Wetlands
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ARTICLE

Nutrient Release from a Recently Flooded Delta Wetland: Comparison of Field Measurements to Laboratory Results

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Abstract Wetland restoration is expected to reduce external phosphorous (P) loading to hypereutrophic Upper Klamath Lake in Oregon, USA, where P was identified as the primary driver of lake productivity. However, previous laboratory experiments showed that a large P release could occur when former agricultural land is re-flooded for restoration, thus presenting an initial challenge for wetland restoration projects. We tested whether results from those experiments were supported during the initial phase of restoration at the Williamson River Delta adjacent to Upper Klamath Lake. Our objectives were to document post-flood surface water nutrient concentrations, estimate the P mass released from the newly flooded wetlands, and compare these results to the laboratory experiments. Phosphorus concentrations in the wetlands ranged from 0.1 to 0.63 mg P/L and were up to six times greater than in the lakes, corroborating that the wetlands released P upon flooding. However, we estimated 2 Mg P released within three weeks of flooding, which is much lower than the anticipated 64-Mg release from these wetlands, and a fraction of the annual 21–25 Mg load from the Delta before reconnection. This pulse is expected to be short-term; longer term studies will address the role of these wetlands in retaining nutrients.

Keywords Wetland restoration · Water quality · Phosphorus release · Nutrient retention

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Introduction

Wetlands are recognized for their ability to provide a number of ecosystem services to watersheds, including refuge for fish and wildlife, stabilization of water supplies, flood control, opportunities for recreation and aesthetics, and improved water quality in downstream water bodies (Mitsch and Gosselink 2000). Draining and altering wetlands has occurred on a global scale, however, and estimates are that roughly half the world's historical wetlands have been altered and drained due to human activities (Zedler and Kercher 2005).

In the Upper Klamath Basin (the Basin) in southern Oregon, USA, 85–90% of wetlands present in 1890 have been drained and converted for agricultural use (Gearhart et al. 1995). The draining of marshes adjacent to Upper Klamath and Agency Lakes, the two largest lakes remaining in the watershed, over the past century is considered to be a contributing factor leading to the hypereutrophic state of these lakes (Snyder and Morace 1997; Bradbury et al. 2004; Eilers et al. 2004; National Research Council 2004). Excessive phosphorus (P) loading was identified as the primary factor driving nuisance blue-green algae blooms and water quality impairment in the lakes including low

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dissolved oxygen concentrations, elevated pH (9–10), and ammonia concentrations at toxic levels (>0.5 mg/L) (Boyd et al. 2002), conditions which undermine the recovery of two endangered fish species endemic to the Basin (US Fish and Wildlife Service 1988).

Wetland restoration is regarded as an important conservation strategy to reduce nutrient levels in the two lakes (Gearhart et al. 1995), and multiple wetland projects within the Basin are underway. However, the capacity of wetlands to reduce nutrient loads to downstream water bodies is uncertain (Fisher and Acreman 2004). Whether wetlands serve as a source or sink for nutrients depends on a number of different factors including hydrologic and geomorphic conditions, seasonal patterns of uptake and release, and ecosystem succession (Mitsch and Gosselink 2000). Restoring wetlands on former agricultural land poses additional challenges because years of cultivation often leave behind enriched and subsided soils which affect the hydrology and plant communities of the restoration wetlands (Zedler 2003; Graham et al. 2005). Furthermore, research has shown that restoring wetlands on former agricultural land releases large amounts of nutrients upon initial flooding (Novak et al. 2004; Aldous et al. 2005; Aldous et al. 2007; Montgomery and Eames 2008). If the wetland soils being restored are naturally nutrient-rich, or the area has been fertilized while previously in agricultural production, the potential for release upon flooding presents concerns when wetland restoration or construction activities are proposed (Pant and Reddy 2002; Aldous et al. 2005; Aldous et al. 2007).

In wetlands adjacent to Upper Klamath and Agency Lakes, the timing, amounts, forms, and mechanisms of nutrient release from flooded soils have been measured in soil core experiments (Aldous et al. 2005; Graham et al. 2005; Aldous et al. 2007; Stevens 2008; Duff et al. 2009) and *in situ* using porewater samplers at small spatial scales (Kuwabara et al. 2010). The underlying goal of such studies has been to provide information on how wetland restoration and management strategies can minimize the external nutrient load into the lakes.

In 2007, we documented the actual nutrient concentrations and loads in surface waters after initial flood-restoration of the Williamson River Delta (the Delta), and compared our results to those from prior laboratory experiments intended to predict the load. Specific objectives of this study were to: (1) document P and nitrogen (N) concentrations in the water column after initial flooding of the Delta; (2) quantify the total nutrient load after flooding; and (3) compare results with those from previous laboratory experiments in Aldous et al. (2007) which predicted the amount and timing of P release from the restored Delta wetlands once they were flooded. Based on the laboratory experiments, we hypothesized that nutrients would be released into the water column upon initial flooding.

