Ecological Engineering 63 (2014) 27–33

Contents lists available at ScienceDirect

Ecological Engineering

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

Effects of sediment load on the seed bank and vegetation of *Calamagrostis angustifolia* wetland community in the National Natural Wetland Reserve of Lake Xingkai, China

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ARTICLE INFO

Article history:
Received 20 March 2013
Received in revised form 21 October 2013
Accepted 4 December 2013
Available online 29 December 2013

Keywords:
Irrigation return runoff
Marsh
Sediment
Seed bank
Seed germination
Vegetation

ABSTRACT

Sedimentation in wetlands due to agricultural irrigation runoff is a significant threat to the conservation of natural areas. Sediment accumulation rates in a wetland adjacent to a paddy field were much higher than in a natural wetland within Lake Xingkai, Northeast of China (0.3–1.0 cm yr^{-1} vs. 0.03–0.2 cm yr^{-1}, respectively), which may create negative ecological impacts to the wetland system, particularly the vegetation community. We conducted a germination experiment in the greenhouse to evaluate the effects of different sediment loads on the seed bank of *Calamagrostis angustifolia* wetland community under two hydrological regimes (0 and 10 cm water depth), and a vegetation survey in the natural and sediment disturbed sites to investigate the changes of vegetation community with high sediment accumulation. Results revealed significant effects of sediment load on the germination rates in the greenhouse. Species richness and seedling emergence decreased significantly with the addition of 0.5 cm of sediment, and species responded differently to the addition of sediment. The number of seedlings of species such as *C. angustifolia* and *Typha orientalis* decreased significantly with 0.25 cm sediment addition level. *Eleocharis mamillata* and *Galium trifidum* decreased significantly with the addition of 0.5 cm sediment levels, while *Sagittaria trifolia*, *Alisma orientale* and *Salix* spp. germinated with the addition of 1 and 2 cm sediment levels. The number of species that germinated in non-flooded conditions was significantly higher than in flooded conditions. Vegetation survey showed that the number of species present in the natural wetland was higher than that in wetland adjacent to the paddy field (species richness: 20 vs. 11, respectively), some native species including two annuals (*Polygonum hydropiper* and *Pycreus sanguinolentus*) disappeared from the disturbed site. To protect and restore the wetland vegetation community in the Sanjiang Plain, irrigation and watershed management strategies designed to reduce sediment inputs into wetlands may aid in the conservation of these natural wetlands.

1. Introduction

Wetlands provide significant environmental functions and play an important role in controlling soil erosion and alleviating pollution. Wetlands within agricultural landscapes are increasingly receiving higher sediment inputs (Piao and Wang, 2011). In many cases, wetlands are purposely used to improve water quality by removing sediments and pollution from rivers and irrigation return flow or runoff (Zhao et al., 2009; Piao and Wang, 2011). Increasing rates of sedimentation in the past 25–50 years (e.g., Johnston et al., 1984; Hupp and Bazemore, 1993; Piao and Wang, 2011), often as a result of increasing deforestation, urbanization and land reclamation, may push natural wetlands past a tolerance limit, which could have increasingly negative impacts on the integrity of these wetlands. Most research indicates that high amounts of sediment deposition in wetlands may influence the soil properties and topography, inhibit seed germination and seedling growth and lead to the changes in vegetation composition (Ewing, 1996; Luo et al., 1997; Werner and Zedler, 2002; Koning, 2004; Lowe et al., 2010; Wang et al., 2013).

In freshwater wetlands, sediment accumulation could impact species richness and overall abundance through inhibiting seed

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http://dx.doi.org/10.1016/j.ecoleng.2013.12.004

germination and smothering plants (Dittmar and Neely, 1999; Werner and Zedler, 2002; Wang et al., 2013). The specific response depends on the plant species, hydrology, depth of burial and the mass of the seeds (Leck, 1996). Seed germination cues, such as light, oxygen availability, soil temperature, and moisture, are altered by sediment deposition, which may result in lower germination rate (Peterson and Baldwin, 2004). In a freshwater wetland in central Iowa, USA, sediment loads as low as 0.25 cm significantly reduced the species richness and density recruited from the wetland seed bank. The addition of sediment decreased the density of individuals germinating for most species (Jurik et al., 1994). Survival of vegetative tubers of *Vallisneria americana*, a submersed aquatic plant, declined 90% or more when buried in 10 cm of sediment and the species could not survive if buried under more than 25 cm of sediment, while *Typha* sp. was greatly affected by 0.25 cm of sediment (Rybicki and Carter, 1986).

