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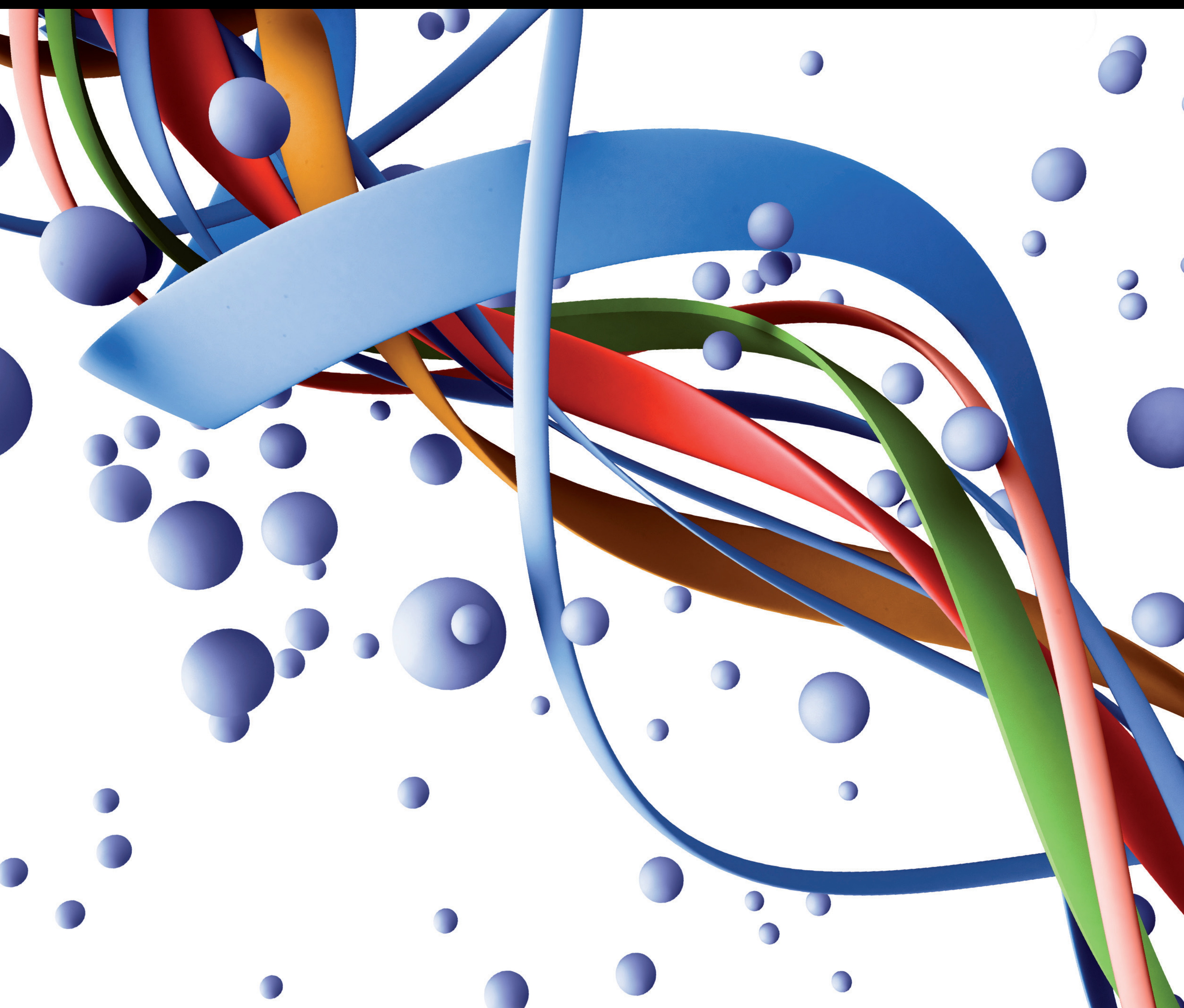
Restoration and Management of Healthy Wetland Ecosystems

Guest Editors: Dong Xie, Qiang Wang, Zhongqiang Li, Rogar P. Mormul, and Liandong Zhu



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Editorial

Restoration and Management of Healthy Wetland Ecosystems

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Wetland ecosystem is one of the most important ecosystems in the world. However, with the large-scale urbanization and rapid economic development, the wetland ecosystem is facing increasing ecological and environmental issues including eutrophication, water pollution, biodiversity loss, and ecological function decreases. Therefore, improving and restoring wetlands and creating healthy wetland ecosystem are the pressing issues these days.

Healthy ecosystem means not only ecological health itself, but also the long-term maintenance of healthy human population and the sustainable promotion of social-economic development. Healthy wetland ecosystem would provide not only suitable habitats for wildlife but also important ecosystem functions for local sustainable development. To create healthy wetland, two aspects are mainly included: one is restoration and remediation of degraded wetland ecosystems and the other is protection and management of wetland ecosystems which have not yet been disturbed, so that these wetlands can undergo benign development.

The restoration and remediation of the degraded wetlands are the most important issues of wetland researches and have developed rapidly with the development of wetland restoration theories in recent years. Our understanding of how wetland ecosystem changes is becoming more deeper than any time in human history, but descriptions of the variety and severity of changes have been repeatedly stated for decades now, yet the alarming trends continue. Similarly, wetland management traditionally derives its knowledge base from the fields such as aquatic chemistry and biology and hydrology. However, this disciplinary training does not equip wetland managers for the challenge of addressing the drivers of ecosystem change as described above, the societal processes that produce the need for more food and more water and land use change.

In this special issue, we focused on restoration and management of healthy wetlands. There are 3 aspects and 11 papers on biodiversity protection, hydrology management, and habitat restoration in the current special issue. For biodiversity protection, W. Du et al. monitored the composition and biomass of aquatic vegetation in the Poyang lake; H. L. Clipp et al. surveyed winter water bird community composition in West Virginia, USA; C. Yin et al. monitored the *Microcystis* biomass changes at Meiliang Bay, Lake Taihu, China; Y. Yuan et al. researched the effect of mowing on the competition of *Phragmites australis* and *Spartina alterniflora* in the Yangtze Estuary; Y. Zhang et al. monitored migratory shorebird responses to prey distribution in a large temperate arid wetland, China; Z. Ge et al. studied that emerged community enhanced the enrichment of nitrogen and phosphorus in the wetland. For hydrology management, Y. Zhang et al. studied the long-term relationship between precipitation and aquatic vegetation succession in east Taihu Lake, China. X. Chen et al. simulated the effect of artificial water transfer on carbon stock of *Phragmites australis* in the Baiyangdian wetland, China; N. Yang et al. studied the effect of hydrologic alteration on the community succession of macrophytes at Hanjiang River, China. For habitat restoration, J. Cao et al. studied improvement of urban water environment in Eastern China; Y. Li et al. studied a carbon cycle model for the social-ecological process in coastal wetland, East China.

The restoration and management of healthy wetland ecosystem mainly adopt biological, ecological, and engineering technology, gradually restore the structure and function of degraded wetland ecosystem, and finally reach the self-sustaining state of wetland ecosystem. The core messages and directions of this special issue would ameliorate the ecological and environmental effects on wetlands; at least, they might reflect some questions or problems, which grabbed the researches and managers attention.

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Research Article

Winter Waterbird Community Composition and Use at Created Wetlands in West Virginia, USA

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Information on nonbreeding waterbirds using created wetlands in the Central Appalachian region of the United States is limited. We compared waterbird communities of two managed wetlands, created in 2013 and 2001, in West Virginia. We observed 27 species of waterbirds. Species richness and diversity were generally similar between the wetlands, but species composition and use differed. *Branta canadensis* (Canada Geese), *Anas strepera* (Gadwall), *Bucephala albeola* (Buffleheads), *Aythya affinis* (Lesser Scaup), and *Aythya collaris* (Ring-Necked Ducks) used the older wetland most frequently. Disparities in species use were the highest in March. The older wetland differed from the younger in supporting species such as diving ducks, possibly due to differences in size, vegetation, water depth, and microtopography. However, the ability to provide habitat for waterbirds during the winter was determined to be comparable between wetlands, despite their age difference.

1. Introduction

Wetlands provide an assortment of ecosystem services, such as flood control, nutrient cycling, water filtration, and pollution removal [1, 2]. They can improve water quality, control shoreline erosion, provide natural products, and contribute to the economics of fishing, hunting, agriculture, and recreation [3]. In addition, wetlands are complex ecosystems that provide habitat for a diversity of animals, including insects, mollusks, fish, amphibians, mammals, and birds [4]. Though wetlands comprise a small percentage of the nation's total land area (~5.5%), they harbor a disproportionately high number of unique plants and animals [5]. In the United States, at least one-third of threatened and endangered species lives in or depends on wetlands [6].

