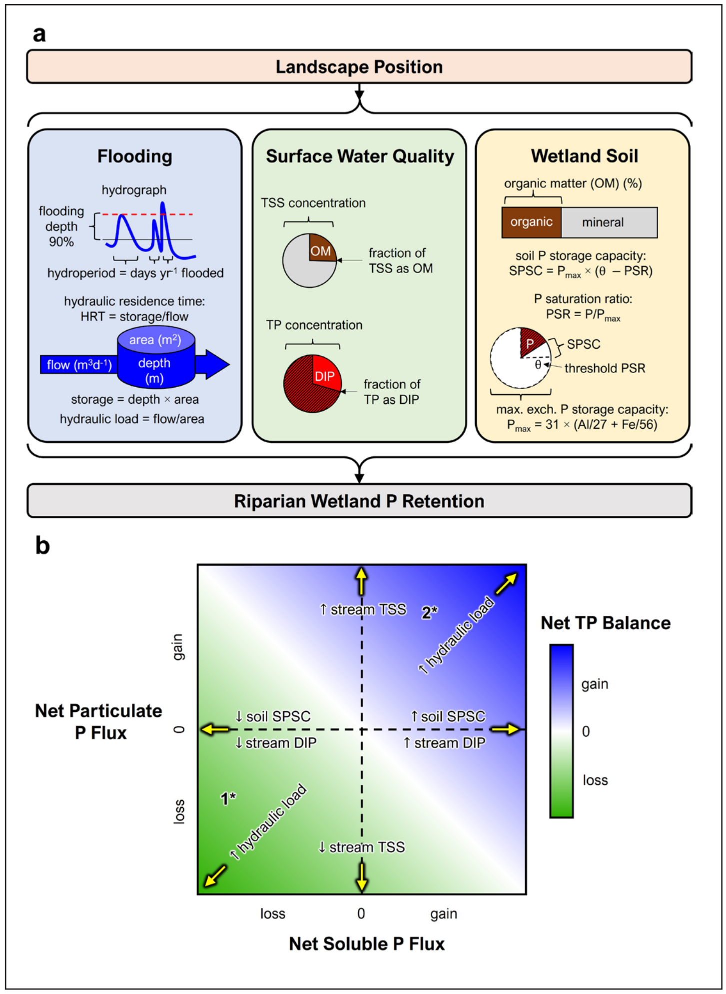

Fig. 8 Conceptual framework including: (a) key variables related to landscape position that influence net TP retention in riparian wetlands, and (b) hypothesized effects (yellow arrows) of increasing (↑) or decreasing (↓) a metric's value on particulate P flux (y-axis), DIP flux (x-axis), and overall net TP retention (blue = net TP gain, green = net TP loss). In (b), numbers marked with * correspond to hypothetical sites described in the text of Section 4.2.

4.2 Conceptual Framework for Ecological Engineering Design

Here we present a conceptual framework of key factors driving net TP retention in restored riparian wetlands on formerly farmed land (Fig. 8) that synthesizes the findings from the present field and modeling study and recent research on soils in the VT-LCB (Wiegman et al. 2022). Three factors influenced by landscape position have important influence on net TP retention (Fig. 8a): (1) flooding dynamics (e.g., flooding depth, frequency, hydraulic residence time), (2) influent surface water quality (TSS, TP, and DIP concentrations), and (3) wetland soils (organic matter, SPSC). Net particulate P flux is a function of river/stream sediment concentrations, with increases in hydraulic load exacerbating gain or loss (Fig. 8b). River/stream DIP concentration and SPSC are the key drivers of whether net DIP flux is likely to be negative (net release) or positive (net retention), with hydraulic load again exacerbating outcomes in either direction (Fig. 8b).

Ecological engineers and designers can leverage this conceptual framework to prioritize riparian wetland restoration activities when P load reduction is a primary objective, and to avoid scenarios where substantial net TP retention is clearly not favored. For example, imagine a scenario where 2 sites are being considered for riparian wetland restoration and water quality improvement via P capture is the primary objective. At Site 1, the available water quality data for the adjacent river during high flow conditions suggests that DIP concentrations are relatively low (averaging ~0.01 mg P L^{-1}), TSS averages

~20 mg L^{-1}, and the SPSC is substantially negative (near -1000 mg P kg^{-1}) for the recently farmed soils. At Site 2, river DIP and TSS are both greater (averaging ~0.05 mg P L^{-1} and ~100 mg L^{-1}, respectively), while SPSC is positive. Hydraulic loads to Sites 1 and 2 are similar. Based on Fig. 8, Site 2 would clearly be the superior option for enhancing net TP load reductions through wetland restoration, while Site 1 has features that indicate risk of net TP export. At Site 1, we would recommend that managers initially minimize hydraulic alterations that increase hydraulic load to the floodplain and that they evaluate the feasibility of actions to increase SPSC, such as biomass harvest (Carson et al. 2018), topsoil removal (Oldenborg and Steinman 2019; Zak et al. 2017), and/or amendment of soil with materials that have high P sorbing capacity (Ament et al. 2021; Hurst et al. 2022).