Increased sediment deposition can affect microtopography and soil properties such as organic matter, bulk density and nutrient relationships, which may subsequently influence species richness (Werner and Zedler, 2002; Jolley et al., 2010; Cui et al., 2013). In wetlands of the upper midwestern USA, high sedimentation rate often results in the decline of native species, and species-rich wet prairies, sedge meadows, and fens tend to become monotypic stands of *Phalaris arundinacea* L. or *Typha* spp. (Galatowitsch et al., 2000). One important reason that monotypic stands develop is because sediment accumulation may reduce the microtopography of tussocks in sedge meadows. Werner and Zedler (2002) reported a loss of one species per 1000 cm^2 of lost tussock surface area, and loss of 1.2 species for every 10 cm addition of sediment over the sedge meadow surface. In an Alaskan sedge wetland, van der Valk et al. (1983) found decreased shoot density of 35%, 72%, and 93% with burial depths of 5, 10, and 15 cm, respectively.

Wetland hydrologic regime is a major determinant of plant community development from seed banks (van der Valk, 1981). Seed germination of various species is very dependent on water level (Willis and Mitsch, 1995; Liu et al., 2005). Flooding may decrease light levels and available oxygen, leading to lower survival of dormant seed than in non-flooded conditions. Atmospheric oxygen levels are required for the germination and early seedling growth of most wetland species (Nicol et al., 2003; Lu et al., 2012). Ultimately, water and sediment level impacts germination and ultimately influence vegetation structure. As the water level fluctuates, species composition changes quickly; some populations may disappear and other populations may become established. Flooding regime is probably the most important factor affecting the distribution of community type and biodiversity (Willis and Mitsch, 1995; Nicol et al., 2003; Liu et al., 2005; Cui et al., 2009).

Sanjiang Plain is a vast complex of marshes, meadows and forests, which is located in the northeast area of Heilongjiang Province, China. Over the past five decades, the natural wetlands in this region have been extensively reclaimed for agriculture with a total loss of nearly 80% of the surface area (Wang et al., 2011). Lake Xingkai, the largest freshwater lake in northeast China, is situated at the lower watershed of the Heilongjiang River and plays an important role in providing water for irrigation. However, farming causes soil erosion, severe water pollution and other environmental problems. During the irrigation return flow (runoff) period, vast amounts of irrigation flow with high sediment load were drained to the river and wetlands, and then returned to Lake Xingkai (Piao and Wang, 2011). Although soil erosion was serious and the sediment accumulation rate in disturbed wetland was high, we found no studies regarding the effects of sediment load on the vegetation structure and seed bank germination in the freshwater marshes at the mid-high latitudes, northeast of China. In order to understand the effects of sediment load on the seedling emergence and vegetation structure of the *Calamagrostis angustifolia* wetland community, we selected one natural site which was relatively intact and one disturbed site which was largely disturbed by sediments from irrigation return flow. The objectives of this project were to gain insights into the responses of wetland vegetation in mid-high latitudes to environmental disturbances such as increasing sediment load and flooding. This information is important to the protection of aquatic plants and the restoration of impaired wetlands.

2. Methods

2.1. Study site

The study site is located in the National Natural Wetland Reserve of Lake Xingkai of the Sanjiang Plain in northern China. The annual mean precipitation is 750 mm and the annual mean temperature is approximately 3.1 °C (Wang et al., 2006). A large area of the original wetland was converted to a paddy field for agriculture. Water is pumped from Lake Xingkai to the paddy field for irrigation. During the irrigation return flow period, large amounts of irrigation return water with sediment are drained to wetlands, which are used to remove sediments and pollutants. Because of the agricultural return water, the wetland immediately adjacent to the paddy field was disturbed by sediments from irrigation return flow while the wetland immediately adjacent to the lake received relatively small amounts of sediment. During a recent study, the sediment accumulation rate for the natural site was estimated at 372–2776 $g\,m^{-2}\,yr^{-1}$, 0.03–0.2 $cm\,yr^{-1}$, while it was estimated at 3298–12889 $g\,m^{-2}\,yr^{-1}$, 0.3–1.0 $cm\,yr^{-1}$ for the disturbed site in our study sites (Professor Wang Guoping and Zou Yuanchun, 2012, unpublished data).

The study sites are dominated by freshwater marsh with shallow and intermittent water levels, varying from no standing water to an average depth of approximately 12 cm. The flora mainly consists of *C. angustifolia* Kom., with less dominant, but common species such as *Glyceria spiculosa* (Fr. Schmidt.) Rosh., *Phragmites australis* (Clav.) Trin., *Pycreus sanguinolentus* (Vahl) Nees., *Menyanthes trifoliate* L. and *Carex humida* Y.L. Chang et Y.L. Yang. We chose one 50 m × 50 m area of *C. angustifolia* community (45.354196° N, 132.32749° E) immediately adjacent to the lake as natural wetland site to do vegetation survey and soil samplings, and one 50 m × 50 m area of *C. angustifolia* community (45.360137° N, 132.306676° E) immediately adjacent to the paddy field as disturbed wetland site to do vegetation survey.