Wetlands within the migratory and wintering ranges of waterbird species are critical to conserve and sustain their populations. Waterbirds use coastal and inland wetlands as stopover sites during migration and as habitat to rest, feed, or overwinter [7, 8]. For example, vegetated playa wetlands on the Southern High Plains of Texas can support thousands of waterbirds between November and January [7]. Tens of thousands of waterbirds use wetlands in the San Joaquin Valley of California in January and February [9]. Wetlands can also be important in conserving endangered and threatened bird species, such as *Rallus crepitans* (Gmelin) (Clapper Rails) and *Ammospiza maritimus* (Wilson) (Seaside Sparrows) [10]. The loss of wetlands may explain the declining populations of certain waterbirds [11].

Despite their many benefits, wetlands tend to conflict with competing land and resource development interests. Over the past 2 centuries, many wetlands have been destroyed, converted for agricultural purposes, developed, or manipulated for other human uses. From the 1780s to the 1980s, the conterminous United States lost 53% of its original wetlands [12]. Due to the severe historic loss of wetlands, the United States adopted a national policy of "no net loss of wetlands." Destruction or degradation of wetlands now requires permits and usually entails either on-site mitigation or mitigation of wetlands of the same size or larger and similar functions in another location. Due to the "no net loss" policy, thousands of hectares of wetlands have been created or restored in compensation for wetland destruction and disturbance due to human activities. For instance, 198,230 ha of former upland were converted to wetlands and an estimated 83,890 ha of freshwater ponds were created from 2004 to 2009 [13].

Several studies have focused on wetland functions and communities within the Central Appalachian region. In West Virginia, Gingerich and Anderson [14], Gingerich et al. [15], and Balcombe et al. [16] examined litter decomposition and plant communities, respectively, in mitigated and reference wetlands. Francl et al. [17] surveyed small mammal communities at wetlands in West Virginia and Maryland. Strain et al. [18] investigated the diet composition and selection of prey by *Notophthalmus viridescens viridescens* (Rafinesque) (Red-spotted Newt) in created and natural wetlands in the Central Appalachians to assess functional equivalency between the wetlands. In addition, Balcombe et al. [19–21] compared aquatic macroinvertebrate, anuran, and breeding season avian assemblages in mitigation and reference wetlands. Although Balcombe et al. [19, 20] found that the mitigation wetlands in their study provided quality habitat for wildlife, they do not all match the function and structure of natural or reference wetlands [22–26]. Thus, it is critical to assess and monitor how created and mitigated wetlands function in offering the same ecological services as natural or reference wetlands. The aforementioned research has been valuable in evaluating the success of mitigation wetlands in supporting wildlife taxa, but there are few studies that specifically focus on waterbird use of created wetlands in the Central Appalachians and even fewer that focus on winter or nonbreeding waterbird communities.

In the summer of 2013, the West Virginia Division of Natural Resources (WVDNR) partnered with West Virginia University and AllStar Ecology LLC to create a mitigated wetland in the Pleasant Creek Wildlife Management Area (WMA), located in north-central West Virginia. The created wetland (hereafter referred to as PC2013) is one of few wetlands in West Virginia managed specifically for the benefit of migratory and wintering waterbirds (e.g., food-producing vegetation was planted and water levels are manipulated). The WVDNR's primary goal was to develop the wetland for waterfowl use and for both consumptive and nonconsumptive waterfowl recreation. It is generally assumed that created wetlands will provide the same ecological services as a natural wetland, but it is not guaranteed, and wetland age may be a confounding factor [24]. Therefore, the purpose of this study was to assess and compare the winter waterbird communities of the recently created wetland and an adjacent older wetland created in 2001 (hereafter referred to as PC2001) in the Pleasant Creek WMA. Our objectives were to (1) perform weekly waterbird surveys at PC2013 and PC2001 from November to March of 2013-2014 and 2014-2015 to determine nonbreeding waterbird use in this region; (2) compare annual and monthly waterbird species richness, diversity, composition, and use at the 2 differently aged wetlands; (3) examine trends in waterbird use during the study period; and (4) determine whether the recently created PC2013 was providing comparable winter waterbird habitat to an older wetland.

2. Materials and Methods

2.1. Field-Site Description. Our study took place in the Pleasant Creek WMA, located in the Tygart Valley watershed of north-central West Virginia, USA (Figure 1). The 2 study sites included the newly created wetland and an established wetland, which are found in the eastern portion of the Pleasant Creek WMA, near the junction of Taylor and Barbour Counties. A portion of the Pleasant Creek WMA is part of the U.S. Army Corps of Engineers (USACE) Tygart Lake flood control project. The USACE owns land up to the elevation of 362.7 m on the WMA, and the remainder is owned by WVDNR; the entire area is managed by the WVDNR Wildlife Resources Section. The Pleasant Creek WMA consists of mixed hardwood forest and wetland area, totaling 1,226 ha, with moderately steep slopes rising to 488 m in elevation. The area is primarily used for hunting, viewing wildlife, and recreational fishing.

The Pleasant Creek WMA is located within the Appalachian Plateau physiographic province. The underlying rock in this region is sedimentary, and streams tend to be dendritic. The regional climate is generally considered to be humid continental, with humid summers and cool to cold winters. The average precipitation for this region falls between 381 and 442 cm, with temperatures ranging from −3.3 to 5.0°C in January and 19.4 to 24.4°C in July. Because of the area's valley topography, dense fogs are a common occurrence. Cloudy skies are also frequent due to the damming of moisture from the Appalachian Mountains.

Prior to creation, PC2013 had been a maintained field dominated by *Phalaris arundinacea* L. (Reed Canary Grass), an invasive species with minimal value as waterbird and wildlife habitat. In conjunction with West Virginia University, AllStar Ecology LLC, and the Tygart Valley Conservation District, the WVDNR oversaw the creation of the 2.96-ha wetland. Funded by wetland mitigation money received by the WVDNR, the restoration project commenced in June 2013 and construction was mostly completed by August 2013. AllStar Ecology LLC developed the site plans, and the Tygart Valley Conservation District conducted the earthwork. Drainage tiles were removed, deep pockets were excavated, berms were created, and water control devices were installed. The Reed Canary Grass was controlled and a more natural hydrology was restored, creating conditions more suited to native wetland vegetation. *Trifolium repens* L. (Will Ladino Clover), *Lolium perenne* L. (Perennial Rye), and *Echinochloa esculenta* (A. Braun) H. Scholz (Japanese Millet) were planted on the berm and in the wetland after construction during the 2013 growing season. A native wetland seed mix was sowed during the 2014 growing season.

PC2013 is mostly bordered by forest, with the northern portion partly under tree and shrub cover (Figure 2). A small stream (Pleasant Creek) runs along the eastern and southern boundaries of the wetland, adding to habitat complexity. The water depth of PC2013 is relatively shallow, averaging 0.45–0.61 m, with a maximum of 1.4 m. The depth can be manipulated by the WVDNR to meet waterbird needs or other objectives. In comparison, PC2001 is larger in area (13.78 ha) and contains 7 islets. Pleasant Creek runs through