Methods

Study Area

The Delta occupies approximately 2,200 hectares (ha), and lies within the Williamson River watershed in southern Oregon (Fig. 1). The Delta straddles the last six kilometers of the Williamson River before it empties into Upper Klamath Lake, a large and shallow lake (surface area 232 km^2, average depth 2.8 m at full pool; Wood et al. 2006). The Williamson River is largely fed by the Sprague River, and together the two watersheds account for about 51% of the inflow into Upper Klamath Lake and 47% of the external P load into the lake (Boyd et al. 2002). The 2,200-ha Delta wetlands represent about 0.2% of the entire drainage area of Upper Klamath Lake (Boyd et al. 2002) and about 6% of the lake's surface area. Additionally, the wetlands contribute about 21.6×10^6 m^3 to the 600.5×10^6 m^3 active water storage volume of Upper Klamath Lake at full pool, which represents an approximate 3.6% increase in

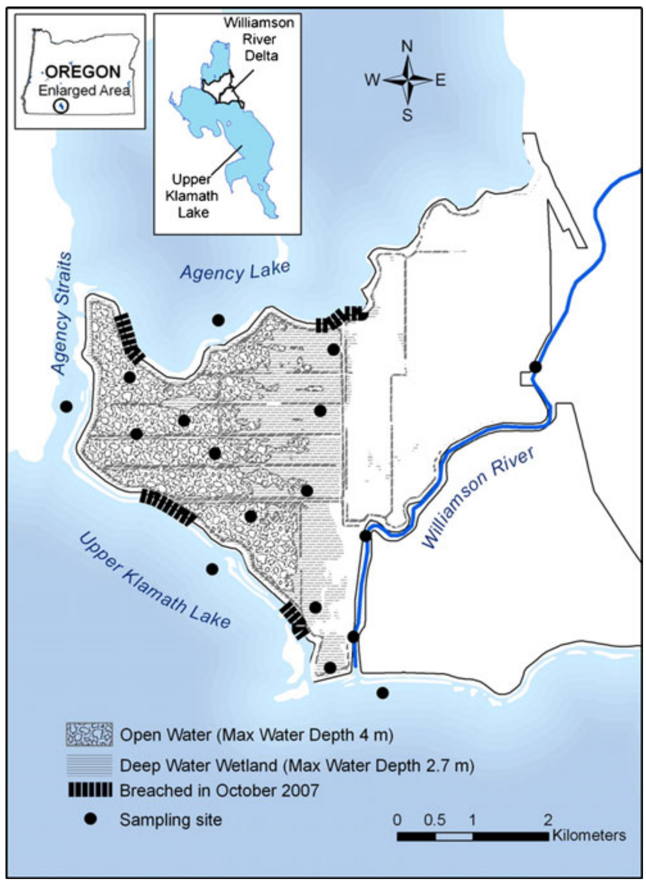

Fig. 1 Map of the Williamson River Delta in Upper Klamath Basin, Oregon showing the area initially flooded during the October–November 2007 study period (open water and deep water wetland), breach locations, and sampling sites. Lake water levels were low during this study, and lake water entered the Delta only through small openings (<10 m) within breach locations

active storage after flooding (David Evans and Associates, Inc. 2005).

Historically, the Delta was a fully functional freshwater marsh ecosystem that hydrologically connected the Williamson River and Upper Klamath and Agency Lakes. During the 1940s, the Delta was leveed, drained, and used for agriculture until the late 1990s. Estimated P losses from the Delta during its time under cultivation were about 21–25 Mg total P annually (Snyder and Morace 1997), which accounts for about 12% of the total annual external P load into Upper Klamath Lake (Boyd et al. 2002). Restoration of the Delta was identified as a key strategy to reduce external nutrient loading into the lake (Gearhart et al. 1995). Concerns about the initial release of stored P from the drained wetland soils once flooded were investigated (Aldous et al. 2005, 2007), and planning and implementation of the Delta restoration were aimed toward minimizing impacts.

Levees surrounding the Delta west of the Williamson River were breached (excavated to an elevation below the lowest allowable lake levels) with explosives at four locations on October 30, 2007 to commence flooding and facilitate wetlands restoration (Fig. 1). Once breached, the Delta re-flooded within a period of three days. Breaches are situated on the north and southwest perimeters of the Delta and were sited based on hydrologic modeling (Daraio et al. 2004). The breaches are approximately 640–823 m in length, allowing for water movement between Agency and Upper Klamath Lakes and the Delta at all but the lowest lake levels.

The hydrology of the Delta is influenced by that of the lakes and river. The US Bureau of Reclamation currently regulates surface water levels in Upper Klamath Lake, with lake levels generally fluctuating approximately 1.5 m seasonally. Substantial subsidence has occurred on western portions of the Delta such that these areas are inundated year-round, resembling open water conditions, while eastern portions of the Delta nearer the Williamson River function as seasonally-flooded emergent and riparian wetlands.

During the sampling period in November 2007, water movement occurred from the lakes to the Delta through small openings (<10 m wide) in the four breaches and not at all through the perimeter levees because they were above lake level. The Williamson River was not hydrologically connected to the sampling area in the Delta because of low fall/winter flow. Three days after levee breaching, the western portion of the Delta became fully inundated and sufficient for accessing sampling sites. Neither groundwater discharge nor recharge is a significant part of the water balance in this wetland (David Evans and Associates, Inc. 2005).