2.2. Seed bank collection

The seed bank was sampled during 2–4 May, 2012. Soil samples from 10 replicate plots (25 cm × 25 cm × 5 cm) in natural site were taken and placed into soil bags. Sediment (top 5 cm) was collected from an irrigation ditch adjacent to the disturbed site using a shovel. All samples were taken back to the greenhouse. In the laboratory, each soil sample was sieved to remove stones and plant fragments, and mixed thoroughly. Sediments collected from the irrigation ditch were placed in an oven at 105 °C for 10 h to kill seeds, and then ground to a fine powder and passed through a 2 mm soil sieve.

2.3. Seedling germination assay

Seed banks from the natural site were studied with two treatment factors, water regime and sediment addition. Two levels of water regime were used: one tank was assigned to the non-flooded (moist soil) treatment and one tank to a flooded treatment of 10 cm of continuous inundation. For each water regime treatment, six levels of sediment addition: 0, 0.25, 0.5, 0.75, 1.0 and 2.0 cm depth were used. Nine replicates were used for each level of sediment addition and water regime, resulting in a total of 108 replicates.

Seedling germination assays were conducted in the greenhouse during 9–12 May, 2012. The greenhouse was well ventilated to maintain an inside temperature comparable to that of the outside. Monthly air temperature in the greenhouse during the study period ranged from 16.4 °C in May to 23.8 °C in July. Each soil sample was spread as an even layer, 2 cm thick, in experimental pots (25 cm × 25 cm × 11 cm) previously filled with washed vermiculite to an 8 cm depth, a procedure similar to that described by van der Valk and Rosburg (1997) and Middleton (2003). The depth of sediment desired in each pot was achieved by calculating the volume of sediment required to fill the pot to the depth of each treatment level. The volume was measured with a graduated cylinder and sprinkled on top by hand and smoothed evenly over the seed bank sample. Newly emerged seedlings were identified, counted, and removed from the pots. The seedling germination assays continued until no additional seedlings emerged. The germination assay lasted approximately 5 months. Nomenclature follows Yi et al. (2008).

2.4. Vegetation survey

We conducted a vegetation survey both in the natural and disturbed sites on 10–11, August 2012. We chose one 50 m × 50 m area in the natural and disturbed sites respectively and placed ten 1-m^2 quadrats over the top of each area in the two sites. Species name, percent cover (%) and height (cm) of each species, and water depth were recorded.

2.5. Data analysis

A one-way ANOVA and a subsequent Tukey's test were used to test the difference in mean number of species, the mean seedling density and number of seedlings of each species among the sediment additions under non-flooded and flooded water regimes. Significance was determined at an alpha level of 0.05. Data of species richness and seedling density were transformed ($\log(x+1)$) to satisfy the assumption of homogeneous variances. Composition of standing vegetation (percent cover and height) of natural and disturbed wetland sites was described using means and standard errors. All statistics were conducted using SPSS version 16.0.

3. Result

3.1. Effect of sediment load on species richness and seedling emergence

Species richness and seedling emergence were significantly affected by sediment depth both in non-flooded ($F = 49.0$, $p < 0.001$; $F = 79.283$, $p < 0.001$) and flooded condition ($F = 8.241$, $p < 0.001$; $F = 17.21$, $p < 0.001$) (Fig. 1). In non-flooded condition, seedlings emerged from 0 and 0.25 cm sediment depths were large and the highest seedling emergence (49.17 ± 2.701) occurred from 0 cm sediment depth. There was a significant decline in emergence at 0.5 cm sediment depth and seedling density at 0.5 cm was only 34% of that at 0 cm. The number of species and seedlings were much lower at 0.75, 1 and 2 cm than at 0.5 cm, and only 3 and 4 species survived at 1 and 2 cm sediment depths, respectively. There were no significant difference of the number of species and seedling at 0, 0.25, 0.5 and 0.75 cm sediment depths in flooded condition, and they decreased significantly when sediment depths reached at 1 and 2 cm.

Twenty-three species germinated from the seed bank, and the number of species under the non-flooded condition was higher than the flooded treatment (20 vs. 7, respectively) (Table 1). The densities of seeds germinating of *C. angustifolia*, *Typha orientalis*, *Lythrum salicaria*, *Juncus effusus* decreased significantly at 0.25 sediment depth, and the number of seedlings from *Eleocharis mamillata*, *Alisma orientale*, *Galium trifidum*, *Stellaria longifolia* decreased significantly at 0.5 cm than at 0 and 0.25 cm sediment depths, while *Sagittaria trifolia*, *P. sanguinolentus*, *A. orientale*, *Salix rosmarinifolia*, *Salix myrtilloides* still survived at 1 and 2 cm sediment depths although the number of seedlings from them was very low in non-flooded condition. The number of seedlings from *Ceratophyllum demersum*, *Potamogeton crispus*, *T. orientalis* decreased significantly or disappeared at 0.5 cm sediment depth, while *S. trifolia*, *Monochoria vaginalis*, *A. orientale* still survived at 1 or 2 cm sediment depths in flooded condition.

