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Sedimentation in created freshwater riverine wetlands: 15 years of succession and contrast of methods

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ABSTRACT

This study summarizes five separate sedimentation studies spanning 15 years (years 3–17 following wetland creation in 1994) of two 1 ha experimental flow-through wetlands. Included are methods and analyses of the most recent (2009–2010) comparative study that attempted to quantify both erosion and bioturbation processes. Depending on techniques used, two distinct types of sedimentation rates were estimated—gross and net sedimentation. Gross sedimentation in 2004–2005 (years 11 and 12) using sediment trap bottles was 45 kg m^{-2} for 4 months during and after spring flood pulsing conditions in 2004 and 39 kg m^{-2} for the same 4 months during and after steady flow conditions in 2005. Annual sediment accretion using feldspar and other horizon markers was 31.7 ± 4.4 kg m^{-2} yr^{-1} (4.2 ± 0.6 cm yr^{-1}) in 1996 (year 3 after wetland creation) and 34.4 ± 4.5 kg m^{-2} yr^{-1} (5.5 ± 0.8 cm yr^{-1}) in 2009 (year 16 after wetland creation). Net sedimentation using soil cores to estimate accumulation of sediments over antecedent soil horizon layers was 4.7 ± 0.3 kg m^{-2} yr^{-1} (0.9 ± 0.07 cm yr^{-1}) in 2004 (year 11) and 6.0 ± 0.4 kg m^{-2} yr^{-1} (0.9 ± 0.06 cm yr^{-1}) in 2009 (year 16). Net sedimentation, using estimates of sedimentation and erosion with the sediment erosion table (SET) method in 2009–2010 (years 16–17) was 3.9 ± 6.1–9.0 kg m^{-2} yr^{-1} (1.3 ± 0.8–1.4 cm yr^{-1}). Bioturbation by macrofauna significantly decreased sedimentation rates during the 2009 (year 16) study. Spatial patterns, consistent among horizon marker and net sedimentation studies, showed that deep, open-water areas had higher rates of sedimentation than shallow areas with emergent vegetation, and that sedimentation rates were higher when closer to the inflow than at the outflow of these flow-through wetlands. Net sedimentation of these created riverine wetlands ranged from 1.2 cm/yr to 1.4 cm/yr, suggesting that these wetlands will accumulate about 30 cm of sediments in a little more than two decades.

1. Introduction

Sedimentation and erosion are important processes in wetlands. Sedimentation improves water quality (Hupp and Morris, 1990; Johnston, 1991; Gilliam, 1994; Mitsch and Gosselink, 2015), increases water clarity for improved submersed plant accessibility to sunlight (Nahlik and Mitsch, 2008), and retains nutrients that otherwise cause eutrophication downstream (Mitsch et al., 2001). Sedimentation is also a fundamental process that leads to carbon sequestration in wetlands (Anderson et al., 2006; Bernal and Mitsch, 2012, 2013). Because water velocity decreases dramatically in wetlands compared to streams and rivers, sedimentation occurs to a greater extent in riparian wetlands than in the adjacent river systems (Brueske and Barrett, 1994). This means that both productivity of the wetlands themselves and pollution control of the watershed can be advanced through sedimentation in wetlands. Sedimentation in freshwater wetlands has been investigated by Brueske and Barrett (1994) and Fennessy et al. (1994) at created wetlands at the Des Plaines River Wetlands in northeastern Illinois; Braskerud et al. (2000) in southeast Norway; Hupp et al. (2008) at the Atchafalaya Basin of Louisiana, USA; Kleiss (1996) in eastern Arkansas, USA; Craft and Casey (2000) in Georgia,

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USA; Mann and Wetzel (2000) in Alabama, USA; Darke and Megonigal (2003) at coastal marshes of Virginia, USA; and Sánchez-Carillo et al. (2001) at Las Tablas de Daimiel, Spain.

Accurate estimates of erosion patterns are necessary in wetland sedimentation studies to determine the net sedimentation resulting from fluvial and autogenic processes. The study of erosion; however, has only recently become an accurate science and there are few published papers that include erosion measurements in freshwater wetlands. Erosion increases with exposure to strong winds and wave energy (Scarton et al., 1998) while the presence of vegetation and corresponding tensile root strength can help to prevent erosion (Ward et al., 1984; VanEerdt, 1985; Stevenson et al., 1988). Marsh surface elevation changes and/or sediment erosion have been estimated by Rybczyk et al. (1998) in Louisiana, USA; Pasternack (1998) in Chesapeake Bay USA; Morris et al. (2002) in South Carolina, USA; and Perez-Arlucea et al. (2005) in Spain.

Sedimentation has been measured with several methodologies, including: rare-earth stable tracer horizons like Cs-137 and Pb-210 (Craft and Richardson, 1993; Bernal and Mitsch, 2012), horizon markers (Knaus and Van Gent, 1989; Harter and Mitsch, 2003; Hupp et al., 2008), sediment traps and bottles (Mitsch et al., 1979a; Brueske and Barrett, 1994; Fennessy et al., 1994; Braskerud et al., 2000; Braskerud, 2001; Nahlik and Mitsch, 2008), dendrogeomorphic techniques (Hupp and Morris, 1990; Hupp et al., 1993), and sedimentation plates (Mitsch et al., 1979b; Braskerud et al., 2000; Braskerud, 2001). Variable methodologies have led to inconsistent and poorly comparable sedimentation rates. Insight is needed to translate results using one methodology into information comparable with other studies. An accurate method of estimating net sedimentation is needed for wetlands to quantify the net effect of both gross sedimentation measurements and erosion rates.

Erosion has been studied by very few methods, one of which has been stakes and rods (Harbord, 1949; Pestrong, 1965; Reed, 1989). The accuracy of these experiments has yet to be determined. Day and Boumans (1993) developed the SET (sedimentation-erosion table) to measure erosion, specifically in coastal marshes and shallow sub-tidal areas like wetlands. This method is effective, but in many cases is not economical to implement. None of these methods has yet been applied to sedimentation/erosion studies in freshwater wetlands.