Soils within the Delta consist predominantly of Lather muck and Tulana silt loam (Cahoon 1985). Lather muck soils are poorly drained, low bulk density organic soils formed in deep deposits of partially decomposed fibrous organic material with thin layers of silt (Cahoon 1985). These soils are found in the lowest topographic elevations of the Delta on the west side. During the sampling period, all 10 sampling sites within the Delta occurred in the deeper water areas underlain by lather muck soils. Repeated draining and flooding of the land during cultivation resulted in soil subsidence on western portions of the Delta due to organic matter decomposition, compaction, and erosion. Soil losses from subsidence range up to 2–3 m in depth compared to historic elevations, and 30–40% of the Delta is now 2–3 m below average water levels in Upper Klamath and Agency Lakes (David Evans and Associates, Inc. 2005). Increased soil bulk density at the Delta wetlands compared to natural marshes in surrounding areas is a likely consequence of subsidence (Aldous et al. 2005; Graham et al. 2005).

In some areas within the study area, large tracts of wetland vegetation had been established in prior years by means of pumping water onto former agricultural parcels (Elseroad et al. 2006) while other areas remained in agricultural production. Thus some spatial variability within the sampled area existed because of differences in water and land management across the Delta prior to the fall 2007 restoration.

Water Sampling

Surface water samples were collected at 17 sites and analyzed for constituents of N and P (Fig. 1). Sample sites inside the Delta were stratified based on water depth ranges in which different plant communities were predicted to colonize (Elseroad 2004) and by water movement patterns through the Delta (T. Wood, US Geological Survey, personal communication). The water depth ranges of the sampled wetland area were classified as open water (4 m maximum) and deep water wetland (2.7 m). For this study, five sites were sampled in open water and five in deep water wetland. Emergent and transitional wetlands were not sampled because they were not flooded due to low lake levels. Seven sites were sampled outside the Delta as experimental controls: three in the Williamson River and four in the lakes surrounding the Delta (Fig. 1).

Surface water grab samples were collected one week prior to re-flooding the wetland (October 23, 2007), and two and three weeks after re-flooding (November 13–14 and 20–21, 2007). During the pre-flood sampling event, water samples were collected only at the seven locations surrounding the Delta. Water samples were collected at all sites two and three weeks after the levees were breached, when areas within the Delta were fully inundated and sampling site access was feasible.

Samples were collected at mid-depth in the water column using a 3.2 L Van Dorn horizontal beta sample collection device. When water depths exceeded 2 m, samples were collected at 1 m below the water surface. Water for nutrient analysis was transferred from the Van Dorn to a churn splitter, mixed, and then transferred to triple-rinsed 500 mL clear polyethylene bottles. Quality assurance samples were collected during each sampling event to validate accuracy of surface water samples and included split, rinsate, laboratory, and equipment blank samples.

All samples were stored in a cooler on ice at about 4°C and then delivered to the The Klamath Tribes' Sprague River Water Quality Laboratory in Chiloquin, Oregon immediately after sampling.

Physico-chemical parameters, including water temperature, dissolved oxygen (DO), pH, and electrical conductivity were measured instantaneously during each grab sample collection event using a calibrated multi-probe instrument (Hydrolab Quanta). Measurements were taken at mid-depth in the water column or at 1 m below the water surface if water depth exceeded 2 m.

Chemical Analysis

Water samples were analyzed by the Sprague River Water Quality Laboratory for P and N constituents. Phosphorus constituents included total phosphorus (TP) and soluble reactive phosphorus (SRP). Nitrogen constituents included total nitrogen (TN), nitrate+nitrite (NO_3^-+NO_2^-), nitrite (NO_2^-), and ammonium (NH_4^+). Approximately 120 mL of unfiltered sample water was transferred to triple-rinsed 125 mL amber polyethylene bottles for analysis of TN and TP and acidified with 1 mL of 4.5 N H_2SO_4. Water samples for TP and TN analysis were digested using potassium persulfate, autoclaved, and analyzed with an automated spectrophotometer (SM 4500-P H and Enzymatic NO3).

On the day of sampling, samples for dissolved nutrient (SRP, NO_3^-+NO_2^-, NO_2^-, NH_4^+) analysis were filtered through 0.45 μm sterile membrane filters (Millipore®) using a vacuum pump and a 300 mL magnetic filter funnel (Pall Gelman®). Filtered samples were transferred to triple-rinsed 125 mL amber polyethylene bottles. Samples of SRP, NO_3^-+NO_2^-, NO_2^-, and NH_4^+ were analyzed with the colorimetric method using the same automated spectrophotometer (SM 4500-P F, Enzymatic NO3, SM 4500-NO2, and MD Krom).

We report our data in three ways, first as standard concentrations, summarized in Table 1. Organic N concentration was calculated by subtracting dissolved inorganic N (DIN; assumed to be the sum of NH_4^+ and NO_3^-+NO_2^-) from TN. Second, we corrected for depth at each location because we were interested in nutrients generated from benthic sources in habitats at different water depths. These calculations correct for differences in dilution at each sampling site, making the assumptions that N and P in the newly restored areas were generated from benthic sources. We assumed little mixing of water between the two habitat types that would confound these results because they each cover large areas and there was little hydraulic gradient that would lead to significant mixing. We illustrate these data in Fig. 2, in units of g/m^2. We also calculated N and P benthic fluxes for the first sampling period (days 0–14) by dividing the depth corrected concentration data by 14 days, and report those flux values in units of g/m^2/day.