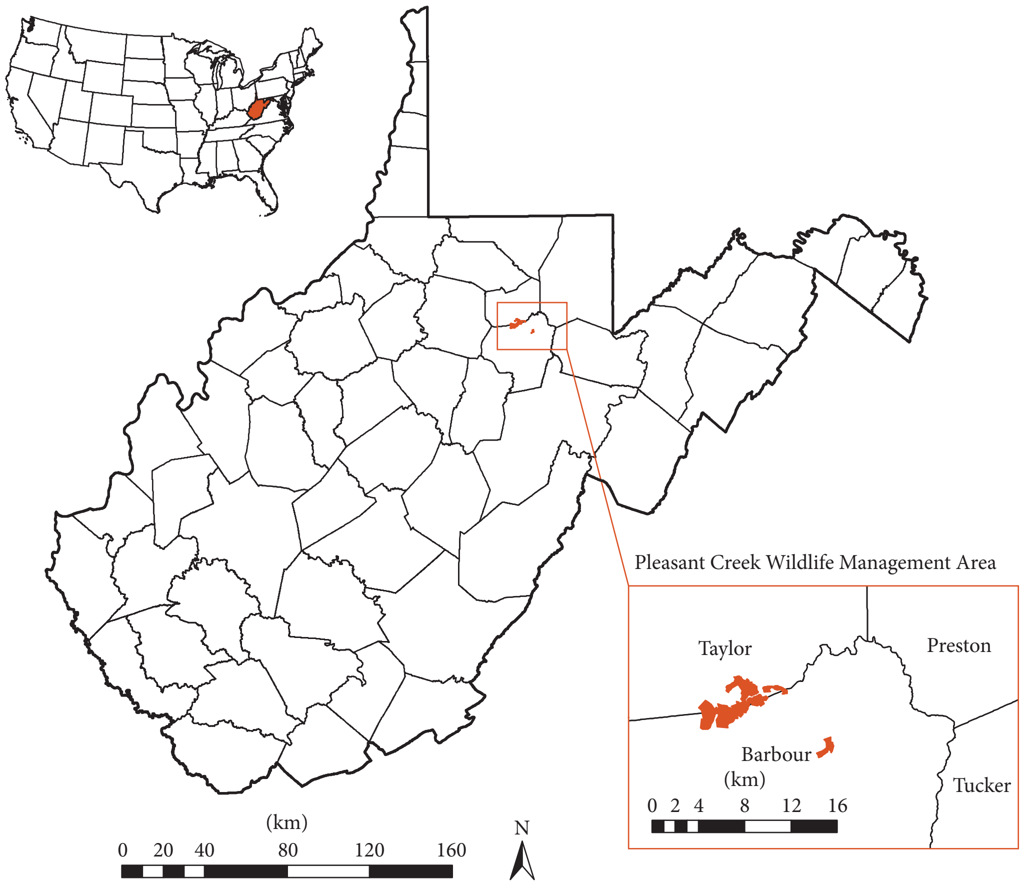

Figure 1: Location of Pleasant Creek Wildlife Management Area, WV. The shaded area at the border of Taylor and Barbour Counties represents the Pleasant Creek Wildlife Management Area, which is managed by the West Virginia Division of Natural Resources.

the wetland, entering from the southern end and exiting from the northern portion (headed downstream to PC2013). A combination of forest, small fields (<0.4 ha), and roads border the wetland (e.g., Pleasant Creek Road runs along the western edge while Route 119 flanks the eastern edge). The water of PC2001 is generally deeper, with a maximum depth over 1.83 m in some areas. According to land cover data from the WVDNR's 2015 Terrestrial Habitat Map, both wetlands are characterized predominantly by open water and small stream riparian habitat, with minor developed sections from bordering roads [27]. Based on a 25 m buffer around the edge of each wetland, PC2001 has 5.29 ha of core habitat (39.2%), while PC2013 has 0.16 ha of core habitat (5.3%). Together, the 2 wetlands total an area of roughly 16.74 ha.

2.2. Waterbird Surveys. We conducted weekly surveys from November to March in 2013-2014 and 2014-2015. Each wetland was surveyed 2 or 4 times per week by 1-2 trained observers. We conducted half of the surveys during morning hours, beginning within 30 minutes of sunrise, and half of the surveys during the evening, ending within 30 minutes of sunset, as dawn and dusk are primary waterbird foraging hours [28, 29]. Surveys were conducted on foot and by vehicle, and birds were identified from a distance with binoculars and a spotting scope to avoid disturbance. Waterbirds were considered waterfowl, seabirds, shorebirds, wading birds, and *Megaceryle alcyon* L. (Belted Kingfishers) [30]. Half of the morning and evening surveys proceeded starting at PC2001 and half starting at PC2013. The amount of time spent at each wetland was standardized at 30 minutes. Surveys at PC2013 were conducted by walking along the water's eastern edge and at PC2001 by walking or driving along the northern and western boundaries of the wetland, periodically stopping at locations from which a large portion of the wetland could be observed. Each waterbird was identified to species and sex when possible. To avoid pseudoreplication or double-counts, we systematically and sequentially surveyed sections of the wetland that did not overlap, making note of the species, sex, and number of waterbirds that flushed or swam from one section to another.

The study was focused on waterbirds that were actively using the wetlands; therefore, we recorded only waterbirds observed in the wetland or within 10 m of the wetland's boundary. The small size and accessibility of the wetlands allowed for total counts of waterbirds. Birds that flew over the wetlands but were not foraging or actively using the wetland were not included in the analyses. Additional data collected included the Julian date, times that the survey

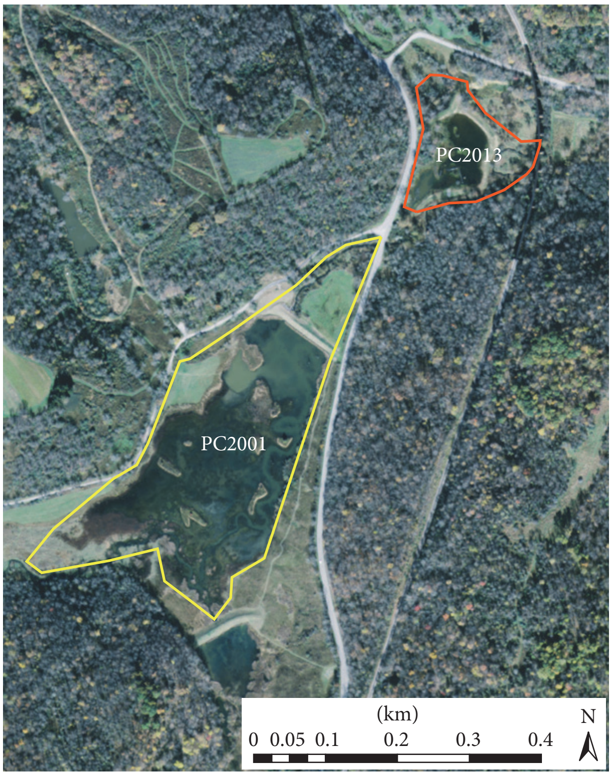

Figure 2: Aerial view of the wetland created in 2001 (PC2001) and the wetland created in 2013 (PC2013). PC2001 is outlined in yellow, while PC2013 is outlined in orange.

started and ended, air temperature, and percent ice cover (i.e., the percentage of wetland area covered by ice). Ice cover was tested as a possible explanatory factor for no waterbird detection during the winter surveys.

2.3. Comparing Waterbird Communities. To analyze the data and assess the ability of PC2013 to provide waterbird habitat in comparison to the older PC2001, we compared overall, annual, and monthly waterbird species richness, diversity, composition, and use (dependent variables) at each wetland (independent variable). Because we could not confidently identify when individual waterbirds used the wetland multiple days, species use was quantified as the highest species count of each week's surveys (e.g., if 2 *Anas platyrhynchos* L. [Mallards] were seen during one survey and 5 were seen during a second survey within the same week, the quantity of 5 Mallards would be used). Species richness was compared using single-factor analysis of variance (ANOVA) with an alpha level of 0.05. For significant results, we further divided species richness by the area of the wetland (ha) and performed ANOVA analyses again to compensate for size differences. Species diversity was calculated using the Shannon-Wiener Diversity Index and compared using single-factor ANOVA. Percent species composition was determined by dividing the total number of an individual species by the total amount of individuals detected within the time period (e.g., month, year), and then it was compared using Schoener's Index to determine percent overlap of communities and *G*-tests. Species use was designated as waterbirds per ha and then compared using single-factor ANOVA.

Variation between wetlands was compared using overall metrics that were combined from the entire survey period. The means from the ANOVA tests were derived from monthly count data (e.g., overall species diversity for PC2001 was calculated using the monthly diversity values from November to March 2013-2014 and 2014-2015 data) and weekly high species counts (e.g., for comparing species use). Variation between wetlands by year was compared using annual metrics. Thus, the means from the ANOVA tests were derived from values from the monthly count data (e.g., annual species diversity for PC2001 in 2013-2014 was calculated using the diversity values from November 2013, December 2013, January 2014, February 2014, and March 2014) and weekly high species counts. Variation between wetlands by month was compared using average monthly metrics that were derived from the weekly surveys conducted within that month. *G*-tests were used to compare the species composition of both wetlands for each month in the 2 survey periods. To determine differences in monthly species use, ANOVA was used to compare average weekly high species counts. Each statistical test was then run through the sequential Bonferroni approach to minimize type I errors [31].

In addition, we examined trends in waterbird use by creating use curves (waterbirds/ha plotted against time) for waterbird species that comprised at least 2.0% of the species composition at either wetland during the 2 years of surveys. Monthly use was calculated by averaging the weekly high species counts and dividing by the number of hectares encompassed by the corresponding wetland. Average waterbird use per ha was plotted for each month in the study. The use curves allow us to determine when waterbirds are using the 2 wetlands and visualize in which months use is similar or diverges.

3. Results

3.1. Survey Data. During November to March 2013-2014, we conducted 127 surveys and observed 1,831 waterbirds ($n = 1{,}749$ at PC2001, $n = 82$ at PC2013) belonging to 23 species ($n = 23$ at PC2001, $n = 7$ at PC2013). The average temperature and percent ice cover were 1.8°C and 54%, respectively. In the following year (November to March 2014-2015), we conducted 121 surveys and observed 1,509 waterbirds ($n = 1{,}309$ at PC2001, $n = 200$ at PC2013) belonging to 24 species ($n = 24$ at PC2001, $n = 10$ at PC2013). The average temperature was similar to the previous year (1.2°C), while average percent ice cover was slightly lower (49%).