Harter and Mitsch (2003),Anderson and Mitsch (2006), Nahlik and Mitsch (2008), and Bernal and Mitsch (2013) all published estimates of sedimentation in the created wetlands at the Olentangy River Wetland Research Park (ORWRP). The goals of this paper are to integrate these previous sedimentation studies with additional field measurements at the 16- to 17-year-old experimental wetlands at the Olentangy River Wetland Research Park to: (1) compare current rates of gross sedimentation in these wetlands with those taken up to 15 years before; (2) to explore spatial and successional patterns of sediment retention in newly created wetlands; and (3) compare sedimentation methodologies in freshwater riverine wetlands. This paper also presents new estimates of net sedimentation in these wetlands determined by estimating erosion and investigates the importance of animal bioturbation using exclosure fences around horizon markers.

2. Materials and methods

2.1. Site description

Two identical flow-through wetlands were created in 1994 at the Olentangy River Wetland Research Park on the campus of the Ohio State University in the urban center of Columbus, Ohio, USA (Mitsch et al., 1998, 2005, 2012). These freshwater, riparian wetlands receive pumped inflow water from the adjacent Olentangy River. Both wetlands have statistically similar hydraulic loads and both have weir-controlled outflows. The Olentangy River carries runoff from agricultural fields, concentrated feeding operations, wastewater treatment plants, suburban and urban runoff and is subjected to occasional combined sewer overflow events. When first constructed, one wetland was planted with 13 obligate wetland species (referred to as Wetland 1) and the other was left to naturally colonize (Wetland 2) (Mitsch et al., 1998). After having appeared to converge in function after only 3 years, succession in the two wetlands since then has shown the naturally colonizing wetland to be more productive whereas the planted wetland is more diverse (Mitsch et al., 2005, 2012). Several studies suggest that the plantings had little long-term effect on nutrient retention, nitrous oxide emissions, and denitrification (Mitsch and Gosselink, 2015) although recent studies have illustrated some functional differences that remain in the two wetlands (Nahlik and Mitsch, 2010; Mitsch et al., 2012). Methods and results for previous sedimentation studies conducted at the Olentangy River Wetland Research Park are provided in Harter and Mitsch (2003), Anderson and Mitsch (2006), Nahlik and Mitsch (2008), and Bernal and Mitsch (2013).

2.2. Definitions

Various terms used to describe sediment processes were used in this study. Gross sedimentation is defined as the sedimentation rate that includes resuspended and other sediment but does not undergo negative sedimentation caused by erosion or resuspension processes. The term annual sediment accretion is used to describe short-term (1-year) sedimentation in a system, which collects sediment from resuspension and particle settling, and can incur negative sedimentation, but cannot incur net negative sedimentation via erosion. Net sedimentation refers to sedimentation in a system, which is undergoing both positive and negative sedimentation processes and can result in both positive and negative sedimentation rates. Resuspension refers to sediment that is transitive within the wetland, temporarily settling in multiple areas throughout the wetland and never leaving the system. Lastly, erosion refers to sediment that was once temporarily settled on the wetland basin and is carried out of the system. Methods used for measuring each of these fluxes are shown in Fig. 1 and described here.

2.3. Gross sedimentation: sedimentation bottle methods

Sediment bottles were used to estimate gross sedimentation rates in 2004 and 2005, 11 and 12 years after wetland creation and to compare sedimentation during hydrologic pulsing and steady flow conditions imposed on both wetlands in those two years (Nahlik and Mitsch, 2008). At 16 sampling sites in each wetland, two 500 mL wide-mouth Nalgene bottles were attached to a stake and driven into the wetland soil so that the opening was approximately 5 cm above the soil surface (Fennessy et al., 1994). Sediment traps were filled with water and capped for 30 min after deployment to prevent unwanted sediment from being trapped during placement. Sediment traps were sampled once per month (April–July) in hydrologically pulsed (2004) and steady-flow (2005) years. In 2004, mean inflow rates fluctuated every 6–8 days between 52 ± 3 cm day^{-1} and 7 ± 0.2 cm day^{-1}. In 2005, a mean inflow rate of 11 ± 0.01 cm day^{-1} was constant throughout the sampling period. A larger percent (63%) of sites were located nearer to the inflow than outflow. Sediments were dried and weighed as described by Nahlik and Mitsch (2008).

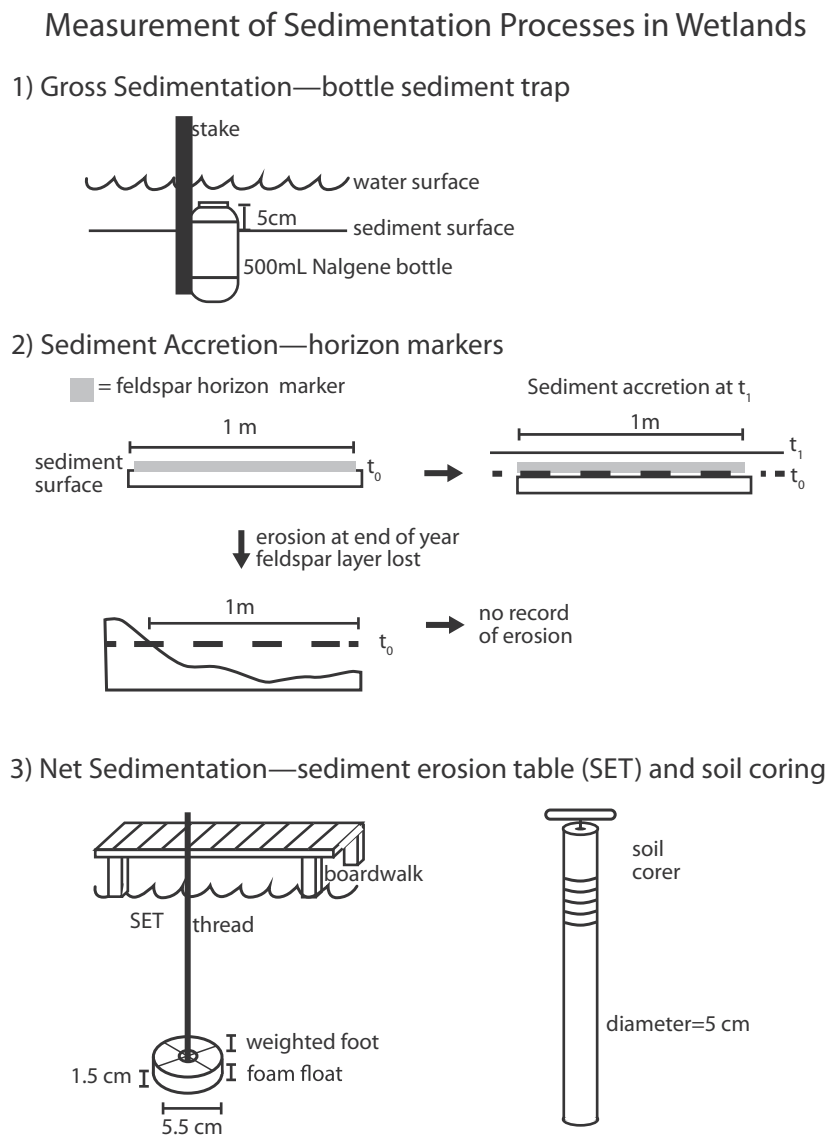

Fig. 1. Schematic of sedimentation sampling methods used at the freshwater marshes at the Olentangy River Wetland Research Park, 1996–2010.

