Ecological Engineering 108 (2017) 255–262

Contents lists available at ScienceDirect

Ecological Engineering

journal homepage: www.elsevier.com/locate/ecoleng

Interplay of water quality and vegetation in restored wetland plant assemblages from an agricultural landscape

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ARTICLE INFO

Keywords:
Agricultural runoff
Eutrophication
Nitrogen
Sediment
Wetland restoration
Plant assemblage

ABSTRACT

Water quality degradation from excessive fertilizer use and runoff is a worldwide problem. While this degradation impacts wetlands, these systems can also be a vehicle for water quality improvement. Restoration of wetlands in agricultural landscapes has recently increased, but little work has evaluated the relationship of plant assemblages and water quality parameters in restored, non-treatment wetlands. This study examines the impact of self-designed wetland plant assemblages on nitrogen and sediment dynamics. Thirty mesocosms were seeded with soil from restored wetlands and allowed to develop from the seed bank to emergent assemblages. During the 2015 growing season (seven to nine months after establishment), these assemblages were exposed to treatment loads of nitrogen and sediment, common stressors to wetlands in agricultural landscapes. Water samples were taken up to five days post-treatment in July and September to quantify interactions between the stressors and plant assemblages. Analyses showed plant assemblage identify was not structured by treatment, but by the site of soil origin. Treatment removal rates were influenced by total amount of the stressor present, with nitrogen removal rates being higher, in relative terms, in low nitrogen amended treatments. Additionally, plant quality, not quantity, was linked to nitrogen and sediment loss rates, and over time, elevated nitrogen and sediment loads were associated with decreased plant assemblage quality. This study demonstrates the ability of plants from restored wetlands to affect nutrient and sediment dynamics, with three significantly differing plant assemblages all exhibiting substantial nutrient and sediment reduction capacity. Nevertheless, we also found that in a relatively short time (seven to nine months) common stressors in agricultural settings can significantly impact wetland plant assemblage quality, and that this may be linked to a reduced capacity for nutrient and sediment removal.

1. Introduction

Worldwide, aquatic environments have been degraded by land use change associated with agricultural production. In particular, excessive fertilizer use and subsequent runoff lead to eutrophication in freshwater and marine systems (Gilbert et al., 2014). Nutrient pollution is a primary cause of water quality degradation, with agricultural inputs acting as a major source of these nutrients (Ribaudo et al., 2001). Synthetic nitrogen (N) fertilizer entering waterways from agricultural field runoff is the largest input of N to the Mississippi River Basin (Howarth 2008) and is estimated to contribute approximately 50% of N entering the Gulf of Mexico (Ribaudo et al., 2001). This N input results in the formation of a hypoxic zone varying in size annually from about 7500 km^2 to 17,500 km^2 (Rabalais et al., 2001; http://www.gulfhypoxia.net). Water quality degradation is expedited and perpetuated through land conversion, with agricultural landscapes producing increased levels of surface runoff containing nutrients and sediments than undisturbed soils (Woltemade, 2000).

As primary aquatic systems, wetlands are disproportionately affected by these water quality degradations. Nutrient levels can affect plant assemblage dynamics on a species-by-species basis (Mahaney et al., 2004), but high levels of N seem to promote invasive establishment and/or decreased diversity (Beas et al., 2013). Extended periods of N loading, as well as periodic N pulses, contribute to wetland eutrophication and have the potential to alter plant community structure within aquatic systems (Thiébaut and Muller, 1999). While sedimentation rates vary between wetlands and geographic regions, anthropogenic impacts on sediment runoff are of major concern in agriculturally impacted wetlands (Richards et al., 1996) and affect seed bank and subsequent vegetative development (Peterson and Baldwin, 2004).

While water quality degradation impacts wetlands, these systems are also seen as a vehicle for water quality improvement. Increased hydrologic retention time allows for alteration of wetland processes to

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http://dx.doi.org/10.1016/j.ecoleng.2017.08.034
Received 2 May 2017; Received in revised form 15 August 2017; Accepted 30 August 2017
Available online 10 September 2017

occur, as wetlands develop unique suites of nutrient fixing microbes (Bourgues and Hart, 2007). Within wetlands, hydrophytic vegetation has significant and positive effects on water quality mitigation, with pollutant removal efficiencies varying among species (Brisson and Chazarenc, 2009). Although some uncertainty exists in causal mechanisms behind differences in species' abilities to alter water quality, certain traits have been suggested to contribute to greater removal efficiency.

Tanner (1996) found a linear correlation between plant biomass and removal of total N, while differences in growth form affect the location of nutrient uptake. Increased plant surface area allows for greater microbial attachment (Brisson and Chazarenc, 2009), however Pettit et al. (2016) suggest plant structural complexity was more influential on microbial biomass than surface area alone. Development of oxidized rhizospheres impacts soil biogeochemical environment, and subsequently, microbial communities. Aldridge and Ganf (2003) found differences in species' abilities to alter oxidation-reduction (redox) potential in soils, with *Typha domingensis* increasing levels 218 mV above bare sediment compared to 41 mV for *Bolboschoenus caldwellii*, while *Potamogeton crispus* had no influence on redox potential in flooded soils. Even in heavily reduced conditions, the presence of plants increases soil microhabitat heterogeneity, engendering a diversity of processes conducive for nutrient removal (Brix, 1994; Sorrell and Armstrong, 1994; Jesperson et al., 1998; Shoemaker and Kröger, 2017).