3.2. Vegetation composition of the natural and disturbed sites

The species richness of the natural site was higher than that in disturbed site (20 vs. 11 species, respectively) (Table 2), and there are 12 and 6 dominant species present both in the seed bank and in the vegetation of the natural and disturbed wetlands respectively (Tables 1 and 2). As the perennial herbaceous species, *C. angustifolia* was the most dominant species in both sites, with less dominant, but common species such as *G. spiculosa*, *P. australis* and *P. sanguinolentus* in natural site and *G. spiculosa* in disturbed site. Shrubs including *S. myrtilloides*, *S. rosmarinifolia* and *Spiraea salicifolia* were present in both sites, and two annuals including *Polygonum hydropiper* and *P. sanguinolentus* were present in natural site while they were all absent from disturbed site. As the dominant species, the mean height of *C. angustifolia* and *G. spiculosa* was taller in the natural site than in the disturbed site ($p < 0.05$) (Table 2). For other species, we did not find significant difference of mean height between the two sites.

4. Discussion

Flood retention and sediment capture are notable ecosystem services provided by wetlands, but there are limits to the amount of sediments that can accrete without causing damage to the habitat and vegetation. In this study, we documented the influence of sediment accumulation on seed bank germination and vegetation in freshwater wetlands. In the greenhouse experiment, we found that species richness and seedling emergence were negatively affected by increasing sediment accumulation. In the vegetation survey, we found that sediment accumulation was associated with lower species richness in the disturbed wetland.

4.1. Effects of sedimentation and inundation on seed germination

Sedimentation rates vary in different wetlands and geographical regions due to natural factors such as the hydro-geological conditions and the regional level of agricultural activities (van der Valk et al., 1983; Kleiss, 1996; Greiner and Hershner, 1998; Mou and Sun, 2011; Riis et al., 2013). The accumulation rates of sediment for freshwater wetlands under natural conditions were usually on the order of 0.5 cm yr^{-1} or less in Alaska and other places of the world (van der Valk et al., 1983; Greiner and Hershner, 1998). In some cases, sediment addition in wetlands may be very high e.g., in coastal settings with intentionally deposited dredge spoil application (Middleton and Jiang, 2013). Farming activities accelerate the rate of sediment accumulation. Sediment accumulation observed in wetlands influenced by agricultural activities were more than 1 cm yr^{-1} and even higher than 3–4 cm yr^{-1} (Johnston et al., 1984;

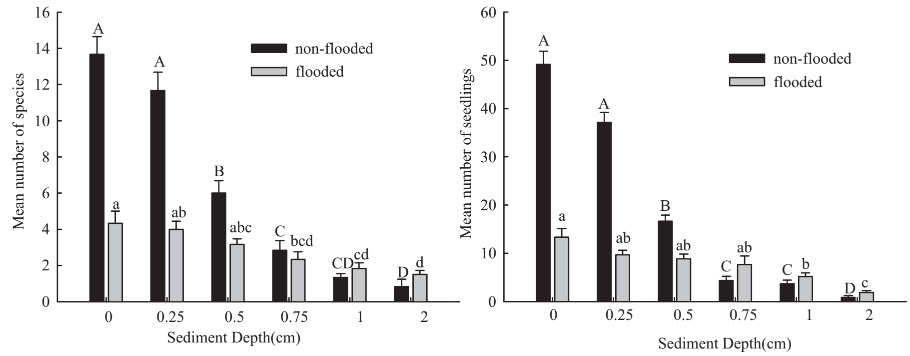

Fig. 1. Effect of sediment depth on the mean number of species and seedlings emerging in the seed banks of *Calamagrostis angustifolia* wetland community. Values are means ± SE ($n = 9$ pots per treatment). Different letters indicate significant difference in different sediment depths ($p < 0.05$).

Fennessy et al., 1994; Giroux and Bédard, 1995; Kleiss, 1996). To meet the water supply needs for the three large state agriculture farms located in the Lake Xingkai National Reserve, 5.04×10^8 m^3 of water were pumped from Lake Xingkai for irrigation of the paddy field constructed around the lake (Piao and Wang, 2011), and consequently, large amount of sediment was returned to the wetland in the agricultural return water delivered to wetlands immediately adjacent to the paddy field. The accumulation rates of sediment in the natural wetland were 0.03–0.2 cm yr^{-1}, while it was estimated at 0.3–1.0 cm yr^{-1} for the disturbed wetland near the paddy field in our study sites.