2.4. Annual sediment accretion: horizon marker method

Annual sediment accretion was measured using horizon markers at 16 sampling sites in each of the two 1 ha experimental wetlands in 1996–1997 (Harter and Mitsch, 2003) and repeated in 2009–2010 (this study). Permanent sedimentation marker sites were based on accessibility from boardwalk, water depth, and distance from inflow (Fig. 2). For the current study, a 1 cm layer of feldspar with a 1 m^2 area was applied to the same 16 sampling sites in each of the two wetlands on March 19, 2009. They were sampled approximately one year later (May 5, 2010). Samples were taken from each marker plot with a cryogenic coring device (Knaus and Cahoon, 1990) as was done in the 1996–1997 study. Four or more cores per site were taken, one from each quadrant of the m^2 area. When a feldspar marker horizon was observed, the accumulated sediment was measured using a micrometer, cut from the core, and placed in a tin baking tray for nutrient analysis. Additional cores were collected from each site, cut to a 5–10 cm length, measured for width to calculate volume, and placed in a tin tray for bulk density analysis. At least two cores per site were dried at 60 °C for 48 h and weighed to calculate bulk density, then averaged per site. Sedimentation rates ($kg\,m^{-2}\,yr^{-1}$) were estimated from accumulation rates and bulk density. As soil cores were taken, any remnant marker layer laid in 1996 by Harter and Mitsch (2003) were noted and measured for length.

2.5. Erosion and net sedimentation: sedimentation/erosion table (SET) and soil core method

Sediment erosion was estimated with sedimentation-erosion tables (SETs) similar to those used by Day and Boumans (1993) in the current study (2009–2010). Sampling sites were located along the main entry boardwalks at 6 locations in each wetland (Fig. 2). This part of the study assumes that the boardwalks, constructed in 1995–1996 to a depth of 1.0 m into the soil, are stable and no longer shift with loading or ice. The erosion-measuring device consisted of a 1.5 m-long thread attached to a foam float and weighted foot that gently settle to the wetland floor. The float is designed to reduce the compression of sediments caused when the weighted foot reaches the bottom. Lowering the weighted foot to the wetland floor and measuring the length of thread from a benchmark on the boardwalk to the sediments can quantify the extent of sedimentation or erosion. Multiple measurements of different known depths were used to determine an SET accuracy of

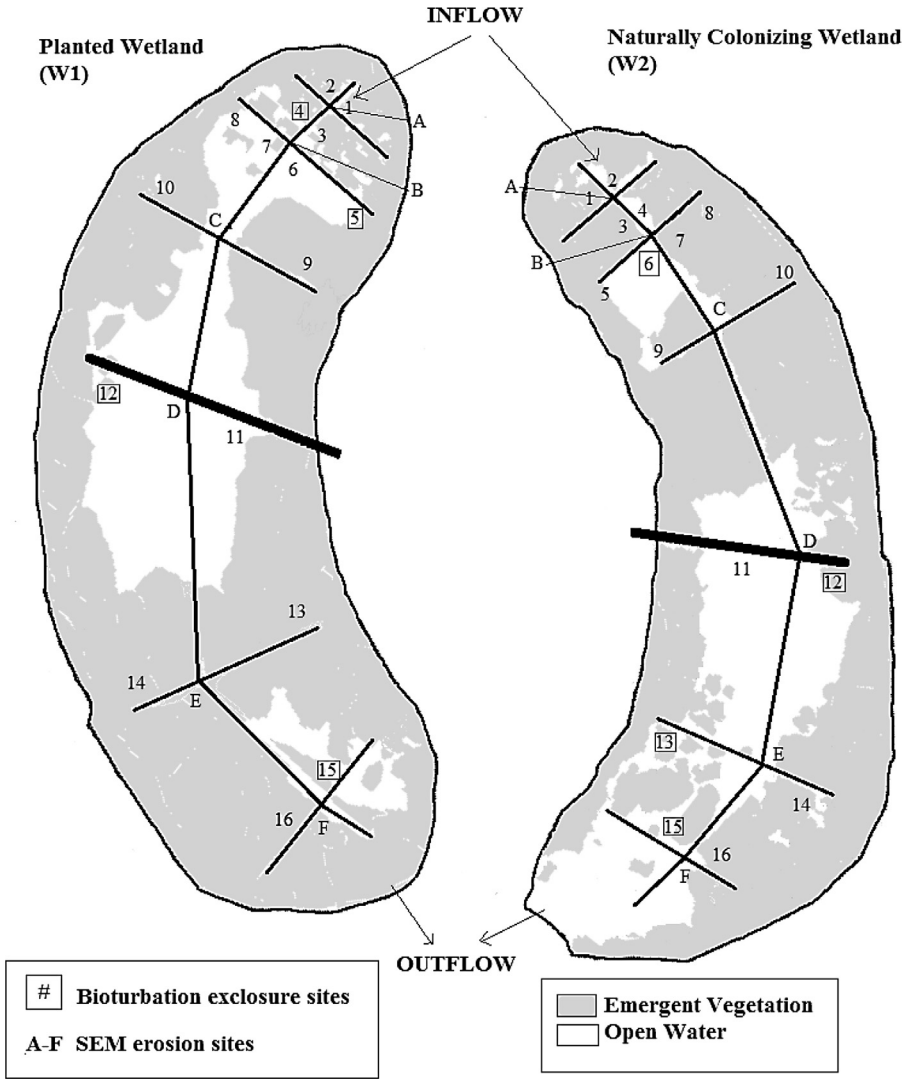

Fig. 2. The two 1 ha created wetlands at the Olentangy River Wetland Research Park showing sampling sites for 2009–2010: sampling sites numbered 1–16 are feldspar horizon marker sites; sites with rectangles were fitted with bioturbation exclosures; sites A–F are sedimentation/erosion method (SEM) for net sedimentation estimates. Areas with emergent vegetation in 2009 are shaded.