Total P and N Load Calculations

To calculate the TP and TN loads into the water column from the newly flooded wetland, we assumed that all water entering the Delta came from Upper Klamath and Agency Lakes. Thus the nutrient loads were calculated by subtracting the pre-flood lake TP or TN concentration from the TP or TN concentration in the open water and deep water wetlands, and then multiplying by the volumes of those wetland types. This was done for both post-flood sampling events, where the first event gives the load for days 0–14 and the second for days 0–21. Loads for days 14–21 were obtained by subtracting these two load values. Because the Williamson River was not hydrologically connected to the study area during the sampling period, pre-flood river concentrations were not included in the calculation. ArcGIS® version 9.3 (ESRI, Inc. 2009) was used to determine the area of flooded wetland based on average lake surface elevation during the sampling dates. The volume of flooded area was calculated by multiplying the area in open water and deep water wetland by the average water depths of those wetland types. Average water depth in each wetland type was determined using an as-built bathymetric xyz survey grid of approximately 40,000 points established across the Delta. The equation used to estimate the TP load is summarized below:

$$\text{TP load} = \left\{\left(\text{Average } [\text{TP}]_{\text{DW}} - \text{Pre-flood } [\text{TP}]_{\text{Lake}}\right) * \text{Volume}_{\text{DW}}\right\} + \left\{\left(\text{Average } [\text{TP}]_{\text{OW}} - \text{Pre-flood } [\text{TP}]_{\text{Lake}}\right) * \text{Volume}_{\text{OW}}\right\}$$

where $[\text{TP}]_{\text{DW}}$, $[\text{TP}]_{\text{Lake}}$, and $[\text{TP}]_{\text{OW}}$ are the TP concentrations in the deep water wetlands, lake, and open water, respectively. The same calculations were used for estimating TN loads.

Statistical Analysis

Depth corrected (DC) nutrient concentrations were statistically compared across four habitat types—two wetland

Table 1 Post-flood nutrient concentration means and ranges in mg P or N/L over two sampling periods for lake, river, and wetlands within and surrounding the Williamson River Delta, Oregon, November 2007

		Lake (n=8)	River (n=6)	Open water (n=10)	Deep water wetland (n=10)
Total phosphorus (mg P/L)	Mean	0.066	0.063	0.168	0.40
	SE	0.006	0.004	0.019	0.05
	Min	0.046	0.053	0.100	0.17
	Max	0.089	0.073	0.264	0.63
Soluble reactive phosphorus (mg P/L)	Mean	0.037	0.072	0.133	0.29
	SE	0.007	0.001	0.020	0.05
	Min	0.022	0.070	0.051	0.11
	Max	0.070	0.074	0.244	0.53
Total nitrogen (mg N/L)	Mean	0.78	0.08	1.16	1.68
	SE	0.17	0.00	0.06	0.17
	Min	0.09	0.07	0.85	0.85
	Max	1.36	0.09	1.54	2.47
Organic nitrogen (mg N/L)	Mean	0.55	0.06	0.99	1.59
	SE	0.12	0.01	0.07	0.16
	Min	0.06	0.04	0.74	0.79
	Max	1.01	0.07	1.46	2.36
Ammonium (mg N/L)	Mean	0.114	0.013	0.066	0.068
	SE	0.024	0.001	0.013	0.013
	Min	0.017	0.011	0.017	0.019
	Max	0.224	0.015	0.145	0.147
Nitrate+Nitrite (mg N/L)	Mean	0.122	0.009	0.097	0.024
	SE	0.025	0.001	0.020	0.007
	Min	0.010	<0.008	<0.008	<0.008
	Max	0.182	0.010	0.174	0.055

SE Standard Error

habitat types and two 'outside' habitat types that served as controls. The two wetland habitat types included open water and deep water wetlands. The two outside habitat types included lake (Agency Lake, Agency Straits, and Upper Klamath Lake sites) and river (three sites in the Williamson River).

We used two separate repeated measures models (PROC MIXED in SAS 9.2, SAS Institute, Inc. 2008; Littell et al. 2006) to examine how DC nutrient concentrations varied: (1) from pre- to post-breaching in lake and river only; and (2) across all four habitats after breaching. The first analysis tests the assumption that temporal variations in DC nutrient concentration are negligible for lake and river, allowing us to interpret significant post-breach habitat differences as reflective of hypothesized nutrient pulses from the wetland habitats.

We fit two-way models for each response variable, testing for main Habitat and Event (i.e., sampling event) effects after testing for their interaction. Event was the repeated measures factor and sampling site within each habitat was the within-subjects factor. The procedure described in Littell et al. (2006) was used to fit covariance models considered appropriate for repeated measures data to both raw and transformed data (natural log or square root). Compound symmetry (or the version allowing unequal variances) provided the best fit for every variable. Residual plots were used to assess agreement with assumptions of normality and homogeneous variances (i.e., for selecting raw vs. transformed data). We used Schwartz's Bayesian information criterion (BIC) for selecting the best covariance model that met these assumptions. BIC favors models with fewer estimated covariances and has more statistical power than AIC at the expense of a higher Type I error (Littell et al. 2006). We tested for fixed effects at α=0.05 with the selected model and compared differences in post-breach concentrations among habitats with Tukey's HSD. We used Slice tests to examine the pattern of simple effects driving statistically significant interactions. Only post-flood NH_4^+ concentration yielded a significant interaction. This was driven by uneven temporal effects among habitats, so no additional pairwise comparisons were required. The KENWARDROGER option was used for calculating degrees of freedom for all tests (Littell et al. 2006).