Sediment accumulation might impact species richness by smothering plants. van der Valk et al. (1983) documented a decrease in shoot density in areas of experimental sediment addition. In a controlled greenhouse experiment with seedlings grown in flats receiving small amounts of sediment (0.25–0.5 cm), Jurik et al. (1994) demonstrated that the richness of aquatic species per flat and the total number of seedlings decreased significantly with increasing depth of sediment. Wang et al. (2013) found species richness and seedling emergence of three plant communities decreased significantly at 0.5–0.75 cm of sediment addition and species responded differently to the addition of sediment. By limiting oxygen availability, burial and changes in soil particle size can also have a negative effect on recruitment, thereby disrupting the composition of a species-rich wetland by inhibiting regeneration of sensitive species (Keddy and Constabel, 1986). The specific response depends on the species, hydrology, sediment depth and seed mass.

Seed burial has both positive and negative consequences for seedling emergence, so there is an optimal range of burial depth to

Table 1

Mean number of seedlings per pot in each sediment depth treatment in *Calamagrostis angustifolia* wetland community in Xingkai Lake. Species are listed in order of decreasing abundance in the 0-cm sediment addition treatment. Contrasts (F and p values) are based on ANOVA comparisons and Tukey's test of log-transformed data for different sediment depth treatments. Different letters indicate that the comparison was significantly different (Tukey's test; $p < 0.05$; $n = 9$ pots per treatment). * indicates that the species germinated both in flooded and non-flooded conditions.

Water regime	Sediment depth (cm)						Contrast	
	0	0.25	0.5	0.75	1	2	F	p
Non-flooded								
Eleocharis mamillata	15.33 ± 0.99^a	14.5 ± 1.43^a	4.83 ± 0.31^b	0.5 ± 0.22^c	0 ± 0^c	0 ± 0^c	267.17	<0.001
*Alisma orientale**	6 ± 0.82^a	5.33 ± 0.61^a	2.33 ± 0.33^b	0.5 ± 0.22^c	0 ± 0^c	0.17 ± 0.17^c	62.776	<0.001
*Sagittaria trifolia**	5.83 ± 1.01^a	3.67 ± 0.56^a	4.5 ± 0.99^a	2.67 ± 0.67^a	3.33 ± 0.84^a	0.17 ± 0.17^b	13.529	<0.001
Calamagrostis angustifolia	3.33 ± 0.33^a	0.33 ± 0.21^b	0 ± 0^b	0 ± 0^b	0 ± 0^b	0 ± 0^b	71.482	<0.001
*Typha orientalis**	3 ± 0.45^a	0.67 ± 0.33^b	0 ± 0^c	0 ± 0^c	0 ± 0^c	0 ± 0^c	36.698	<0.001
Galium trifidum	2.5 ± 0.43^a	2.17 ± 0.4^a	0.33 ± 0.21^b	0 ± 0^b	0 ± 0^b	0 ± 0^b	34.619	<0.001
Lythrum salicaria	2.33 ± 0.42^a	1.33 ± 0.21^b	0 ± 0^c	0 ± 0^c	0 ± 0^c	0 ± 0^c	70.795	<0.001
Lysimachia thyrsiflora	1.83 ± 0.6^{ab}	1.33 ± 0.61^{bc}	3 ± 0.45^a	0 ± 0^c	0 ± 0^c	0 ± 0^c	13.263	<0.001
*Juncus effusus**	1.67 ± 0.33^a	1.83 ± 0.4^b	0 ± 0^b	0 ± 0^b	0 ± 0^b	0 ± 0^b	44.012	<0.001
Stellaria longifolia	1.33 ± 0.42^a	1.33 ± 0.33^a	0.33 ± 0.21^b	0 ± 0^b	0 ± 0^b	0 ± 0^b	9.689	<0.001
Rorippa palustris	1.17 ± 0.48^a	0.67 ± 0.21^{ab}	0.33 ± 0.21^{ab}	0 ± 0^b	0 ± 0^b	0 ± 0^b	4.82	0.002
Pycreus sanguinolentus	1.17 ± 0.31^{ab}	1.5 ± 0.43^a	0.33 ± 0.21^{abc}	0.33 ± 0.21^{abc}	0.17 ± 0.17^{bc}	0 ± 0^c	5.555	0.001
Stachys baicalensis	1 ± 0.63^a	0.17 ± 0.17^a	0 ± 0^a	0.17 ± 0.17^a	0 ± 0^a	0 ± 0^a	2.358	0.064
Saussurea amurensis	0.83 ± 0.4^a	0.17 ± 0.17^a	0.17 ± 0.17^a	0 ± 0^a	0 ± 0^a	0 ± 0^a	2.721	0.038
Carex humida	0.5 ± 0.22^a	0.33 ± 0.21^a	0 ± 0^a	0 ± 0^a	0 ± 0^a	0 ± 0^a	3.118	0.022
Hieracium hololeion	0.5 ± 0.22^{ab}	0.83 ± 0.31^a	0 ± 0^b	0 ± 0^b	0 ± 0^b	0 ± 0^b	5.824	0.001
Ranunculus japonicus	0.33 ± 0.21^a	0 ± 0^a	0 ± 0^a	0 ± 0^a	0 ± 0^a	0 ± 0^a	2.5	0.052
Salix rosmarinifolia	0.33 ± 0.21^a	0.67 ± 0.49^a	0.33 ± 0.21^a	0 ± 0^a	0.17 ± 0.17^a	0.17 ± 0.17^a	0.702	0.626
Chamaenerion angustifolium	0.17 ± 0.17^a	0 ± 0^a	0.17 ± 0.17^a	0 ± 0^a	0 ± 0^a	0 ± 0^a	0.8	0.558
Salix myrtilloides	0 ± 0^a	0.17 ± 0.17^a	0 ± 0^a	0.17 ± 0.17^a	0 ± 0^a	0.33 ± 0.21^a	1.111	0.375
Flooded								
*Sagittaria trifolia**	6.83 ± 1.1^a	4.5 ± 0.56^a	5.17 ± 0.79^a	5.17 ± 0.79^a	4.33 ± 0.88^a	1.17 ± 0.31^b	9.42	<0.001
Monochoria vaginalis	2.67 ± 0.42^a	2 ± 0.73^{ab}	2.33 ± 0.62^a	1.17 ± 0.6^{abc}	0.33 ± 0.21^{bc}	0 ± 0^c	7.947	<0.001
Ceratophyllum demersum	1.5 ± 0.62^a	0.83 ± 0.4^{ab}	0 ± 0^b	0 ± 0^b	0 ± 0^b	0 ± 0^b	5.399	0.001
*Alisma orientale**	0.83 ± 0.4^a	1.67 ± 0.33^a	1 ± 0.26^a	1.17 ± 0.54^a	0.5 ± 0.22^a	0.67 ± 0.21^a	1.264	0.305
Potamogeton crispus	0.67 ± 0.33^a	0.33 ± 0.21^a	0 ± 0^a	0 ± 0^a	0 ± 0^a	0 ± 0^a	3.147	0.021
*Juncus effusus**	0.5 ± 0.22^a	0.33 ± 0.21^a	0.33 ± 0.21^a	0.17 ± 0.17^a	0 ± 0^a	0 ± 0^a	1.467	0.23
*Typha orientalis**	0.33 ± 0.21^a	0 ± 0^a	0 ± 0^a	0 ± 0^a	0 ± 0^a	0 ± 0^a	2.5	0.052