0.39 ± 0.09 cm. Erosion was measured four times during the study (May 1, July 27, October 22, 2009 and March 4, 2010).

Net sedimentation by extraction of sediment cores using an identical 10 m grid spatial sampling pattern were carried out in 2004 (year 11 or 10 years after wetland creation) by Anderson and Mitsch (2006) and in 2009 (year 16 or 15 years after wetland creation) by Bernal and Mitsch (2013). An even distribution of samples was collected as a function of distance from inflow, water depth, and vegetative cover. The sediment cores were 7 cm in diameter and measured the accumulated sediment overlying nonhydric soil layer (10–35 cm depth).

2.6. Bioturbation exclosures

Four of the feldspar horizon marker sites in each wetland were chosen for bioturbation exclosures during 2009–2010. The exclosures prevented fish, waterfowl, and mammals from disturbing the sedimentation process so that the impact of bioturbation on sedimentation could be quantified. Each exclosed site was compared with the nearest adjacent site as a control. For example, site 15 was compared with site 16 and site 11 with 12 (Fig. 2). Exclosures were made of 1 in. plastic mesh fencing wrapped around PVC pipes over the 1 m^2 horizon-marked areas to a height of 50 cm. Fencing was attached using zip-ties and PVC pipes penetrated 30 cm into the ground. Muskrat and animal activity was monitored throughout the study period so sites that incurred significant animal damage could be dropped from the study.

2.7. Data analysis

For the 2009–2010 sampling, all sedimentation and erosion data met the conditions of normality. A two-sample *t*-test assuming equal variance was used to determine significance of unequal sample sizes (planted and unplanted wetland feldspar sites). Any comparison with $p < 0.05$ was assumed significantly different. Net sedimentation estimates for the SEM were calculated by subtracting the starting elevation from the final elevation recorded in March. ANOVA was used to determine seasonal differences in SEM data and differences between vegetative ranks. Data collected for spatial and biological variables were normalized using the standard score method (Sterne and Kirkwood, 2003). Relationships between data sets were analyzed using Pearson correlation analysis and were further compared using a stepwise

regression. All analyses were performed in Minitab 15 for Windows (2007).

2.8. Sediment budget

The following equation was used to estimate sediment flux at the inflow and outflow of the wetlands.

$$F = \frac{S \times T}{A} \quad (1)$$

where F = sediment flux ($g\,m^{-2}\,day^{-1}$). A = area of wetland (m^2). S = inflow or outflow ($m^3\,day^{-1}$). T = total suspended solids, TSS ($g\,m^{-3}$).

Inflow rates were recorded daily for planted and naturally colonizing wetlands and adjusted periodically to the Olentangy River water level staff gage according to a predetermined scale. Outflow rates were calculated using a v-notch weir and outflow water level staff gage according to newly modified equations similar to those recorded for Wang and Mitsch (1999). Water samples were analyzed for turbidity twice per week during March 2009–May 2010 with a HACH 2100N Turbidimeter (Eaton et al., 2005) and total suspended solids (TSS) were estimated for turbidity (Harter and Mitsch, 2003). Daily values given by Eq. (1) were added to give total sediment influx and outflow from March 19, 2009 to May 5, 2010 for the two wetlands.

3. Results and discussion

3.1. Gross and annual sediment accretion

Of the three studies conducted to estimate gross and annual sediment accretion, an approximate range of annual gross sedimentation of 30–90 $kg\,m^{-2}\,yr^{-1}$ was observed in these created wetlands by the three separate studies (Table 1) (Harter and Mitsch, 2003; Nahlik and Mitsch, 2008; this study). Annual accretion rates observed using the feldspar horizon marker method resulted in $31.7 \pm 4.4\,kg\,m^{-2}\,yr^{-1}$ ($4.2 \pm 0.6\,cm\,yr^{-1}$) in 1996 and $34.4 \pm 4.5\,kg\,m^{-2}\,yr^{-1}$ ($5.5 \pm 0.8\,cm\,yr^{-1}$) in 2009, with a range of 13.6–63.8 $kg\,m^{-2}\,yr^{-1}$ (1.82–8.6 $cm\,yr^{-1}$) and 14.1–74.5 $kg\,m^{-2}\,yr^{-1}$ (2.1–10.8 $cm\,yr^{-1}$), respectively (Harter and Mitsch, 2003). There has been no significant change in annual sediment accretion over the 13 years from 1996 to 2009. These rates of sedimentation are comparable to those measured by others with feldspar horizon markers (DeLaune et al., 1983; Cahoon and Turner, 1989) and within the expected range for freshwater riverine wetlands as suggested by Pasternack (1998), Sánchez-Carillo et al. (2001), and Darke and Megonigal (2003). Gross sedimentation rates were observed in 2004–2005 (64–90 $kg\,m^{-2}\,yr^{-1}$) using the sediment bottle method (Nahlik and Mitsch, 2008). For all three studies, no significant differences in sedimentation rate were observed between the two wetlands (planted and naturally colonizing). This suggests that planting of created freshwater wetlands does not have a long-term effect on sedimentation rate.

In 2009, one feldspar horizon was detected from the 1996 study as a depth of 18.4 cm, yielding a net accumulation rate of 1.4 $cm\,yr^{-1}$ over the 13-year period, 1996–2009. The accumulation rate given from this 1996 feldspar layer is a better approximation of net sedimentation than gross sedimentation because that marker underwent 13 years of counteracting sedimentation and erosion processes.

3.2. Erosion

The average sedimentation rate for the two created wetlands as measured by the sedimentation/erosion table (SET) method in 2009–2010 was 5.0 $cm\,yr^{-1}$ with a range from $-3.4\,cm\,yr^{-1}$ to 17.2 $cm\,yr^{-1}$ (Table 2). Sites A and E in the planted wetland (W1) and site F in the naturally colonizing wetland (W2) were the only three sites to undergo net erosion after a year of study. Seasonally, however, 83% of sites eroded at some time during the year. Sediment elevation changed dramatically throughout the year with an average fluctuation of $\pm 6.41\,cm\,yr^{-1}$ for each site, with the greatest percent change occurring from May to July (47%). The average fluctuation rate is higher than the net sedimentation rate, which indicates that sediment undergoes significant resuspension.