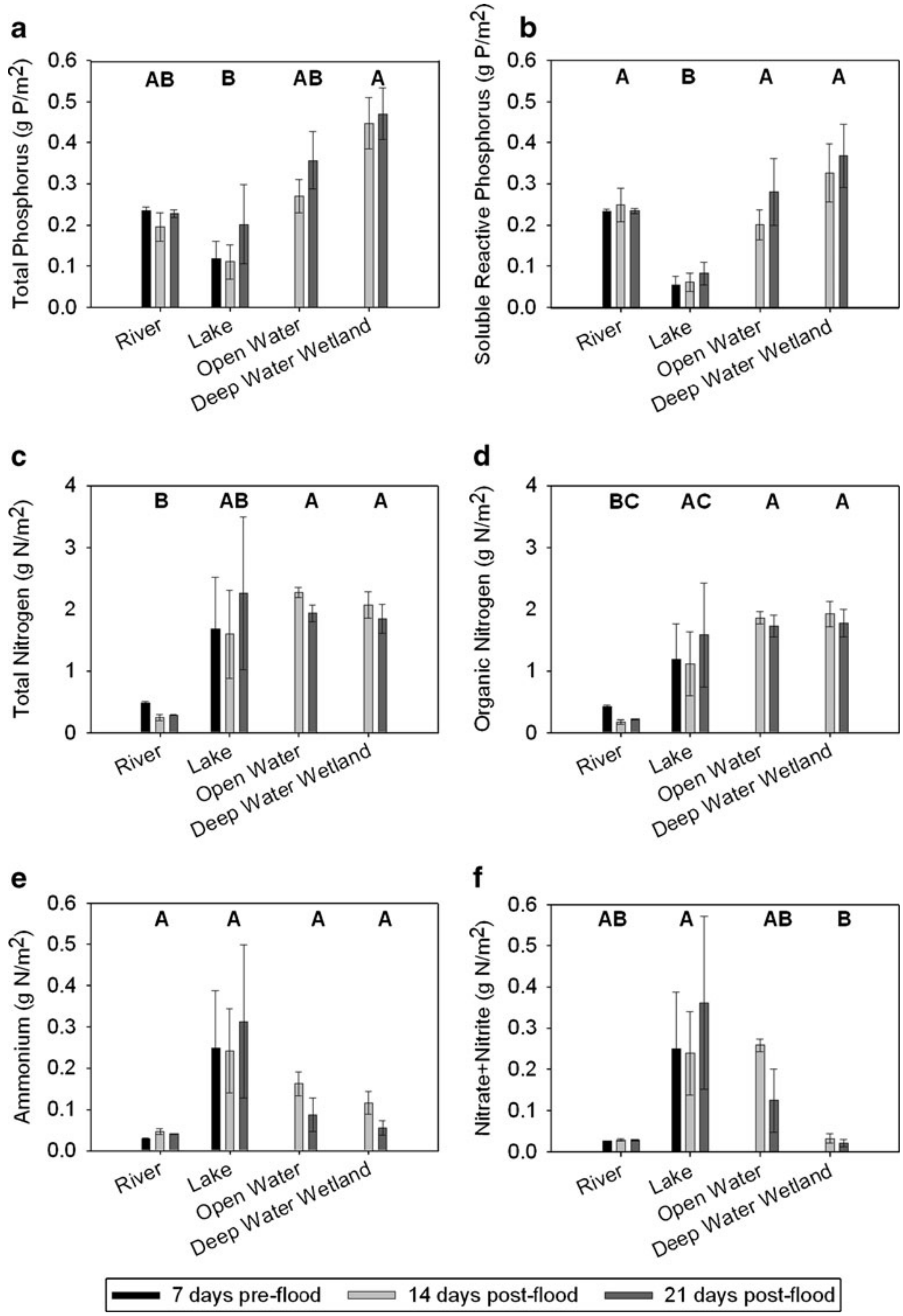

Fig. 2 Depth corrected nutrient concentrations in river, lake, open water, and deep water wetland habitats for one sampling event before and two events after levee breaching on the Williamson River Delta, Oregon, October–November 2007. *Panels a–f* show TP, SRP, TN, organic N, NH_4^+, and $NO_3^-+NO_2^-$ in g P or N/m^2, respectively. Habitats sharing letters indicate no statistical difference between the habitat types for post-flood data. Data were LN transformed for TN, NH_4^+, SRP (post-flood only), and $NO_3^-+NO_2^-$ (post-flood only). The SQRT transformation was used for organic N. All others were untransformed. See Methods for more detail on statistical analyses

Results

Total P concentrations among wetland sites ranged from 0.1 to 0.63 mg P/L during the post-flood sampling period, while river and lake concentrations ranged from 0.053 to 0.073 mg P/L and 0.046–0.089 mg P/L, respectively (Table 1). Total N concentrations ranged from 0.85 to 2.47 mg N/L among wetland sites, 0.07–0.09 mg N/L in the river, and 0.09–1.36 mg N/L in the lake. Water depths measured during the post-flood sampling period averaged 1.3 m in deep water wetland, 1.9 m in open water, and 1.2 m in Upper Klamath Lake.