Table 2
Vegetation composition of natural and disturbed *Calamagrostis angustifolia* wetland community sites including mean ± S.E. percent cover and height (cm) in the National Natural Wetland Reserve of Lake Xingkai on August 10–11, 2012.

Species	Cover (%)		Height (cm)	
	Natural	Disturbed	Natural	Disturbed
Alisma orientale	0.3 ± 0.2	0.4 ± 0.2	57.2 ± 4.5	57.7 ± 5.6
Anemone dichotoma	2.3 ± 1.1		32.5 ± 2.5	
Calamagrostis angustifolia	76.4 ± 5.3	87.4 ± 4.2	89.0 ± 8.7	74.6 ± 1.9
Carex humida	2.0 ± 1.5	0.4 ± 0.2	76.0 ± 7.5	55.8 ± 6.4
Equisetum arvense		0.6 ± 0.3		37.2 ± 5.3
Galium trifidum	0.4 ± 0.3		35.7 ± 10.8	
Glyceria spiculosa	11.5 ± 5.8	14.7 ± 6.1	61.7 ± 5.6	52.5 ± 2.5
Juncus effusus	0.2 ± 0.2		32.0 ± 0	
Lycopus lucidus	3.9 ± 1.5		35.9 ± 4.7	
Lysimachia thyrsiflora	1.8 ± 0.7		39.0 ± 6.2	
Menyanthes trifoliata	2.7 ± 1.3	3.9 ± 0.9	27.4 ± 4.8	31.7 ± 6.0
Phragmites australis	6.9 ± 2.8	4.6 ± 2.8	106.0 ± 6.7	97.2 ± 5.7
Polygonum hydropiper	0.4 ± 0.3		55.5 ± 5.0	
Pycreus sanguinolentus	5.0 ± 3.1		55.8 ± 6.4	
Sagittaria trifolia	0.5 ± 0.5	0.4 ± 0.2	57.0 ± 0	49.5 ± 7.2
Salix myrtilloides	1.5 ± 1.1	1.3 ± 0.8	45.0 ± 7.6	60.0 ± 7.5
Salix rosmarinifolia	0.7 ± 0.5	0.3 ± 0.2	80.0 ± 12.4	50.0 ± 10.0
Sium suave	0.5 ± 0.5		100.0 ± 0	
Spiraea salicifolia	1.4 ± 1.1	1.3 ± 0.7	68.3 ± 9.3	70.0 ± 12.5
Stachys baicalensis	0.2 ± 0.1		32.5 ± 3.3	
Typha orientalis	0.5 ± 0.2		89.2 ± 7.3	