Combining the 2 years, a grand total of 248 surveys (124 surveys at each wetland) were conducted from November 2013 to March 2015, and 3,340 waterbirds ($n = 3{,}058$ at PC2001, $n = 282$ at PC2013) belonging to 27 species ($n = 27$ at PC2001, $n = 11$ at PC2013) were observed. Common disturbances at the 2 wetlands during both years included the presence of hunters and noise from a nearby shooting range and railroad trestle. Average temperatures during surveys across the 2 years ranged from −3.7°C to 4.6°C, and average ice cover was approximately 50%. The proportion of surveys in which no waterbirds were observed was similar between years (40%).

3.2. Variation between Wetlands. Overall average species richness initially appeared higher at PC2001, but further analyses to compensate for the effects of wetland size revealed no significant difference ($P = 0.108$). Species diversity did not significantly differ either (Table 1). There was a 33.0% overlap in species composition between the 2 wetlands. PC2001 had a significantly higher percent composition of *Bucephala albeola* L. (Bufflehead; PC2001 = 10.73%, PC2013 = 0.71%, $G_1 = 10.5$, $P < 0.005$), *Anas strepera* L. (Gadwall; PC2001 = 7.03%, PC2013 = 0.0%, $G_1 = 9.7$, $P < 0.005$), *Aythya affinis* (Eyton) (Lesser Scaup; PC2001 = 8.24%, PC2013 = 0.0%, $G_1 = 11.4$, $P < 0.001$), and *Aythya collaris* (Donovan) (Ring-Necked Duck; PC2001 = 17.5%, PC2013 = 0.0%, $G_1 = 24.3$, $P < 0.001$), while PC2013 had a greater percent composition of Mallard (PC2001 = 4.74%, PC2013 = 23.40%, $G_1 = 13.5$, $P < 0.001$) and *Aix sponsa* L. (Wood Duck; PC2001 = 1.86%, PC2013 = 33.33%, $G_1 = 34.2$, $P < 0.001$). The percent composition of the other 21 species was similar between the 2 wetlands ($P > 0.05$).

Average total species use over the course of the 2 survey periods was higher at PC2001. Three of the 27 species had significant differences in use. Buffleheads, *Branta canadensis* L. (Canada Goose), and Gadwall all had higher use values at PC2001 (Table 1). Furthermore, the proportion of surveys in which no waterbirds were observed was greater at PC2013 (PC2001: 0.23, PC2013: 0.58, $P < 0.05$), but there was no difference between the average percent ice cover ($P > 0.05$).

3.3. Variation between Wetlands by Year. Average species richness and average species diversity were not significantly different during either year (Table 1). Both average species richness and diversity values for PC2013 increased slightly in the second winter. There was a 42.4% overlap in species composition between the 2 wetlands during 2013-2014 and a 28.7% overlap during 2014-2015. During the first winter period, PC2001 had a significantly higher percent composition of Bufflehead (PC2001 = 10.92%, PC2013 = 0.0%, $G_1 = 15.1$, $P < 0.001$), Gadwall (PC2001 = 10.52%, PC2013 = 0.0%, $G_1 = 14.6$, $P < 0.001$), Lesser Scaup (PC2001 = 10.41%, PC2013 = 0.0%, $G_1 = 14.4$, $P < 0.001$), and Ring-Necked Duck (PC2001 = 17.90%, PC2013 = 0.0%, $G_1 = 24.8$, $P < 0.001$), while PC2013 had a greater percent composition of *Lophodytes cucullatus* L. (Hooded Merganser; PC2001 = 2.74%, PC2013 = 25.61%, $G_1 = 21.3$, $P < 0.001$) and Wood Duck (PC2001 = 2.06%, PC2013 = 20.73%, $G_1 = 17.8$, $P < 0.001$). The percent composition of the other 16 waterbird species was similar between the 2 wetlands ($P > 0.05$). During the second winter, PC2001 again had significantly higher percent compositions of Bufflehead (PC2001 = 10.47%, PC2013 = 1.0%, $G_1 = 9.1$, $P < 0.005$) and Ring-Necked Duck (PC2001 = 16.96%, PC2013 = 0.0%, $G_1 = 23.5$, $P < 0.001$), along with Canada Goose (PC2001 = 33.77%, PC2013 = 13.0%, $G_1 = 9.6$, $P < 0.005$). Meanwhile, PC2013 had significantly higher percent compositions of *Anas rubripes* (Brewster) (American Black Duck; PC2001 = 1.30%, PC2013 = 12.50%, $G_1 = 10.5$, $P < 0.005$), Mallard (PC2001 = 6.72%, PC2013 = 27.0%, $G_1 = 13.1$, $P < 0.001$), and Wood Duck (PC2001 = 1.60%, PC2013 = 38.50%, $G_1 = 42.1$, $P < 0.001$). The other 16 waterbird species were similar in composition between the wetlands ($P > 0.05$).

There was no significant difference in average total species use between the 2 wetlands in either year (Table 1). Average total species use appeared to increase at PC2013 from the first winter to the second. Of the 23 species observed in 2013-2014, there were no significant differences in the average species use. The proportion of surveys in which no waterbirds were observed was greater at PC2013 (PC2001: 0.18, PC2013: 0.63, $P < 0.05$) during the first winter, but not during the second. There was no difference in average percent ice cover at the 2 wetlands during either year ($P > 0.05$).

3.4. Variation between Wetlands by Month. Species richness was originally found to be significantly greater at PC2001 in November 2013 (PC2001: 5.50 ± 0.87, PC2013: 0.25 ± 0.25, $F_{1,6} = 33.92$, $P = 0.001$), March 2014 (PC2001: 11.0 ± 1.41, PC2013: 2.60 ± 0.51, $F_{1,8} = 31.22$, $P < 0.001$), and March 2015 (PC2001: 11.40 ± 0.40, PC2013: 3.20 ± 0.58, $F_{1,8} = 134.5$, $P < 0.001$). However, those results were found to be insignificant when species richness was quantified as the average number of species per ha per week, which compensates for differences in the 2 wetlands' areas. The difference in species richness was insignificant in all other months ($P > 0.05$). Species diversity followed a similar trend. Diversity was higher at PC2001 in November 2013 (PC2001: 1.173 ± 0.16, PC2013: 0.0 ± 0.0, $F_{1,6} = 54.78$, $P < 0.001$) and March 2015 (PC2001: 1.775 ± 0.04, PC2013: 0.942 ± 0.0, $F_{1,8} = 25.49$, $P = 0.001$). Diversity values of both wetlands in all other months were not significant ($P > 0.05$). The proportion of surveys during which no waterbirds were observed tended to be greater for PC2013 than for PC2001. Average percent ice cover similarly tended to be greater at PC2013, though there was a differences of less than 13% between the 2 wetlands in all months. Ice cover and the proportion of surveys without waterbirds were correlated at PC2001 ($R^2 = 0.71$, $P = 0.002$) but not at PC2013 ($R^2 = 0.22$, $P = 0.17$).

Species composition varied between wetlands and among months (Tables 2 and 3). PC2001 tended to have significantly higher percent compositions of *Fulica americana* (Gmelin) (American Coot), Bufflehead, Canada Goose, and Gadwall, while PC2013 tended to have a higher percent composition of Hooded Merganser and Wood Duck. Individual species use did not differ in any month during the first winter ($P > 0.05$), but total species use in March 2014 was greater at PC2001 (PC2001: 10.19 ± 1.10, PC2013: 1.93 ± 0.39, $F_{1,8} = 50.29$, $P = 0.0001$) (see Supplementary Data Table in Supplementary Material available online at https://doi.org/10.1155/2017/1730130). In addition, the number of observed species was almost four times higher at PC2001 ($n = 19$; for PC2013, $n = 5$) in March 2014. In the second winter, there were no differences in average individual or total species use from November 2014 to February 2015 ($P > 0.05$). In March 2015, total species use was not significantly different, but use by

Table 1: Summary of overall and annual waterbird species use, richness, and diversity with their means and standard errors at 2 created wetlands (PC2001 and PC2013) in Pleasant Creek WMA, WV, from November to March 2013-2014 and 2014-2015. Individual and total species use was calculated using weekly high species counts per ha, while richness and diversity were calculated using total waterbird counts. Bolded means are significant following the use of the sequential Bonferroni approach. Italicized means indicate a result that was rendered insignificant after compensating for differences in wetland size.