Net sedimentation rates from May to July (early growing season) were higher ($p = 0.0003$) than those measured from July to October (late growing season) (Fig. 3). In the late growing season, erosion in the naturally colonizing wetland was 0.01 ± 0.02 cm day^{-1}, while sedimentation of 0.06 ± 0.02 cm day^{-1} occurred in the early growing season. Similarly, the planted wetland experienced net erosion in the late growing season of 0.005 ± 0.02 cm day^{-1} with net sedimentation of 0.04 ± 0.02 cm day^{-1} in the early growing season. This seasonal pattern is most likely due to differences in hydrologic inflow. During May–July the average inflow for both experimental wetlands was 1206 $m^3\,day^{-1}$ with 25 flood pulse events (>1640 $m^3\,day^{-1}$) while from July to October there were only 13 flood pulse events and an average inflow of 1136 $m^3\,day^{-1}$. Previous studies have also shown a correlation between hydraulic loading and sedimentation (Brueske and Barrett, 1994; Nahlik and Mitsch, 2008). Likewise, average total suspended solids (TSS) at the wetland inflows were higher in the period from May to July (22 $g\,m^{-3}$) than during July–October (17 $g\,m^{-3}$). When water is deprived of sediment, there is more kinetic energy available to erode the wetland surface (Ward and Trimble, 2004). Decreased hydraulic loading and decreased turbidity during the late summer dry season were sufficient to change the wetland from a net sink to a net source of sediment for the three month period. Cahoon et al. (1995) found that in addition to hydraulic loading and turbidity, plant production and decomposition processes may strongly influence marsh surface elevation.

Table 1
Methodologies and sedimentation rates for five studies spanning 15 years of research.

Method	Authors	Sedimentation ($kg\,m^{-2}\,yr^{-1}$)
Gross sedimentation		
Sediment bottle method	Nahlik and Mitsch (2008)	64–90
Annual sediment accretion		
Feldspar horizon marker method	Harter and Mitsch (2003)	31.7 ± 4.4
	Current study	34.4 ± 4.5
Net sedimentation		
10–15 year soil accumulation	Anderson and Mitsch (2006)	4.7 ± 0.3
	Bernal and Mitsch (2013)	6.0 ± 0.4
SET and 1 year horizon marker	Current study	3.9 ± 6.1

Table 2
Annual sediment accretion and net sedimentation estimated for two experimental wetlands at Olentangy River Wetland Research Park, March 2009–May 2010.

Planted wetland (W1)					Naturally colonizing wetland (W2)				
Gross sedimentation (Feldspar horizon method)									
Site	Study period (days)	Accretion ($cm\,yr^{-1}$)	Bulk density ($g\,cm^{-3}$)	Mass accumulation ($kg\,m^{-2}\,yr^{-1}$)	Site	Study period (days)	Accretion ($cm\,yr^{-1}$)	Bulk density ($g\,cm^{-3}$)	Mass accumulation ($kg\,m^{-2}\,yr^{-1}$)
1–3	385	9.6	0.59	56.6	2–3	408	2.1	0.67	14.1
1–6	384	3.9	0.81	31.6	2–6	396	5.7	0.69	39.3
1–7	385	8.9	0.50	44.5	2–7	396	4.1	0.64	26.2
1–8	385	10.8	0.69	74.5					
1–11	374	7.3	0.37	27.0	2–11	388	7.5	0.37	27.8
1–12	377	4.3	0.76	32.7					
1–14	377	2.2	0.97	21.3					
1–15	377	7.7	0.79	60.8					
1–16	378	4.5	0.61	27.5	2–16	392	3.2	0.85	27.2
Avg. ± std err		6.6 ± 1.0	0.68 ± 0.1	41.8 ± 5.8			4.5 ± 1.0	0.64 ± 0.08	26.9 ± 4.0
Net sedimentation (Sedimentation erosion method)[a]									
1-A	305	−0.1		−0.68	2-A	305	17.2		110.1
1-B	305	10.5		71.4	2-B	305	3.4		21.8
1-C	305	6.5		44.2	2-C	305	3.2		20.5
1-D	305	4.9		33.3	2-D	305	8.2		52.5
1-E	305	−1.6		−10.9	2-E	305	3.5		22.4
1-F	305	6.6		44.9	2-F	305	−3.4		−21.8
Avg. ± std err		4.5 ± 1.9		30.4 ± 12.6			5.4 ± 2.8		22.4 ± 18.0

[a] Average bulk density from respective wetland was used.

Differences in root aggregation and volume in late spring to late summer may also have impacted elevation change in this study.

While the SET method did show that erosion occurred in the wetlands throughout the year, the average rate of 5.0 $cm\,yr^{-1}$ is probably too high to be considered an accurate measurement of net elevation change. Elevation change in other freshwater, riparian wetlands in the United States has varied between 0.07 and 0.57 $cm\,yr^{-1}$ (Hupp et al., 1993). A possible reason for the overestimation may be that the sites were not spread equidistantly throughout the wetland. As seen in Fig. 2, three of the sites (A, B, and C) were placed toward the inflow while only two of the sites (E and F) were placed toward the outflow. Also, all of the SEM sites were located in the deepest area of the wetland basins. Because sedimentation rates have been shown to be higher toward the inflow and in deep water sites in these wetlands, this could lead to a disproportionately high sedimentation rate. Future applications of this method should use more than 6 plots per hectare and sample more frequently throughout the year to confirm seasonal shift in sedimentation/erosion patterns and to determine spatial patterns of erosion.

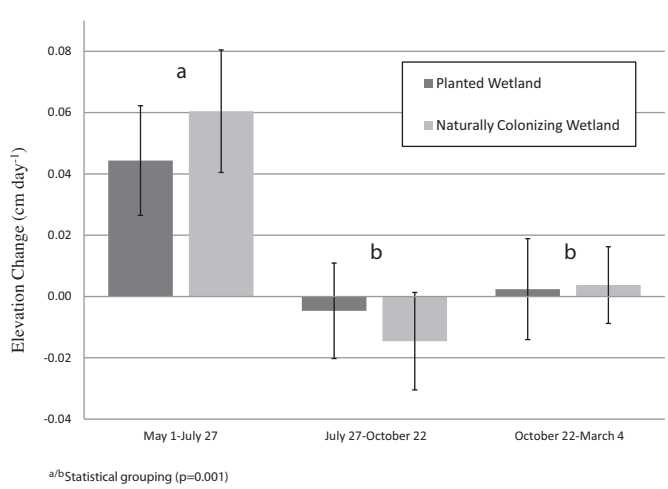

Fig. 3. Seasonal sedimentation/erosion patterns estimated by the Sedimentation Erosion Table (SET) for two created wetlands at the Olentangy River Wetland Research Park.