maximize the seedling emergence and subsequent seedling grown (Li et al., 2006; Middleton and Jiang, 2013). Deep burial prevents the germination of the seed so that the seed remains dormant or loses its longevity in the soil. In our experiment, emergence of seedlings decreased with the increase in sediment depth. The highest seed germination occurred at 0 cm of sediment depth, but germination decreased significantly with 0.5 cm of sediment addition. Similar relationships were also reported in some other studies (Galinato and Van Der Valk, 1986; Jurik et al., 1994; Wang et al., 1994; Gleason et al., 2003; Mou and Sun, 2011; Rivera et al., 2012; Wang et al., 2013). The probable reasons may be the physiological requirements, the structural limitations of seeds and the microenvironment factors (such as oxygen, light, temperature, humidity and nutrients) surrounding the seeds. Increased sediment depth would create a microenvironment for seeds in which the seeds would be unlikely to germinate. Ultimately this situation may cause a barrier for vegetation re-establishment and change the vegetation composition of sediment-affected wetlands.

There is a positive correlation between seed mass and emergence ability of seeds both within and between plant species, and sedimentation acts as an environmental filter for small-seeded species (Jurik et al., 1994; Grundy et al., 2003; Li et al., 2006; Stromberg et al., 2011). Large seed size results in a greater tolerance to a wide range of environmental stresses such as sediment, predators, nutrient deprivation and prolonged periods in deep shade (Armstrong and Westoby, 1993; Thompson et al., 1993; Rivera et al., 2012). Galinato and Van Der Valk (1986) showed that species with seeds weighing 1 mg were more sensitive to sediment than species with seeds weighing 3 mg. A one-time application of 0.5–1.0 cm of sediment can reduce seedling emergence in small-seeded species (Jurik et al., 1994; Peterson and Baldwin, 2004; Wang et al., 2013). For larger seeds, a sediment burial threshold seems to be about 2 cm deep (Zheng et al., 2005; Jolley et al., 2010), therefore, sedimentation could potentially cause a shift in community composition in favor of large-seeded species. Some species in our study showed a similar response to sediment addition as compared to other studies. For example, as a very small-seeded species, *Typha* was greatly affected by 0.25 cm of sediment addition (Jurik et al., 1994; Wang et al., 1994; Gleason et al., 2003), as small-seeded species, *J. effusus*, *C. angustifolia* and *E. mamillata* decreased significantly at 0.25 cm, and disappeared at 0.5 cm of sediment (Gleason et al., 2003; Stromberg et al., 2011; Wang et al., 2013), As for *Sagittaria*, small amounts of sediment did not significantly hamper its seedling emergence until larger amounts of sediment (1–2 cm) were applied (Jurik et al., 1994; Gleason et al., 2003; Wang et al., 2013), As a large-seeded species, *A. orientale* was more resilient to sediment depth (Gleason et al., 2003). So species responding differently to sediment in our study may be related to seed mass.

The hydrologic and sediment regimes of many wetlands around the world have been changed by the development of hydroelectric, irrigation and water supply schemes, and clearing of land for agriculture. Wetland plants often germinate as a result of particular hydrological conditions and seedling emergence from wetland soil seed banks tends to be controlled primarily by hydrological factors (van der Valk, 1981; Liu et al., 2005). In some instances, only 25% of species survived under flooded and non-flooded conditions. A change in water depth by as little as 2 cm could significantly affect seed germination (van der Valk and Davis, 1978; Liu et al., 2005). Water levels in our study site varied from no standing water to depths of approximately 12 cm during the growing season. However, water depth was even greater when irrigation flow was returned to the wetlands (Piao and Wang, 2011). Our results showed significant differences in seed germination under flooded and non-flooded conditions and significantly more species and seedlings germinated under non-flooded condition, which is consistent with other studies (Peterson and Baldwin, 2004; Liu et al., 2005). Seedling emergence seemed to be more tolerant to the sediment under flooded conditions in our study, this is because large-seeded species *S. trifolia*, *M. vaginalis* and *A. orientale* were dominant species under flooded conditions and these species have a greater tolerance to environmental stresses. Other studies found that sediment deposition during flooding can change the tolerance of wetland species to inundation, reducing survival and growth (e.g., *Carex* and *Calotis* species) (Lowe et al., 2010). Wetland species (sedge and tree species) were resilient to cycles of flooding and drying, but sediment deposition resulted in decreased biomass, which was diminished further by high water levels (Ewing, 1996).