Common name	PC2001		PC2013		*P*	2013-14 PC2001		2013-14 PC2013		*P*	2014-15 PC2001		2014-15 PC2013		*P*
	Mean	SE	Mean	SE		Mean	SE	Mean	SE		Mean	SE	Mean	SE	
American Black Duck	0.032	0.02	0.185	0.15	0.313	0.026	0.01	0.000	0.00	0.066	0.038	0.03	0.362	0.29	0.274
American Coot	0.100	0.03	0.007	0.01	0.003	0.053	0.02	0.015	0.02	0.146	0.145	0.05	0.000	0.00	0.009
American Wigeon	0.019	0.01	0.000	0.00	0.156	0.026	0.03	0.000	0.00	0.323	0.013	0.01	0.000	0.00	0.155
Belted Kingfisher	0.031	0.01	0.022	0.01	0.555	0.023	0.01	0.015	0.02	0.653	0.038	0.01	0.029	0.02	0.694
Bufflehead	**0.358**	**0.11**	**0.015**	**0.01**	**0.002**	0.416	0.19	0.000	0.00	0.034	0.303	0.11	0.029	0.03	0.024
Blue-Winged Teal	0.016	0.01	0.000	0.00	0.055	0.023	0.01	0.000	0.00	0.102	0.009	0.01	0.000	0.00	0.323
Cackling Goose	0.005	0.00	0.000	0.00	0.179	0.003	0.00	0.000	0.00	0.323	0.006	0.01	0.000	0.00	0.323
Canada Goose	**1.132**	**0.27**	**0.178**	**0.04**	**0.001**	1.217	0.41	0.182	0.06	0.018	1.051	0.35	0.174	0.06	0.017
Canvasback	0.048	0.02	0.000	0.00	0.010	0.056	0.02	0.000	0.00	0.021	0.041	0.03	0.000	0.00	0.156
Common Merganser	0.006	0.00	0.007	0.01	0.912	—	—	—	—	—	0.013	0.01	0.014	0.01	0.912
Gadwall	**0.211**	**0.06**	**0.000**	**0.00**	**0.001**	0.360	0.12	0.000	0.00	0.003	0.069	0.04	0.000	0.00	0.125
Great Egret	0.003	0.00	0.000	0.00	0.320	—	—	—	—	—	0.006	0.01	0.000	0.00	0.323
Greater Scaup	0.034	0.02	0.000	0.00	0.067	0.036	0.03	0.000	0.00	0.235	0.032	0.02	0.000	0.00	0.155
Great Blue Heron	0.045	0.01	0.030	0.01	0.426	0.053	0.02	0.030	0.02	0.484	0.038	0.01	0.029	0.02	0.705
Green-Winged Teal	0.019	0.01	0.000	0.00	0.106	—	—	—	—	—	0.038	0.02	0.000	0.00	0.103
Horned Grebe	0.002	0.00	0.000	0.00	0.320	0.003	0.00	0.000	0.00	0.323	—	—	—	—	—
Hooded Merganser	0.089	0.02	0.170	0.06	0.191	0.125	0.05	0.258	0.10	0.249	0.054	0.02	0.087	0.05	0.516
Lesser Scaup	0.263	0.10	0.000	0.00	0.007	0.393	0.18	0.000	0.00	0.037	0.139	0.06	0.000	0.00	0.030
Lesser Yellowlegs	0.005	0.00	0.000	0.00	0.320	—	—	—	—	—	0.009	0.01	0.000	0.00	0.323
Mallard	0.155	0.03	0.415	0.17	0.135	0.122	0.04	0.152	0.10	0.781	0.186	0.05	0.667	0.31	0.136
Northern Pintail	0.010	0.01	0.000	0.00	0.179	0.020	0.01	0.000	0.00	0.179	—	—	—	—	—
Pied-Billed Grebe	0.090	0.02	0.015	0.01	0.005	0.063	0.03	0.000	0.00	0.020	0.117	0.04	0.030	0.02	0.074
Redhead	0.008	0.01	0.000	0.00	0.226	0.013	0.01	0.000	0.00	0.323	0.003	0.00	0.000	0.00	0.323
Ring-Necked Duck	0.537	0.19	0.000	0.00	0.006	0.614	0.28	0.000	0.00	0.033	0.464	0.27	0.000	0.00	0.089
Ruddy Duck	0.007	0.00	0.000	0.00	0.041	0.010	0.01	0.000	0.00	0.076	0.003	0.00	0.000	0.00	0.323
Tundra Swan	0.015	0.01	0.000	0.00	0.208	0.030	0.02	0.000	0.00	0.208	—	—	—	—	—
Wood Duck	0.063	0.02	0.430	0.14	0.011	0.086	0.04	0.182	0.08	0.261	0.041	0.03	0.667	0.26	0.020
Total species use	**3.283**	**0.60**	**1.030**	**0.24**	**0.001**	3.770	0.90	0.833	0.19	0.003	2.818	0.81	2.087	0.65	0.483
Species richness	***10.1***	***1.68***	***3.2***	***0.47***	***0.001***	10.8	2.48	2.8	0.66	0.014	9.4	2.50	3.6	0.68	0.056
Species diversity	1.441	0.18	0.831	0.13	0.013	1.448	0.29	0.771	0.22	0.10	1.435	0.25	0.890	0.15	0.099

Table 2: Summary of significant differences in monthly waterbird species percent composition at 2 created wetlands (PC2001 and PC2013) in Pleasant Creek WMA, WV, from November to March 2013-2014. All of the following results are significant following the use of the sequential Bonferroni approach.

Month	% composition overlap	Species	PC2001	PC2013	G_1	P
November 2013	2.7	American Coot	7.59	0.00	10.5	<0.005
		Bufflehead	17.41	0.00	24.1	<0.001
		Canada Goose	39.73	0.00	55.1	<0.001
		Gadwall	8.48	0.00	11.8	<0.001
		Hooded Merganser	2.68	100.0	117.5	<0.001
		Mallard	6.70	0.00	9.3	<0.005
		Pied-Billed Grebe	8.93	0.00	12.4	<0.001
December 2013	8.3	Belted Kingfisher	3.18	50.0	49.6	<0.001
		Gadwall	76.43	0.00	106.0	<0.001
		Great Blue Heron	5.10	50.0	42.4	<0.001
		Ruddy Duck	3.18	0.00	11.5	<0.001
		Tundra Swan	8.28	0.00	11.5	<0.001
January 2014	0.0	American Coot	0.00	11.11	15.4	<0.001
		Canada Goose	87.37	0.00	121.1	<0.001
		Mallard	0.00	66.67	92.4	<0.001
		Wood Duck	0.00	22.22	30.8	<0.001
February 2014	54.5	Bufflehead	12.20	0.00	16.9	<0.001
		Hooded Merganser	21.14	50.0	12.1	<0.001
		Mallard	0.00	16.67	23.1	<0.001
		Ring-Necked Duck	13.01	0.00	18.0	<0.001
March 2014	35.4	Bufflehead	11.91	0.00	16.5	<0.001
		Hooded Merganser	1.04	18.52	19.0	<0.001
		Lesser Scaup	15.83	0.00	21.9	<0.001
		Ring-Necked Duck	25.74	0.00	35.7	<0.001
		Wood Duck	2.00	27.78	26.6	<0.001

Table 3: Summary of significant differences in monthly waterbird species percent composition at 2 created wetlands (PC2001 and PC2013) in Pleasant Creek WMA, WV, from November to March 2014-2015. All of the following results are significant following the use of the sequential Bonferroni approach.

Month	% composition overlap	Species	PC2001	PC2013	G_1	P
November 2014	20.8	American Coot	16.67	0.00	23.1	<0.001
		Belted Kingfisher	0.83	13.33	13.3	<0.001
		Mallard	11.67	0.00	16.2	<0.001
		Pied-Billed Grebe	45.0	13.33	18.2	<0.001
		Wood Duck	0.00	60.00	83.2	<0.001
December 2014	12.8	Canada Goose	57.05	0.00	79.1	<0.001
		Mallard	11.54	0.00	16.0	<0.001
		Wood Duck	0.00	83.33	115.5	<0.001
January 2015	25.0	Canada Goose	66.04	0.00	91.6	<0.001
		Mallard	75.00	0.00	104.0	<0.001
February 2015	63.0	American Black Duck	7.69	41.07	25.1	<0.001
		Hooded Merganser	19.23	0.00	26.7	<0.001
March 2015	29.0	American Coot	6.50	0.00	9.0	<0.005
		Bufflehead	12.16	0.00	16.9	<0.001
		Lesser Scaup	7.34	0.00	10.2	<0.005
		Ring-Necked Duck	23.17	0.00	32.1	<0.001
		Wood Duck	2.20	53.20	58.3	<0.001