3.3. Net sedimentation

A total of three net sedimentation estimates over the life of these created wetlands ranged from 3.9 to 6.0 $kg\,m^{-2}\,yr^{-1}$ (Table 1). Two of the studies measured sediment depths above the antecedent soil layer using soil coring techniques, resulting in a rate of $4.7 \pm 0.3\,kg\,m^{-2}\,yr^{-1}$ ($0.9 \pm 0.07\,cm\,yr^{-1}$) in 2004 and $6.0 \pm 0.4\,kg\,m^{-2}\,yr^{-1}$ ($0.9 \pm 0.06\,cm\,yr^{-1}$) in 2009 (Anderson and Mitsch, 2006; Bernal and Mitsch, 2013). The increase is largely due to a significant increase in soil bulk density of $0.51 \pm 0.02\,g\,cm^{-3}$ in 2004 to $0.67 \pm 0.04\,g\,m^{-3}$ in 2009 ($p = 0.049$) over those 5 years. The increase could be due to increased rhizosphere development and coagulation in the soil, characteristic of maturing created wetlands.

The most accurate way of estimating net sedimentation may be with the accretion results from cores that showed the 1996 feldspar layer in 2010. Unfortunately only 1 horizon marker site showed this layer in 2010 with an 18.4 cm depth or accumulation rate of 1.4 $cm\,yr^{-1}$ (9.0 $kg\,m^{-2}\,yr^{-1}$). This core was recovered from the naturally colonizing wetland at a heavily vegetated site near the inflow (site 7). Low recovery of the 1996 feldspar horizon may be due to its destruction from previous sampling, erosion processes, or bioturbation by burrowing mammals.

Anderson and Mitsch (2006) estimated an average net mass accumulation over the first 10 years of these experimental wetlands at 4.7 $kg\,m^{-2}\,yr^{-1}$. Since our new estimate is double that measurement and is only supported by one core, we estimated current net sedimentation by interpolating data from the SET and feldspar horizon marker methods. Many areas in the wetlands underwent erosion throughout the year, removing the feldspar marker. We conducted the interpolation by assuming these lost feldspar sites underwent erosion, thereby using the negative sedimentation rates collected by the SET method (wetland sites 1A,

1B, and 2E) and determining a new average (McCarthy, 2011). The average net sedimentation rate was then estimated to be 1.2 cm yr^{-1} (3.4 kg m^{-2} yr^{-1}), which is surprisingly similar to previous net sedimentation measurements. We therefore concluded the missing feldspar sites likely underwent erosion, and that a range of 1.2–1.4 cm yr^{-1} of net sediment accumulation is the most reasonable estimate for these created flow-through riverine wetlands.

3.4. Methodology comparison

The highest rates of sedimentation were observed when using sediment bottles (Nahlik and Mitsch, 2008). This is likely due to the fact that sediment traps collect sediment but prevent erosion and resuspension of soil particles. For this reason, sediment traps may be useful for distinguishing rates of particle settling from other sediment processes, such as resuspension and erosion. Sediment traps should not be used to estimate long-term rates of sedimentation.

Similarly, the horizon marker method does not account for elevation change when the horizon marker is eroded, and therefore does not account for erosion processes, but does account for resuspension of soil particles. In this study, and in Harter and Mitsch (2003), nearly 30% of plots lost feldspar horizons. These horizon markers may have been lost due to erosion, net negative resuspension processes, bioturbation by organisms, or mixing of the feldspar layer with darker organic layers (Cahoon and Turner, 1989); therefore, it can be assumed that horizon markers are not appropriate for quantifying these sediment dynamics.

The sediment bottle method appears to overestimate sedimentation in freshwater wetlands to a greater extent than do horizon markers due to permanent entrapment of particles. Young and Rhoades (1971) suggested that only 0.5–2% of sediments deposited in a sediment trap contribute to long-term accumulation. Although more reasonable rates of gross sedimentation were found at the Des Plaines River Wetlands in northeast Illinois using the bottle method, ranging from 1.2 to 12.8 kg m^{-2} yr^{-1}, erosion is an important component of sediment dynamics not accounted for in trap and horizon marker studies (Fennessy et al., 1994; Harter and Mitsch, 2003).

Net sedimentation rates, which do account for erosion and resuspension processes, can be estimated by using the parent soil layer, radiometric dating using ^{137}Cs, and sedimentation erosion tables (SETs). These methodologies are therefore more appropriate for predicting net sediment accretion in wetlands.

3.5. Bioturbation

Our 2009–2010 study found that bioturbation had a significant influence on sedimentation rates. Of the sites with bioturbation exclosures installed, 50% of the exclosures were damaged by apparent animal activity throughout the year. Of the 4 sites that remained unaltered, only 2 of the sites showed feldspar horizon markers. Exclosed sites experienced approximately twice the amount of (49%) sedimentation (46.4 ± 14.1 kg m^{-2} yr^{-1}) as did the open sites (24.0 ± 2.9 kg m^{-2} yr^{-1}). In several sites with bioturbation exclosures, large algae mats formed during May through October, 2009. It is unclear whether the algae mats were artifacts of the exclosures or are attributed to decreased bioturbation of macrofauna. The netting of the bioturbation exclosures may have stabilized algal mats preventing movement toward the outflow. It is likely the higher rates of sedimentation found in exclosed sites could be due to higher algal settling; however, organic carbon content was not significantly different between exclosed and open sites. Differences in organic carbon content may have been observed between sites if soil analysis had not excluded partially decomposed plant matter, which may have included recently deposited algae. The lack of herbivory and dispersal processes may contribute to higher sedimentation rates as waterfowl and large fish have been shown to be important in dispersal and break up of large periphyton mats (Vanni, 2002; Luckeydoo et al., 2002). Without herbivory and dispersal processes, plant matter settles as organic matter to form sediment at the end of the growing season. Further studies of bioturbation in wetlands are needed at different spatial scales to determine biological impacts on sedimentation processes.