4.2. Effect of sediment on vegetation structure

Sediment deposition negatively impacted wetland vegetation, and often contributed to the loss of native species in remnant wetlands (Ewing, 1996; Werner and Zedler, 2002). In our study, species richness was less in the sediment disturbed site than the natural site, and many native species (e.g., *Anemone dichotoma*, *G. trifidum*, *J. effusus*) were not present in the disturbed site. Annuals such as *P. hydropiper* and *P. sanguinolentus* were present in the natural site, but were absent from the disturbed site. The possible mechanisms may be that the high addition of sediment inhibit seed germination and seedling growth, or influence the soil properties and topography, and lead to the changes of vegetation.

The addition of sediment reduced the germination rates of seeds from wetland seed banks in our greenhouse experiment, so that sedimentation is also likely to affect recruitment in field conditions. High sedimentation rates in prairie wetlands (greater than 0.3 cm yr^{-1}) bury organisms and affect germination rates of wetland plant species. In the seed bank experiment, germination rates of small-seeded species *G. trifidum*, *J. effusus*, *Lysimachia thyrsiflora*, *Stachys baicalensis* and *T. orientalis* decreased significantly with 0.5 cm sediment addition, and these species were absent from the vegetation composition of the disturbed site, with sediment accumulation of 0.3–1.0 cm yr^{-1}. The current elevation of Lake Xingkai may have been raised by 0.28–0.36 m as compared to the elevation during the 1970s and 1980s as the result the irrigation return flow, which carried heavy sediment loads from the paddy field (Piao and Wang, 2011). In a sedge meadow in Wisconsin, the sediment was washed in from the surrounding uplands following agricultural and urban development that began approximately 150 years ago, and has accreted to 0.4–1.3 m above histic horizons (Johnston et al., 1984), which also has lead to the decline of native species (Werner and Zedler, 2002).

Increased sediment loadings also alter wetland habitat and influence species richness in many wetlands. Sediment accumulation may influence soil properties, and changes in organic matter and bulk density caused by sedimentation may influence the decrease in species richness by altering soil structure and the moisture regime. For example, sediment accumulation may slow the growth of *Carex* by altering soil structure and moisture (Ashworth, 1997; Werner and Zedler, 2002). Sedimentation creates canopy gaps, fresh substrate, and nutrient additions that appear to enhance a wetland's invisibility (Johnston et al., 1984; Cui et al., 2013). Sediment may also eliminate wetland habitat through filling, and alter the sedge meadows by diminishing microtopography, *Carex* meadows are invaded by *Phalaris* and *Typha* as results of the reduction of microtopographic relief and increase in sediment accumulation (Werner and Zedler, 2002).

The response to sedimentation by different plant species could vary as a function of life-history characteristics. In many perennial-dominated wetlands, annuals are more capable of tolerating sedimentation and the total seedling density of annuals in many times greater than perennials (van der Valk et al., 1983; Dittmar and Neely, 1999). In the seed bank study, as an annual herbaceous species, *P. sanguinolentus* could germinate at 1 cm of sediment depth, while perennial herbaceous species like *C. angustifolia* disappeared at 0.5 cm of sediment depth, which was consistent with studies in this region (Wang et al., 2013). By contrast, sediment addition may significantly affect species that reproduce only by seeds. When the sediment input exceeds sustainable levels, germination of these species may be restricted. For perennial species, they may still re-establish by vegetative propagation. So in wetlands with extensive mudflats and bare soils, or those that depend heavily on seed germination, the addition of small amounts of sediment may be more problematic than in well-vegetated freshwater emergent marshes such as the one studied here. In our seed bank study, perennial species (e.g., *C. angustifolia*, *C. humida*) disappeared at 0.5 cm of sediment depth, but they still present in the vegetation in the disturbed site although sediment accumulation was highly at 0.3–1.0 cm yr^{-1}. This was mainly because perennial species could use the rhizome as their reproductive strategy. So our study showed that increasing sediment additions from agricultural seemed to favor asexually reproducing species by covering up seeds.

5. Conclusions

Our study revealed the negative impacts of sedimentation on seed germination and vegetation composition in irrigation return flow in the National Wetland Reserve of Lake Xingkai. In the greenhouse experiment, we found that species richness and germination density were negatively affected by increasing sediment depth. In the vegetation survey, the wetland disturbed with sedimentation from agricultural return water had lower species richness than the natural wetland. The tolerance to increasing sediment accumulation might be related to the seed mass and life-history characteristics. Small-seeded species and annual species seemed to be more susceptible to the high sediment load, so more attention should be paid to this sensitive wetland species' protection and restoration. Knowledge gained from this and other similar studies will provide important insights into watershed protection strategies that might be implemented to reduce soil loss and conserve wetlands located downstream of agricultural lands.

Acknowledgments

We thank the reviewers and editors for their valuable advice, and Dr. Beth Middleton for suggestions and English editing. This study was supported by National Natural Science Foundation of China (grant # 41271106), Key Technologies R&D Program of China (grant # 2012BAC19B05), Program for Creative Research Teams of Jilin Province (grant # 20130521012JH), and CAS/SAFEA International Partnership Program for Creative Research Teams.

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