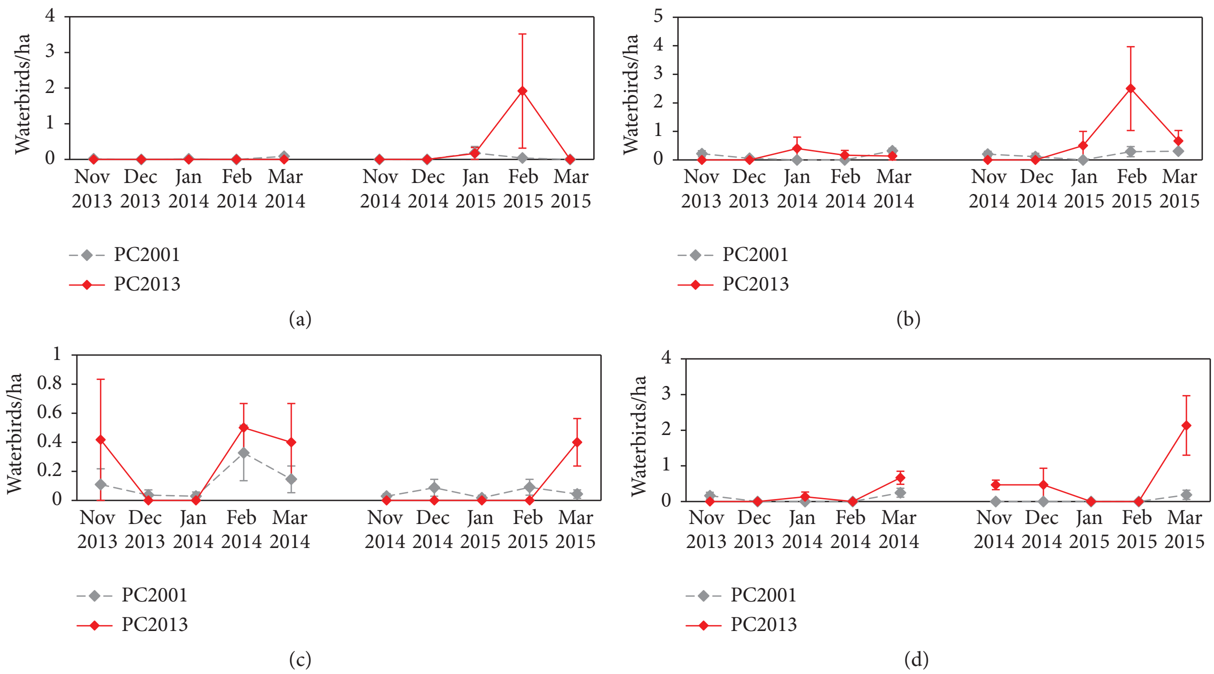

Figure 3: Average American Black Duck (a), Mallard (b), Hooded Merganser (c), and Wood Duck (d) use per ha at PC2001 and PC2013 from November to March 2013-2014 and 2014-2015.

Bufflehead (PC2001: 1.09 ± 0.02, PC2013: 0.0 ± 0.0, $F_{1,8}$ = 29.22, P = 0.0006), Canada Goose (PC2001: 3.50 ± 0.72, PC2013: 0.667 ± 0.0, $F_{1,8}$ = 15.62, P = 0.004), and Lesser Scaup (PC2001: 0.64 ± 0.13, PC2013: 0.0 ± 0.0, $F_{1,8}$ = 23.19, P = 0.001) was higher at PC2001.

3.5. Trends in Waterbird Use. Eleven waterbird species (American Black Duck, American Coot, Bufflehead, Canada Goose, Gadwall, Hooded Merganser, Lesser Scaup, Mallard, Pied-Billed Grebe, Ring-Necked Duck, and Wood Duck) comprised at least 2.0% at either wetland during the 2 years of surveys. Each species differed slightly in patterns of monthly use, which was calculated by averaging the weekly high counts. American Black Ducks, Mallards, Hooded Merganser, and Wood Ducks tended to have higher use at PC2013 (Figure 3). American Black Ducks and Mallards had similar use curves, with a distinct peak in use of PC2013 during February 2015. Hooded Mergansers used the wetlands differently in the first and second years of the study. During the first winter, use was highest in PC2013 in November 2013, February 2014, and March 2014, but in the second year, use was high only in March 2015. In both years, Wood Ducks had the highest use in both wetlands in March.

American Coot and Pied-Billed Grebes had higher uses at PC2001 in November and March of both years (Figure 4). Canada Geese and Gadwall tended to increase use in March. During the first winter, Gadwall use peaked in December 2013, but that trend was not repeated the second winter. Lesser Scaup and Ring-Necked Duck had similar use curves (Figure 5). Few to no ducks were detected until March, when use was highest both years. Similarly, Bufflehead use was highest in March of both years, though they also used PC2001 in November 2013 and both wetlands in December 2014.

4. Discussion

4.1. Waterbird Diversity and Abundance. To our knowledge, this was the first study to evaluate winter waterbird use of differently aged created wetlands in the Central Appalachians. Over the two 5-month periods during our study, we observed the wetlands harboring 3,340 waterbirds belonging to 27 species. They provided food and habitat to a diversity of migratory and wintering waterbirds, which in turn contributed to regional biodiversity and recreational hunting opportunities. At the wetland scale, winter waterbird species richness was not significantly different when wetland area was compensated for. Similarly, average species richness was not significantly different between the 2 wetlands in individual years, though both average species richness and diversity values for PC2013 increased slightly in the second winter, which may indicate that those metrics will increase over time. The greatest disparities in species richness and diversity tended to occur in November 2013, March 2013, and March 2014, while the least disparities occurred in January and February 2013 and 2014. These trends indicate that PC2013 may not attract as many waterbird species during migration (November and March) as PC2001.

The overlap in species composition ranged from 0% in January 2014 to 63% in February 2015, with an average of 25% across months. Differences in species composition were greatest from November to December 2013 and least in January and February 2015. Percent overlap increased or

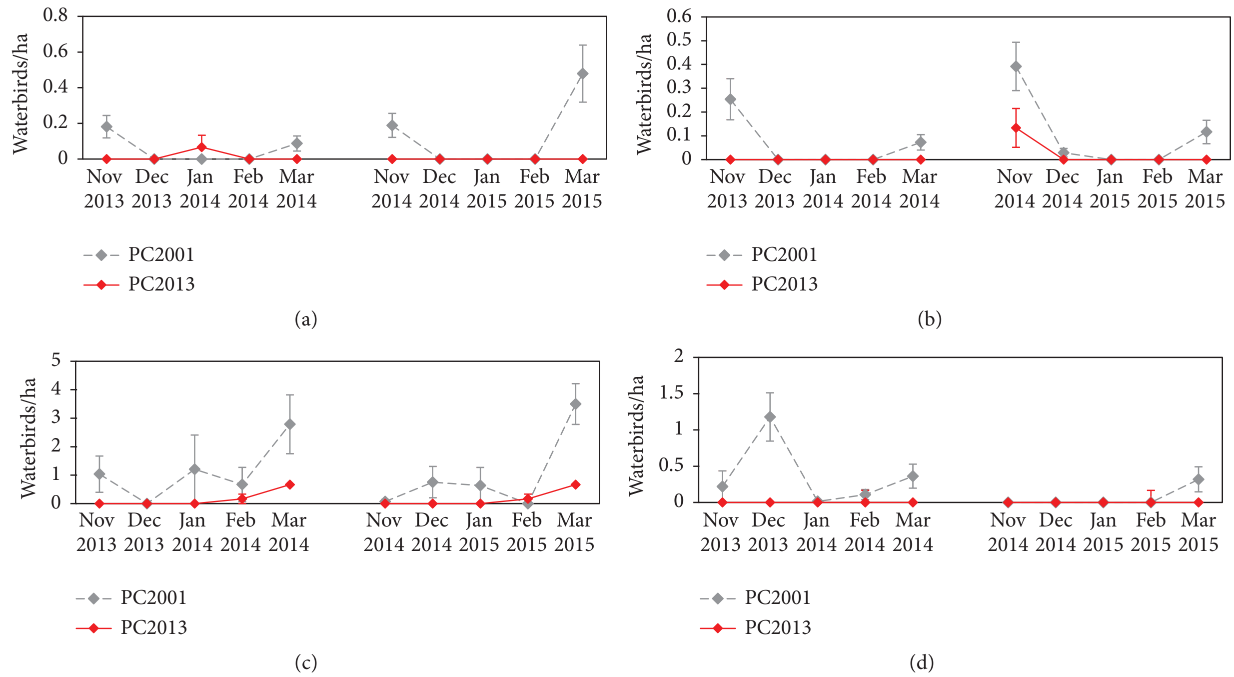

Figure 4: Average American Coot (a), Pied-Billed Grebe (b), Canada Goose (c), and Gadwall (d) use per ha at PC2001 and PC2013 from November to March 2013-2014 and 2014-2015.