3.6. Spatial patterns

In comparing inflow and outflow sites using both the SEM and feldspar data (comparative study), the inflow sites had a higher average sedimentation rate (43.0 ± 11.7 kg m^{-2} yr^{-1}) than did the outflow sites (21.4 ± 9.5 kg m^{-2} yr^{-1}). This difference was not statistically significant, although previous data collected at the wetlands did show a statistically significant difference (44.0 ± 4.1 kg m^{-2} yr^{-1} vs. 30.3 ± 4.5 kg m^{-2} yr^{-1}, $p = 0.04$) (Harter and Mitsch, 2003). For the comparative study, other physiochemical or spatial impacts may be more important.

One possible spatial impact was basin depth. Deepwater sites (21–25 cm) had significantly higher ($p = 0.005$) sedimentation rates (40.4 ± 5.3 kg m^{-2} yr^{-1}) than did shallow water sites (9–17 cm) (8.2 ± 6.7 kg m^{-2} yr^{-1}). A similar difference between shallow and deep water wetland sites was also by Brueske and Barrett (1994), Fennessy et al. (1994), and Meeker (1996). Several sedimentation studies conducted at these wetlands found higher sedimentation rates in deep, open water areas than shallow areas with emergent vegetation (Harter and Mitsch, 2003; Anderson and Mitsch, 2006; Bernal and Mitsch, 2013) (Fig. 4). Nahlik and Mitsch (2008) found the inverse to be true, suggesting that the sediment trap method may be a less effective method at determining spatial sedimentation patterns in freshwater wetlands.

With many processes and spatial patterns affecting sedimentation in the wetlands, it is difficult to determine which processes are the most influential and how interrelating processes affect sedimentation rates. When the effects of distance from inflow and water depth on sedimentation are compared as separate and combined variables, there was a stronger correlation for water depth and distance from inflow as combined variables than as separate ($R^2 = 0.261$ vs. $R^2 = 0.1275$).

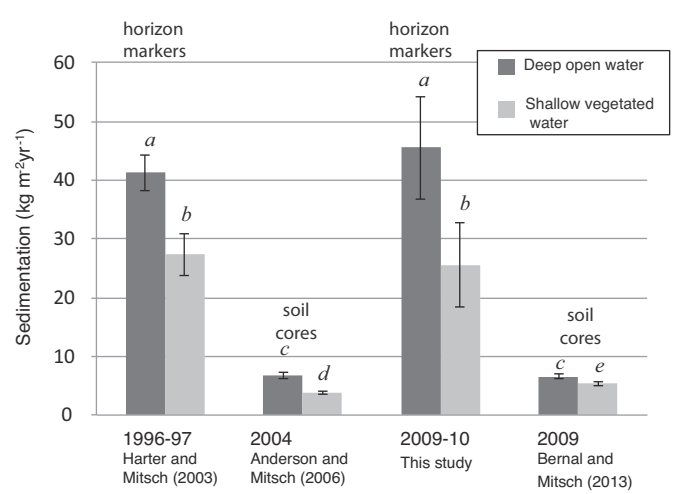

Fig. 4. Comparison of sedimentation rates measured with two methods (horizon markers and soil cores) in 4 independent studies in deep open water areas to shallow areas with emergent vegetation in two 1-ha created wetlands at the Olentangy River Wetland Research Park. Date of sampling is given along with citation for study.

Table 3
An annual inflow and outflow sediment budget estimated for planted and naturally colonizing wetlands at the Olentangy River Wetland Research Park from site data and from sedimentation measurements over March 2009–May 2010.

Parameter	Planted wetland (W1)	Naturally colonizing wetland (W2)
Inflow		
Flow[b], $m^3\,day^{-1}$	1153 ± 22 (413)	1166 ± 23 (413)
Turbidity[b], NTU	22.4 ± 1.8 (158)	22.4 ± 1.8 (158)
TSS[b], $g\,m^{-3}$	9.4 ± 1.3 (158)	9.4 ± 1.3 (158)
TSS[b], $g\,m^{-3}$ (all days)	13.9 ± 1.5 (413)	13.9 ± 1.5 (413)
TSS[b], $g\,m^{-3}$ (weighted average)	15.8 ± 2.3 (413)	15.8 ± 2.3 (413)
Total sediment flux[b], $g\,m^{-2}\,yr^{-1}$	665 ± 0 (413)	673 ± 0 (413)
Outflow		
Flow[a,b], $m^3\,day^{-1}$	822 ± 43 (161)	966 ± 42 (161)
Turbidity[b], NTU	20.4 ± 1.7 (161)	18.4 ± 1.5 (161)
TSS[b], $g\,m^{-3}$	7.5 ± 1.0 (161)	6.1 ± 0.3 (161)
TSS[b], $g\,m^{-3}$ (weighted average)	7.7 ± 1.0 (161)	7.2 ± 0 (161)
Total sediment flux[b], $g\,m^{-2}\,yr^{-1}$	231 ± 0 (161)	254 ± 0 (161)
Sediment retention, %	65	62
Sediment retention, $g\,m^{-2}\,yr^{-1}$	434	419

Data are averages ± standard error (# data points).
[a] Significant difference for planted and naturally colonizing wetlands.
[b] Significant difference for inflow and outflow.

3.7. Sediment budgets

Inflow–outflow sediment budgets for 2009–2010 (years 16 and 17) are summarized in Table 3. The average flow into the wetlands was 1160 $m^3\,day^{-1}$ and the outflow was 890 $m^3\,day^{-1}$. Average turbidity, which was sampled weekly, was statistically higher at the inflow (22 NTU) than the outflow (19 NTU) ($p = 0.005$). Total suspended solids were calculated using equations established in Harter and Mitsch (2003), to give an average of 14 $g\,m^{-3}$ at the inflow and 7 $g\,m^{-3}$ at the outflow. Weighted annual fluxes for the two wetlands calculated by normalizing for a yearly rate and by totaling monthly averages are 669 and 243 $g\,m^{-2}\,yr^{-1}$ for the inflow and outflow respectively, giving an average sediment retention rate of 427 $g\,m^{-2}\,yr^{-1}$ or 64% during this study period. This retention rate is within the range of past findings for nutrient retention at the Olentangy River Wetlands and similar wetlands (Fennessy et al., 1994; Mitsch et al., 2005).