Before and After Flooding

No statistical differences in DC nutrient concentrations were detected among the three sampling events in lake or river habitat ($p>0.05$ for all constituents, Fig. 2).

In the analysis of post-flood DC nutrient concentrations in all habitats, we observed significant changes over time only for TP, SRP, and NH_4^+. Total P and SRP (log-transformed) increased at the second post-breach measurement ($p<0.01$ and $p<0.05$, respectively); most of this increase occurred in lake and open water sites (Fig. 2). Ammonium concentration varied differently over time across habitats ($p<0.05$, log-transformed data), decreasing by 123% in open water and by 127% in deep water wetland ($p<0.01$) but not significantly changing in lake or river.

Inside Wetland Versus Outside

After flooding, the average TP concentration in deep water wetland was about six times greater than in the lake and about two times greater than in open water (Table 1). When corrected for depth, TP concentrations were almost three times greater in deep water wetland than in the lake ($p<0.01$), but not statistically different from open water ($p>0.05$) (Fig. 2).

Soluble reactive P constituted the majority of TP in each habitat type during the post-flood sampling period, although a lower percentage occurred in the lake (55%) compared to the other habitat types (76% in open water, 73% in the deep water wetland, and 100% in the river) (Table 1). The average SRP concentration in deep water wetland was about eight times greater than in the lake and about two times greater than in open water (Table 1). Depth corrected SRP concentrations in deep water wetland and open water were significantly greater than in the lake ($p<0.01$ and $p<0.05$, respectively, for log-transformed data), but were not statistically different from each other ($p>0.05$) (Fig. 2).

Organic N constituted the majority of TN in each habitat type during the post-flood period: approximately 86% in open water, 94% in deep water wetland, 70% in lake, and 72% in the river (Table 1). Deep water wetland had lower DC $NO_3^-+NO_2^-$ concentrations than the lake after flooding ($p<0.05$, log-transformed data). No statistical differences were detected in DC TN, organic N, or NH_4^+ concentrations between lake, open water, and deep water wetland during the post-flood sampling period ($p>0.05$ for all constituents and habitats other than river).

Total P and N Loads

Subtracting the pre-flood lake TP concentration from the post-flood TP concentrations in the open water and deep water wetlands, we estimated that a total of 2.47 Mg TP were released over the 1,100 ha of flooded wetland (Table 2). Open water areas released roughly half of their TP load in each sampling period. In contrast, the entire load from the deep water wetland was released in the first 14 days. The data indicate a small reduction in TP in the deep water wetlands during the second sampling period. However, because of sample size constraints, we do not know if this value is significantly different from zero.

We estimated a net release of 5 Mg TN (Table 2). Both deep water wetland and open water areas responded similarly in this case, releasing all TN before the first sampling event. Similar to the TP release in the deep water wetland areas, negative values were observed during the second sampling period. Again, we do not know if this value is significantly different from zero.

Physico-chemical Parameters

Surface water temperatures ranged from 3.5 to 6.5°C among all habitat types during the post-flood sampling period (Fig. 3). Dissolved oxygen concentration ranged from 3.3 to 12.5 mg/L among all habitat types. The lowest DO concentrations occurred in the deep water wetland (5.9±0.6 mg/L; mean±standard error) and were lower than DO concentrations in the lake (10.2±0.4 mg/L) and river (10.7±0.1 mg/L). During the post-flood sampling period, pH ranged from 7.1 to 8.2 among all habitat types. Electrical conductivity values ranged from 93 to 170 μS/cm among all habitat types. The highest mean electrical conductivity

Table 2 Total loads estimated from the two wetland habitat types after levee breaching in 2007[a]

	TP load (Mg P)		TN load (Mg N)	
	Open water	Deep water wetland	Open water	Deep water wetland
0–14 days post-breach	0.79	1.18	3.4	3.3
14–21 days post-breach	0.54	−0.04	−0.8	−0.9
Total (0–21 days post-breach)	1.33	1.14	2.6	2.4

Loads were estimated for the periods 0–14 days and 0–21 days (the total). The period 14–21 days was obtained by subtracting 0–14 days from 0 to 21 days. Positive values indicate benthic release and negative values indicate benthic uptake

[a] Total loads were calculated using the following areas for open water and deep water wetland: 611 ha for open water; 382 ha for deep water wetland. Approximately 109 ha of wetland classified as 'emergent' (Elseroad 2004) were flooded based on coverage below the average lake level during the sampling period, however these areas were <0.3 m deep, too shallow for sampling.