4.3 Management Considerations

Our findings highlight the importance of considering both particulate and dissolved P in P mass balances for restored wetlands, building on prior studies (Walton et al. 2020; Ury et al. 2023). Basing estimates of TP retention on P deposition (i.e., particle trapping) alone will likely lead to overestimates of net TP retention. Our field data provided evidence of internal DIP loading within the 2 Otter Creek wetlands (Fig. 3, Fig. S13, and Fig. S14), but not at Prindle Road. At Prindle Road, we observed a reduction of DIP and an increase in DO from inflow to outflow (Fig. 3), suggesting net DIP uptake by primary producers.

The trend of decreasing DIP retention efficiency with decreasing influent TP and DIP concentrations shown in Fig. 7 illustrates how legacy P and watershed P buffering capacity could potentially increase the time required to meet P reduction goals. For example, if upstream DIP concentrations decrease due to the adoption of best management practices, stream/wetland/floodplain sediments and soils can likely serve as a buffer and release DIP, moving toward an equilibrium with overlying waters (Reddy et al. 2011; Kusmer et al. 2018; Wiegman et al. 2022). In aggregate, this P buffering capacity causes watersheds to resist change in DIP concentrations while legacy P stocks slowly deplete, creating a time lag between actions (adoption of best management practices) and results (reduced riverine DIP concentrations) (Goyette et al. 2018).

In riparian zones and floodplains where agriculture is economically viable, it is important to contextualize potential P gains or losses of wetland restoration relative to continued farming. For example, in locations in or near the study region, P losses from fields producing corn silage or hay can be in the range of 0.03 g P m^{-2} $year^{-1}$ to 0.23 g P m^{-2} $year^{-1}$ (Eastman et al. 2010; Klaiber et al. 2020; Ruggerio et al. 2022). Therefore, net reduction over time in downstream P transport from both retiring a farm field *and* restoring wetland conditions could be substantially greater than our modeled estimates of net TP retention (or those reviewed by Walton et al. 2020 or Land et al. 2016). More research is needed in this area.

5. Conclusions

In this study, we combined an intensive 2-year field study of 3 restored riparian wetlands on formerly farmed land in the Vermont portion of the Lake Champlain Basin with implementation of a novel model to estimate net P retention, *wetlandP*. We calibrated and verified the *wetlandP* model using field data and used it to examine scenarios that represented a range of plausible conditions for each field site. Our simulations indicated variable net TP retention (approximately -0.1 g P m^{-2} yr^{-1} to 0.5 g P m^{-2} yr^{-1}, mean P retention efficiency = 35%), driven by a trade-off between particulate P trapping and DIP release, with most plausible scenarios (95 out of 108) indicating that the study wetlands serve as net P sinks on the landscape. However, our modeling results also showed that release of legacy soil P as DIP can be sizable in some cases (net DIP balance ranged from -0.11 g P m^{-2} yr^{-1} to 0.02 g P m^{-2} yr^{-1}), especially for riparian wetlands receiving river/stream water with relatively low DIP concentration. We leveraged these results to develop a conceptual framework that illustrates the importance of soils, hydrology, and influent water quality on the potential TP load reduction benefits associated with the restoration of riparian wetlands on formerly farmed land. This framework can help guide riparian wetland restoration by ecological engineers and designers when water quality improvement via P capture and storage is a priority. Future research should include monitoring and modeling of additional field sites and larger flood pulses than those observed here, to inform estimates of TP retention in restored riparian wetlands more broadly across space and time. The *wetlandP* model and conceptual framework should be reevaluated periodically with new data and case studies.

Supplementary Material

The online version of this article contains a link to supplementary material that includes: Text S1 Additional Model Documentation; Text S2 Additional Field Study Documentation; Text S3 Additional Results; Text S4 Flow Analysis. Within these supplementary text sections, 10 supplementary tables and 23 supplementary figures are provided.

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Author Contributions Statement

Conceptualization: ARHW, WBB, EDR; methodology: ARHW, KLU, ICA, EDR; data analysis: ARHW, EDR; laboratory analyses: ARHW, ICA; writing original draft: ARHW, TLC, EDR; review/editing original draft: ARHW, KLU, TLC, WBB, ICA, EDR; investigation: ARHW, EDR, KLU, ICA; resources: EDR; data curation: ARHW; supervision: EDR; project administration: EDR; funding acquisition: EDR, WBB. All authors have read and agreed to the published version of the manuscript.

Conflict of Interest Statement

The authors have no conflict of interest to report.

Data Availability Statement

A stable version of the *wetlandP* model (version 2.1) that was used in this paper (including detailed documentation, input and output data, and scripts for implementation, pre/postprocessing) is freely available on GitHub: https://github.com/arhwiegman/wetlandP_2p1_stable.

Related Publication Statement

Earlier drafts of sections of this manuscript are included in the PhD Dissertation of the lead author at the University of Vermont, as well as in a final report to the funding sponsor (Lake Champlain Basin Program).

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