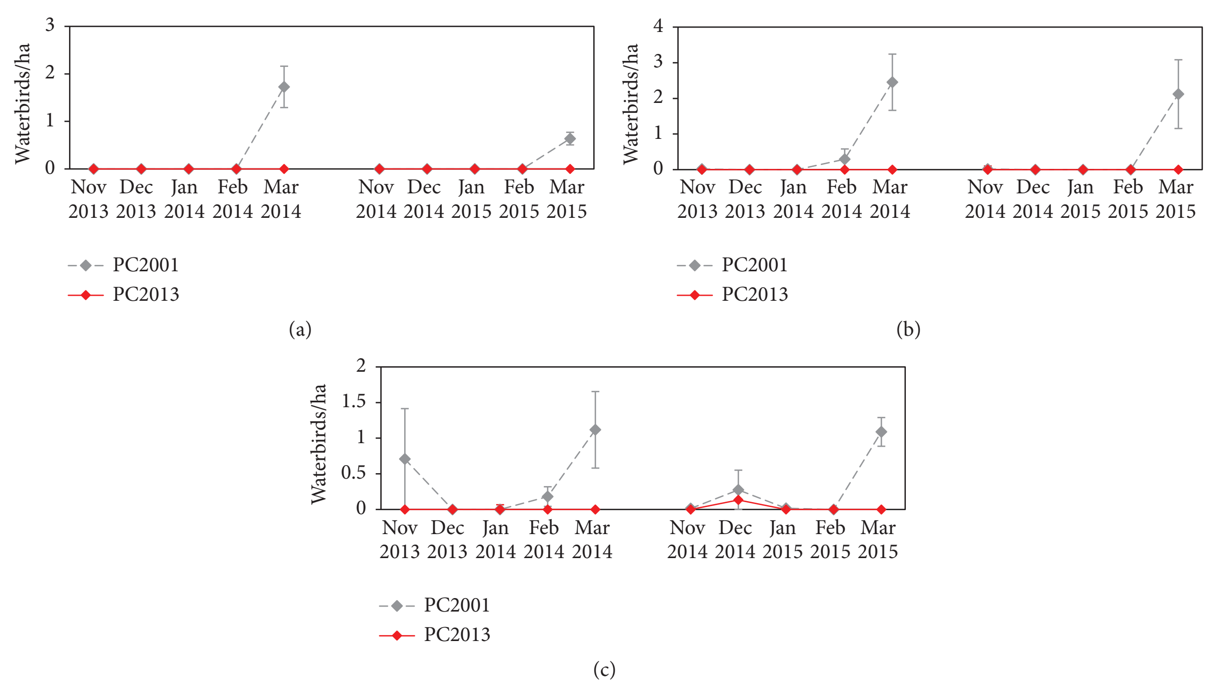

Figure 5: Average diving duck (Lesser Scaup (a), Ring-Necked Duck (b), and Bufflehead (c)) use per ha at PC2001 and PC2013 from November to March 2013-2014 and 2014-2015.

stayed similar in months from the first winter to the second winter. PC2001 generally had higher percent composition of American Coot, Bufflehead, Canada Goose, Gadwall, Lesser Scaup, and Ring-Necked Duck.

Certain waterbirds, including Buffleheads, Canada Geese, Gadwall, Lesser Scaup, and Ring-Necked Duck, tended to have higher use at PC2001. Differences in use were highest in March 2015. Average total species use tended to be higher at PC2001, though there was generally no statistical difference between the wetlands. The difference in total species use was less distinct in the second winter, which may indicate increased habitat availability or quality at PC2013 as the recently created wetland developed and matured. Because waterfowl exhibit site philopatry in the winter and are known to explore new sites, it is also possible that individuals that wintered at PC2013 the first year would come back again the next year and that additional individuals would discover the new wetland, increasing overall use and abundance [32].

Trends in waterbird use were variable. For many waterbirds, particularly diving ducks, use was highest in March during migration. With some exceptions, mid-winter waterbird use was limited. Based on the species richness, diversity, composition, and use results, it appears that the ability of PC2013 is similar to PC2001 in providing habitat for wintering waterbirds but not migrating waterbirds. Furthermore, PC2013 provides different habitat types that appear to favor a different winter waterbird community than PC2001, as its community composition tends to comprise Hooded Mergansers, Mallards, and Wood Ducks. For instance, American Black Ducks, Mallards, and Wood Ducks were observed using the stream within the PC2013 wetland complex when the wetland itself was covered in ice, and Wood Ducks were often found in the upper portion of the wetland, which is interspersed with trees and shrubs.

Certain disparities in species diversity, composition, and use might be explained by differences in wetland area, water depth, microtopography, and vegetative cover. Wetland size can predict waterbird richness and species abundance [33, 34]. We compensated for wetland size differences by dividing species richness and use by the area of the respective wetland. However, larger wetlands tend to have more variable spatial configurations and higher habitat heterogeneity [35]. Concomitantly, these larger wetlands can support a greater diversity of waterbirds with different habitat preferences [1, 9, 36, 37]. For instance, waterbird species that forage in open or deepwater habitats are restricted to relatively large wetlands or ponds [35]. Small wetlands are generally associated with lower species diversity [33]. In our study, both wetlands were relatively similar in habitat on a landscape scale. At a smaller spatial scale, it is possible that the larger PC2001 contained more microhabitat variability. Furthermore, PC2001 had a higher percentage of core habitat (39.2%) than PC2013 (5.3%), due to its size and configuration. With possibly higher microhabitat diversity and more core area, it is reasonable that the larger PC2001 experienced greater species use than PC2013.

Water depth and microtopography also play a role in shaping waterbird communities. Many studies cite water depth as an important variable that affects waterbird use of wetland habitats [9, 38, 39]. Water depth has an impact on waterbirds due to their morphology and feeding habits [40, 41]. Wading and dabbling waterbirds generally require shallow water to forage, and water depth limits their access to foraging habitat [41]. Small shorebirds use water depths of <5 cm, large shorebirds use 5–11 cm, and large dabbling ducks use >20 cm [39]. Diving waterbirds require a minimum depth of >25 cm and can forage in water that is several meters deep [41]. The numbers of waterbird, dabbling duck, and wading bird species tend to increase in shallow wetlands, while the number of diving duck species increases in deeper wetlands [9]. In corroboration with our results, Colwell and Taft [9] found that Gadwall and American Coot tended to occur at higher densities in deep wetlands. In addition, diving ducks such as Bufflehead, Lesser Scaup, and Ring-Necked Duck had higher percent species composition and use at PC2001, which was deeper than PC2013.

Though we were unable to directly discern the effects of age on differences between the 2 wetlands, we can comment on possible indirect effects. As a wetland ages and matures, habitat availability or quality may increase. Wetland vegetation, another major factor that influences waterbird use of wetlands, can become established and flourish over time. Vegetation and habitat heterogeneity are related [42]. Froneman et al. [36] found that structural diversity of vegetation in and around farm ponds was important in determining usage by waterbirds. However, too dense vegetative cover can decrease waterbird use [40]. Vegetation tends to be scarce in winter, the time period of our study, but seeds are an important food resource, and trees are used by species such as *Ardea herodias* L. (Great Blue Herons) and Belted Kingfishers. Craig and Beal [42] posit that marshes of the same size with similar habitat conditions should attract similar species of birds. Thus, the differences in species composition at PC2001 and PC2013 are likely due to varying water depths and divergent habitat variables, possibly mediated by age.

4.2. Related Studies. The results of our study were somewhat similar to other previously conducted in the region. Balcombe et al. [19] evaluated breeding avian and anuran communities in 11 mitigation and 4 reference wetlands throughout West Virginia. They found that Wood Ducks were more abundant in mitigation wetlands, while the density of Great Blue Herons was similar between wetland types. In addition, they found that waterbird and waterfowl abundance were higher in mitigation wetlands than reference wetlands. Balcombe et al. [20] attempted to determine if mitigation wetlands in West Virginia were adequately supporting ecological communities relative to naturally occurring reference wetlands and to attribute specific characteristics in wetland habitat with trends in wildlife abundance across wetlands. They found that abundance of waterbirds at mitigated wetlands was affected by age, benthic invertebrate diversity, percent emergent vegetation, percent open water, size, and vegetation diversity. Furthermore, Balcombe et al. [20] ranked mitigation wetlands consistently higher than reference wetlands.