An overall sediment budget using the net sedimentation range of 3400–9000 $g\,m^{-2}\,yr^{-1}$ and average sediment retention rate of 427 $g\,m^{-2}\,yr^{-1}$ is shown in Fig. 5. The additional sediment load not attributed to sediment influx could be due to missing flow data and/or to the additional organic matter attributed by decomposing plants. It is roughly estimated that nearly 800 $g\,m^{-2}\,yr^{-1}$ more organic carbon is found in wetland sediments than is accounted for by inflow of organic carbon. This indicates that photosynthesized plant material contributes greatly to sedimentation in wetlands.

In Harter and Mitsch (2003), approximately 700 $g\,m^{-2}\,yr^{-1}$ of sediment was trapped in the wetlands in 1997 with a retention rate of 41%. This supports the argument that wetlands become more productive at trapping sediments as they age and develop more efficient basin geomorphology (Braskerud, 2001). Harter and Mitsch (2003) estimated a large amount of sediment-resuspension on the order of 36,000 $g\,m^{-2}\,yr^{-1}$. Although, it was not our goal to predict resuspension, because the average change in sediment elevation using the SET (±6.41 $cm\,yr^{-1}$) was higher than the SET net sedimentation rate (5.0 $cm\,yr^{-1}$), resuspension obviously continues to be an important, constantly on-going process in the two created wetlands. This resuspension often lead to an erratic range of sedimentation rates.

Average sedimentation rates appear to have slightly increased since wetland creation, although the trend is not significant. The lack of a significant trend is more due to spatial variability, which creates a mosaic of outliers in sedimentation data. Average sedimentation rates for replicated studies (annual sediment accretion estimates using feldspar markers and net estimates using sediment cores), do not show any relationship with net primary productivity in these wetland basins (Fig. 6), suggesting that much of the sedimentation is due to inflowing river water and not to autochthonous productivity. Waletzko and Mitsch (unpublished manuscript) found through isotope analysis that the fine particulate organic matter discharging from these wetlands was closely linked to the productivity of the macrophytes in the wetlands. Some of this FPOM could contribute to the accumulation of organic sediments in the wetlands. Coarse particulate organic matter (CPOM) export from the wetlands, on the other hand, was a surprisingly low rate, suggesting, as found by Anderson and Mitsch (2006) and Bernal and Mitsch (2013) that there is surprisingly high organic carbon accumulation in the sediments of these wetlands.

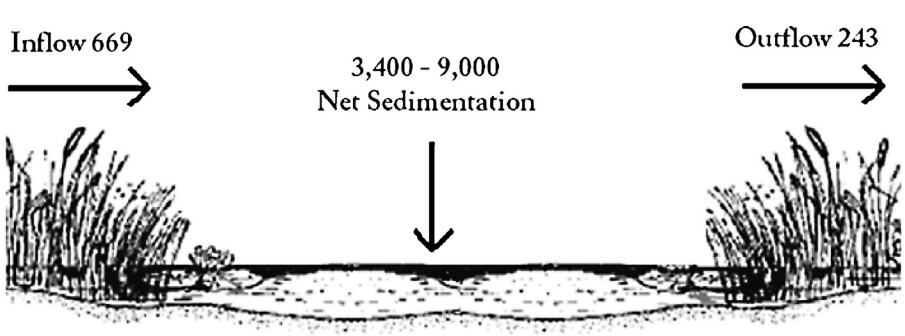

Fig. 5. Most recent sedimentation budget estimating inflow and outflow sediment flux and net sedimentation rates ($g\,m^{-2}\,yr^{-1}$) for the 2009–2010 study period. Figure modified from Harter and Mitsch (2003).

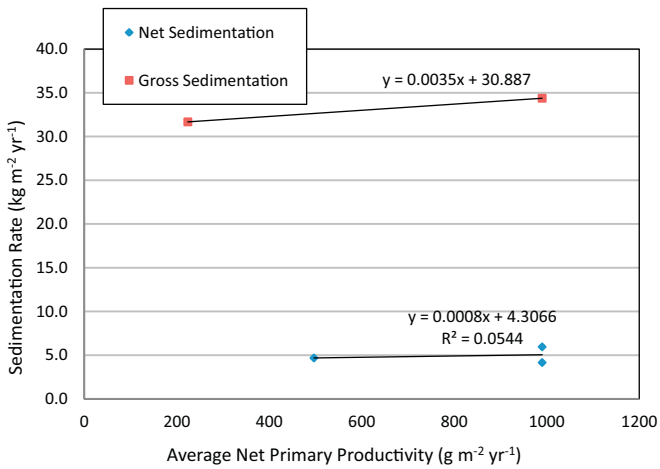

Fig. 6. Average sedimentation rates for two paired studies versus net primary productivity (NPP) of the experimental wetlands: net sedimentation by Anderson and Mitsch (2006) and Bernal and Mitsch (2013); and annual sediment accretion by Harter and Mitsch (2003) and this study. Most NPP data are from Mitsch et al. (2012).

4. Conclusions

1. Erosion in temperate freshwater wetlands is higher in the late rather than the early growing season due to higher sediment loads from hydrologic conditions in the spring and high kinetically charged water flow in the summer.
2. Erosion is ubiquitous in created, freshwater wetlands, occurring frequently within varied spatial and temporal dimensions and despite positive annual rates of net sedimentation.
3. Bioturbation by wetland mammals and waterfowl in freshwater flow-through wetlands appears to decrease sedimentation rates.
4. Deep, open water areas have higher rates of sedimentation than shallow areas with emergent vegetation in flow-through created wetlands. However, methodology including sediment traps may produce different results.
5. On average, sedimentation rates are higher closer to the inflow and decrease toward the outflow in flow-through wetlands.
6. These created wetlands are probably accumulating an overall average of 1.2–1.4 cm/yr of sediments. This suggests that the lifetime of these wetlands may be on the order of several decades with 30 cm net accumulation in a little more than 2 decades and perhaps 60 cm net accumulation in less than 50 years. Those accumulations will become less if the wetlands export more sediments as the water becomes shallower.
7. Many current methods used to study sedimentation in cwetlands overestimate long-term, net sediment accumulation because they do not account for erosive processes. These methods, including sediment traps and short term horizon markers, are more appropriate measures of particle settling and short-term sediment accretion, respectively.
8. No significant difference in sediment accretion rate was found for planted and naturally colonizing wetlands, suggesting that planting of created wetlands has no long term effect on overall sediment accumulation in riverine, freshwater, wetlands.

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