Effects of wetland vegetation on pollutant loads have been extensively noted and studied in wastewater treatment wetlands. In a metaanalysis conducted on vegetated treatment wetlands dosed with livestock effluent, median removal rates of 57% for total suspended sediments (TSS) were observed (Knight et al., 2000) while NO_3^- and PO_4^{3-} removal rates of 86% and 56%, respectively, were observed in stormwater retention ponds planted with duckweed (*Lemna minor*) (Sims et al., 2013). Traditionally, studies on wetland plants' ability to ameliorate water quality occur in wetlands planted with monotypic stands of vegetation or in microcosms (Reddy et al., 1983; Cronk and Fennessy, 2001; El-Sheikh et al., 2010). The applicability of such results to diverse plant assemblages is less well known, especially the impact of assemblages not directly managed for water quality improvement.

The current study is unique in that it addresses effects of plant assemblages on water quality parameters in a replicated experimental study. In particular, a dearth of research exists on water quality improvement in non-flow through wetlands in an agricultural matrix, an area of critical need for mitigation of nutrient and sediment runoff from some 1.5 billion ha of crop-producing land worldwide (FAO, 2015). By determining whether and how self-designed plant assemblages, such as those resulting from passive restoration efforts, affect nutrient and sediment retention in non-treatment wetlands, best management techniques can be developed to ameliorate water quality conditions in areas other than urban and industrial settings. Our specific aim in this study was to measure the relative importance of plant assemblage composition on nutrient and sediment abatement in non-flow-through wetlands, such as those typically enrolled in wetland conservation programs in the United States. We hypothesized that plant species identities, rather than total cover, would be more strongly associated with water quality improvement, and that, as nutrient and sediment exposures increased, differences among plant species assemblage composition would become more evident

2. Methods

This study took place at the Aquatic Plant Research facility located on the R.R. Foil Plant Science Research Center, at Mississippi State University (33° 28′N, 88° 46′W). Mesocosms for this study (30) were 568 L Rubbermaid stock tanks. Each mesocosm was constructed using 57 L of filler sand and 19 L of wetland soil, placed and spread out evenly over top of the sand. The wetland soil was collected on 1 November 2014 from three restored wetlands (Burrell South, Muddy Bayou, Moon Bayou) enrolled in the Wetland Reserve Program (WRP) located in the Mississippi River Alluvial Valley (the Delta). Wetlands from which the soil was taken were greater than 10 years post-restoration, with clay soils of the Alligator-Sharkey-Dundee-Forestdale series. One soil sample from each wetland was placed in its own container, for a total of 10 soil samples per site, across three soil seed bank source wetlands (n = 30 total mesocosms). In addition to the 30, three more mesocosms were constructed with sand only and three constructed with sand and soil (but weeded regularly), to estimate the effect of soil and plants, respectively. Mesocosms were placed on leveled gravel and oriented in the same direction, with approximately 0.5 m of buffer space between neighboring mesocosms. Following their construction in December 2014, all mesocosms were flooded for the duration of the winter.

Mesocosm hydroperiods were managed following a slow, midseason draw-down, common in WRP wetlands (LMJV, 2007), which occurred from 19 April 2015 to 5 May 2015. Water was removed by siphoning. For the time period covered in this study (July–September), water levels fluctuated between moist soil and inundated to a depth of 5 cm, with water levels at a depth of approximately 5 cm before treatment application (see following paragraph). Mesocosm plant assemblages were allowed to grow from seed banks existing in the soil from the three Delta wetlands. Shortly before treatment application (within a week, see below), percent cover was recorded for each plant species present, to quantify plant assemblage composition. Additionally, similar plant species cover data were collected from the three soil source wetlands, these data were collected four times, once each in May and August of 2014 and 2015, to allow for comparison of plant assemblages between mesocosms and the wetlands from which the seed banks were sourced. In the wetlands, plant assemblages were characterized using 50, 0.8m^2 circular quadrats systematically spaced among 10 parallel transects 20 m apart from each other, with quadrat spacing of 20 m along the transect. Species identification followed Godfrey and Wooten (1981), and naming was consistent with the USDA-PLANTS database (plants.usda.gov).

Nitrogen and sediment were selected as stressors for this study because they are common contaminants in wetlands on agricultural landscapes (Baker et al., 2016). Four treatments and a control were included in a randomized complete block design, with the three wetland soil sources serving as a blocking factor (Table 1). In addition to controls, which received no experimental nutrient or sediment inputs, we used two levels of N in the form of NO_3^- and two levels of sediment, in a factorial design. Levels of each contaminant were selected based on observed patterns in Delta wetlands. High and low NO_3^- loads of 20 g and 1.25 g per year were based on observations from agricultural runoff in Delta drainage ditches (Baker et al., 2016), while high and low sediment accumulation rates of 2 cm and 0.5 cm per year were calculated from rates observed in a Delta oxbow lake wetland over the last 200 years (Wren et al., 2008). Sediment accumulation rates were converted to loads by calculating dried sediment density and converting weight to

Table 1
Experimental design of study. Thirty total mesocosms. N = Nitrogen, Sed. = Sediment

	Treatment				
Block (wetland soil site)	(1) High N, High Sed.	(2) High N, Low Sed.	(3) Low N, High Sed.	(4) Low N, Low Sed.	Control
(1) Burrell S (n = 10)	Y_{11} (n = 2)	Y_{12} (n = 2)	Y_{13} (n = 2)	Y_{14} (n = 2)	Y_{1c} (n = 2)
(2) Muddy Bayou (n = 10)	Y_{21}(n = 2)	Y_{22} (n = 2)	Y_{23} (n = 2)	Y_{24} (n = 2)	Y_{2c} (n = 2)
(3) Moon Bayou (n = 10)	Y_{31} (n = 2)	Y_{32} (n = 2)	Y_{33} (n = 2)	Y_{34} (n = 2)	Y_{3c} (n = 2)