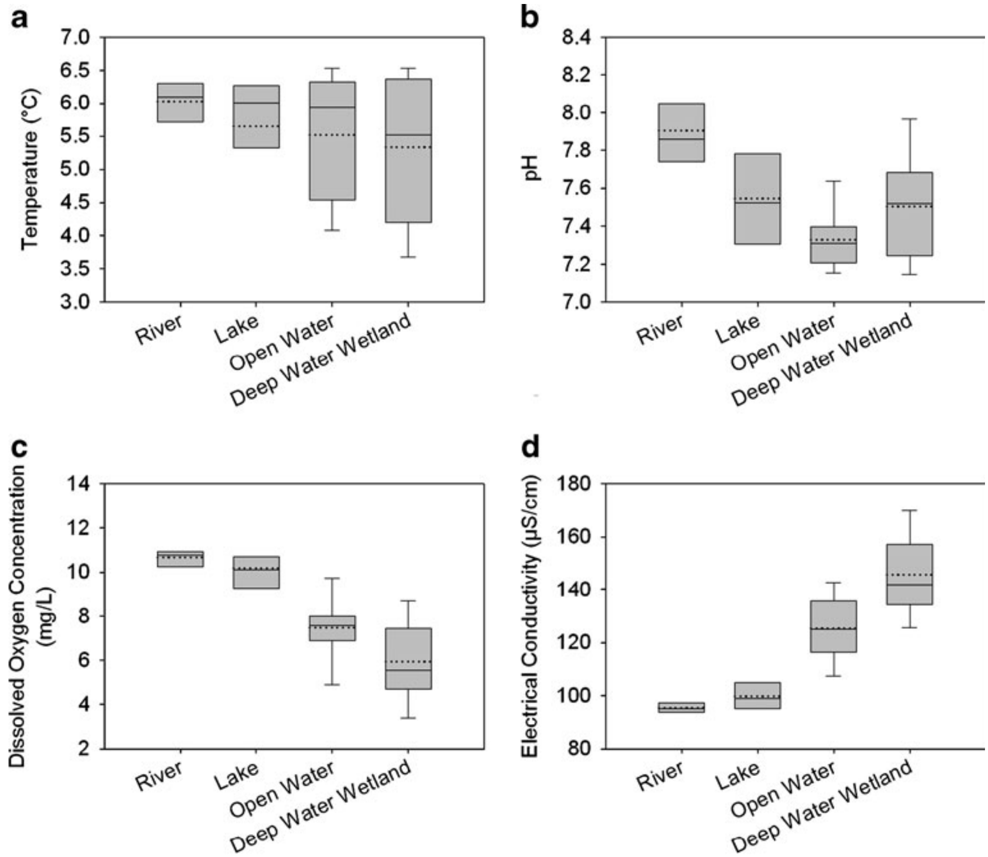

Fig. 3 Box plots for post-flood physico-chemical parameters in river, lake, open water, and deep water wetland habitat types in and surrounding the Williamson River Delta, Oregon, November 2007. *Panels a–d* show temperature, pH, dissolved oxygen concentration, and electrical conductivity, respectively. *Dotted lines* show means and *solid lines* show medians. Standard quartile, 5th, and 95th percentiles are also shown

values occurred in deep water wetland (145±5 μS/cm) while the lowest occurred in the river (99±4 μS/cm).

Discussion

Phosphorus Release

The significantly higher TP concentrations inside the wetland compared to outside, both two and three weeks after reconnection, indicate a release of P from the restoration wetland, thus supporting our initial hypothesis.

We estimated that 2.47 Mg TP were released over 1,100 ha of flooded wetland within three weeks following levee breaching. This value converts to a release of 0.2 g P/m^2 or 0.01 g P/m^2/day. Phosphorus release was previously documented in soil core incubation experiments from the same restoration wetland and nearby wetlands (Aldous et al. 2005; Aldous et al. 2007; Stevens 2008). Aldous et al. (2005) observed P fluxes from restoration wetland soils ranging 0.0086–0.055 g SRP/m^2/day between one and four days after flooding the soils, while Stevens (2008) observed P fluxes from the Delta wetland soils ranging approximately 0.03–0.06 g SRP/m^2/day in fall temperature treatments between one and three days after flooding. Positive benthic P fluxes ranging 0.00066–0.0042 g SRP/m^2/day in pore-water samples collected on the Delta were also documented two days after the levee breaching (Kuwabara et al. 2010). Aldous et al. (2007) observed a range in P release from 1 to 9 g P/m^2, with the majority of release occurring within two days of the experiment, and predicted a total release of about 64 Mg after flooding the wetlands. If we only include the values from areas where the current study took place (Fields 5 and 7 from Aldous et al. 2007), the area-weighted mean predicted by that study was 3.9 g P/m^2. Scaling this value up to 1,100 ha of flooded wetland results in a release of about 43.6 Mg P. While our current P release estimate falls near or within range of values in Aldous et al. (2005) and Stevens (2008), it is more than an order of magnitude less than the original predicted value in Aldous et al. (2007).

There are a number of possible explanations for the discrepancy between the two types of release estimates in the current study and Aldous et al. (2007), either because the laboratory experiment over-estimated potential P release, or because the field study under-estimated it. One likely explanation is differences in air and water temperatures during the two studies. The laboratory experiment was done in the summer and early fall, whereas the current study took place in November. Higher biological activity in the summer may have led to a higher benthic P release from the wetland. The hypothesis of a temperature control on P release is supported by other studies (Marsden 1989; Montgomery and Eames 2008; Stevens 2008). What is not known is how the reconnected wetlands will respond in subsequent years during the growing season. If they respond in a manner similar to the laboratory experiments, then the current study under-estimates long-term P release.

A second factor that might have led to a high P release estimate in the laboratory experiments is greater soil disturbance and edge effects associated with the soil cores. As part of the Delta restoration, construction activities disturbed the soils within the sampled area but probably not to the same extent as in the soil core experiments.

A third possible explanation for the difference between laboratory and field studies is in how each accounted for spatial variation. In the laboratory experiments, the TP load for the Delta was estimated using soil cores from a small number of locations. It was shown that P release was markedly different depending on the soil type. Thus it is possible that those soil cores were not adequately representative of the entire area of the Delta flooded in November 2007. Similarly, in this field study, we made the assumption that the number of locations sampled was adequate to represent the Delta as a whole. We also made the assumption that mixing of water from Upper Klamath and Agency Lakes was adequate within habitat types of the Delta. If there was little mixing, this assumption may not be entirely valid.

There are other factors that may have led to differences in P release estimated between the two studies, including differences in pH or redox controls on P mobilization, or other biogeochemical factors.

Lake Conditions Before and After Flooding

Our results indicate no significant effect of the breaching on lake nutrient concentrations, despite the P release from the wetlands after flooding. This result is not unexpected, given that water exchange between the Delta and lakes was limited due to low water levels during the sampling period. Furthermore, the nutrient release may not have been enough to significantly change nutrient conditions in the lake given the high background P concentrations in Upper Klamath Lake (mean=0.06 mg/L; Boyd et al. 2002), and that the 2.47 Mg of released P would have comprised about 0.5% of the total annual P load in Upper Klamath Lake (467 Mg; internal load + external load) (Boyd et al. 2002).

Post-flood Wetland Conditions

Both depth corrected TP and SRP concentrations increased between days 14 and 21 in all habitats except the river (Fig. 2). This result differs from previous research conducted at the Delta and nearby wetlands, where the majority of P released in soil core experiments occurred within the first two and three days after initial flooding (Aldous et al. 2007; Stevens 2008), and quickly declined and stabilized after the initial peak release. We detected no declines in DC P concentrations in the two sampled wetland types between 14 and 21 days post-flood, suggesting that P uptake was insignificant or was equal to inputs from benthic loading. Similarly, Aldous et al. (2007) documented no P uptake during the four months of the flood experiment.

A benthic flux of N was generated to the water column, similar to P (Table 2), with TN fluxes ranging from 0.12 to 0.19 $gN/m^2/day$. The majority of N released was in the organic form (Table 1 and Fig. 2) similar to a nearby wetland restoration project (Duff et al. 2009). In contrast to the release of organic N, NH_4^+ from lake water was consumed over each time step in both the deep water wetlands and the open water areas, and $NO_3^-+NO_2^-$ was consumed in the deep water wetlands, mostly in the first time step, and in the open water, during the second time step. A similar result of NH_4^+ and $NO_3^-+NO_2^-$ consumption was documented in porewater samples collected two days after the levee breaching (Kuwabara et al. 2010).

The consumption of DIN in the newly flooded wetlands must have resulted from conditions in the Delta that were not present in the lake. There may have been DIN consumption via denitrification, if denitrifying microbes were present in the wetland soils at greater densities than in the lake sediments. Denitrifying microbes may have been limited by the availability of labile carbon in the lake, and dissolved organic carbon generated from flooding the organic soils (Kuwabara et al. 2010) may have promoted denitrification. Ammonium volatilization in the Delta is not a likely explanation as there were no conditions in the Delta wetlands that may have promoted this process any differently than in the lake. While the consumption of DIN occurred at the same time as total organic N production, biological uptake of DIN and conversion to dissolved organic N is unlikely at this time of year when rates of biological productivity are low.

Conclusion

This field study follows up on previous laboratory experiments from the same wetland restoration project and confirms a release of P with initial flooding, but of a much lower magnitude. It is not clear to what extent this discrepancy is due to the timings of the two studies and whether the decision to breach the levees in fall may have reduced the total benthic release. Regardless of which estimate is correct, this load is probably short-term, and is less than the continuous annual load originating from the Delta during its time under cultivation. Accordingly, wetland restoration is clearly a preferred land use over agriculture when considering water quality.

While this study documents and quantifies water quality impacts during the initial phase of wetland restoration—providing a key piece of information for wetland managers undertaking restoration projects—the important outstanding questions include: How long will the wetlands continue to export nutrients, especially P, and under what conditions (if any) will the wetlands transition from net nutrient source to net sink? What contribution will the restored Delta wetlands have in reducing the external nutrient load to Upper Klamath Lake? Finally, as focus throughout this watershed is directed toward wetland restoration and management projects in effort to return ecosystem functions provided by wetlands, what will be the cumulative impacts of restoration of all these lands on water quality in Upper Klamath and Agency Lakes?

Various biotic and abiotic factors not investigated in this study control P mobilization, so future studies will examine long term trends in water and soil chemistry—at scales sufficient to account for temporal and spatial variation—to understand the influence of various factors on nutrient dynamics at the Delta. This kind of work is important in order to address some of the outstanding questions, and to inform other restoration projects in this watershed and elsewhere.

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