However, studies evaluating created versus natural wetland function vary in their results. Outside of the Central Appalachian region, another study investigated avian communities in created and natural wetlands in Virginia. Desrochers et al. [24] tested the hypothesis that created wetlands provide avian habitat lost via wetland destruction by comparing breeding and wintering birds on 11 small created salt marshes with those on 11 natural reference salt marshes. They found that created salt marshes had lower avian abundance and richness than reference salt marshes during the breeding season. However, observed bird use outside of the breeding season did not differ. Desrochers et al. [24] concluded that the created wetlands they surveyed failed to completely replicate the bird communities observed on nearby natural reference salt marshes. Another study in

Virginia assessed ecological conditions in a created tidal marsh and 2 natural reference tidal marshes. Havens et al. [43] found that bird species richness and diversity were similar among the created and natural marshes, and wading birds appeared to show a significant preference for the created marsh. Further to the south, White and Main [29] studied waterbird use of created wetlands in golf-course landscapes in Florida. They found that created golf-course ponds were capable of attracting various species of waterbirds. However, they suggested that the value of golf-course ponds may be enhanced through modifications to the vegetation and hydrology designed to appeal to specific waterbird guilds. In New York, Brown and Smith [44] looked at breeding season bird use of recently restored and natural wetlands. They compared the relative abundance and density of birds using 18 restored wetlands and 8 natural wetlands. Abundances of species did not differ between restored and natural wetlands in any year, but densities were consistently lower at restored sites and bird communities were significantly less similar between restored and natural sites than among restored sites. Brown and Smith [44] conclude that the restoration program successfully increased the amount of bird habitat available in the region, but the restored wetland sites did not entirely replace the habitat functions of natural wetlands during their study. Finally, though they did not specifically survey for waterbirds, Confer and Niering [45] assessed wildlife in created and natural wetlands in Connecticut and observed higher wildlife activity in the natural wetlands.

Similar to the findings of White and Main [29], the 2 created wetlands in our study proved capable of attracting many species of waterbirds. Balcombe et al. [19] indicated that mitigated wetlands in West Virginia were able to support various wildlife species. It is possible that PC2013 will increase in its ability to provide habitat for waterbirds as time progresses. Native wetland plant species diversity and richness have been found to increase with wetland age, which may increase the attractiveness of habitat to waterbirds [46]. Furthermore, average total species use, richness, and diversity of waterbirds using PC2013 seemed to increase from the first winter to the next. Though the community compositions of PC2013 and PC2001 are unlikely to ever be identical due to variation in certain habitat features, species use of PC2013 may continue to increase in coming years.

4.3. Management Implications. Our analyses were designed to quantify the differences and changes in the waterbird communities of 2 wetlands of different ages during 2 winter seasons. Though the results of this study are limited in direct applications to the development and management of PC2013, they provide insight into the potential impacts of newly created wetland habitat on local and migrant waterbird species in the Central Appalachians during the nonbreeding season. Our results further highlight important factors of wetland construction that must be taken into account. When designing and creating wetlands, it is important to consider management objectives, wetland size, water depth, topography, and vegetation. Wetland size and water depth influence habitat diversity and waterbird use [9, 36]. To maximize species richness and diversity of wintering waterbirds, managers should ensure that the water depth of wetlands is an average of 10–20 cm, with a range of depths that will attract a large number of species [9]. Larger wetlands tend to support more waterbirds, but small wetlands that are used seasonally by waterbirds can still be important in maintaining local and regional populations [42]. Finally, structural diversity in vegetation as well as the growth and seed production of beneficial wetlands plants should be promoted to provide cover and food for waterbirds.

Conflicts of Interest

The authors declare that there are no conflicts of interest regarding the publication of this paper.

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References

[1] L.-A. Hansson, C. Brönmark, P. A. Nilsson, and K. Åbjörnsson, "Conflicting demands on wetland ecosystem services: nutrient retention, biodiversity or both?" *Freshwater Biology*, vol. 50, no. 4, pp. 705–714, 2005.

[2] D. L. McLaughlin and M. J. Cohen, "Realizing ecosystem services: wetland hydrologic function along a gradient of ecosystem condition," *Ecological Applications*, vol. 23, no. 7, pp. 1619–1631, 2013.

[3] R. Costanza, R. D'Arge, R. De Groot et al., "The value of the world's ecosystem services and natural capital," *Nature*, vol. 387, no. 6630, pp. 253–260, 1997.

[4] S. L. Melvin and J. W. Webb Jr., "Differences in the avian communities of natural and created *Spartina alterniflora* salt marshes," *Wetlands*, vol. 18, no. 1, pp. 59–69, 1998.

[5] J. D. Williams and C. K. Dodd Jr., "Importance of wetlands to endangered and threatened species," in *Wetland Functions and Values: The State of Our Understanding*, P. E. Greeson, J. R. Clark, and J. E. Clark, Eds., pp. 565–575, American Water Resources Association, Minneapolis, Minn, USA, 1978.

[6] N. A. Murdock, "Rare and endangered plants and animals of southern Appalachian wetlands," in *Wetlands of the Interior Southeastern United States*, C. C. Trettin, W. M. Aust, and J. Wisniewski, Eds., Springer, Dordrecht, The Netherlands, 1995.

[7] J. T. Anderson and L. M. Smith, "Carrying capacity and diel use of managed playa wetlands by nonbreeding waterbirds," *Wildlife Society Bulletin*, vol. 27, no. 2, pp. 281–291, 1999.

[8] O. W. Taft and S. M. Haig, "Historical wetlands in Oregon's Willamette Valley: implications for restoration of winter waterbird habitat," *Wetlands*, vol. 23, no. 1, pp. 51–64, 2003.

[9] M. A. Colwell and O. W. Taft, "Waterbird communities in managed wetlands of varying water depth," *Waterbirds*, vol. 23, no. 1, pp. 45–55, 2000.

[10] R. W. Stewart, "Wetlands as bird habitat," U.S. Geological Survey Water Supply Paper 2425, U.S. Geological Survey, Washington, DC, USA, 2014.

[11] L. D. Igl and D. H. Johnson, "Changes in breeding bird populations in North Dakota: 1967 to 1992-93," *The Auk*, vol. 114, no. 1, pp. 74–92, 1997.

[12] T. E. Dahl, *Wetland Losses in the United States 1780's to 1980's*, U.S. Department of the Interior, Fish and Wildlife Service, Washington, DC, USA, 1990.

[13] T. E. Dahl, *Status and Trends of Wetlands in the Conterminous United States 2004 to 2009*, U.S. Department of the Interior, Fish and Wildlife Service, Washington, DC, USA, 2011.

[14] R. T. Gingerich and J. T. Anderson, "Litter decomposition in created and reference wetlands in West Virginia, USA," *Wetlands Ecology and Management*, vol. 19, no. 5, pp. 449–458, 2011.

[15] R. T. Gingerich, D. G. Panaccione, and J. T. Anderson, "The role of fungi and invertebrates in litter decomposition in mitigated and reference wetlands," *Limnologica*, vol. 54, pp. 23–32, 2015.

[16] C. K. Balcombe, J. T. Anderson, R. H. Fortney, J. S. Rentch, W. N. Grafton, and W. S. Kordek, "A comparison of plant communities in mitigation and reference wetlands in the mid-appalachians," *Wetlands*, vol. 25, no. 1, pp. 130–142, 2005.

[17] K. E. Francl, S. B. Castleberry, and W. M. Ford, "Small mammal communities of high elevation central Appalachian wetlands," *American Midland Naturalist*, vol. 151, no. 2, pp. 388–398, 2004.

[18] G. F. Strain, P. J. Turk, and J. T. Anderson, "Functional equivalency of created and natural wetlands: diet composition of red-spotted newts (Notophthalmus viridescens viridescens)," *Wetlands Ecology and Management*, vol. 22, no. 6, pp. 659–669, 2014.

[19] C. K. Balcombe, J. T. Anderson, R. H. Fortney, and W. S. Kordek, "Wildlife use of mitigation and reference wetlands in West Virginia," *Ecological Engineering*, vol. 25, no. 1, pp. 85–99, 2005.

[20] C. K. Balcombe, J. T. Anderson, R. H. Fortney, and W. S. Kordek, "Vegetation, invertebrate, and wildlife community rankings and habitat analysis of mitigation wetlands in West Virginia," *Wetlands Ecology and Management*, vol. 13, no. 5, pp. 517–530, 2005.

[21] C. K. Balcombe, J. T. Anderson, R. H. Fortney, and W. S. Kordek, "Aquatic macroinvertebrate assemblages in mitigated and natural wetlands," *Hydrobiologia*, vol. 541, no. 1, pp. 175–188, 2005.

[22] G. L. Bruland and C. J. Richardson, "Comparison of soil organic matter in created, restored and paired natural wetlands in North Carolina," *Wetlands Ecology and Management*, vol. 14, no. 3, pp. 245–251, 2006.

[23] C. A. Cole and R. P. Brooks, "A comparison of the hydrologic characteristics of natural and created mainstem floodplain wetlands in Pennsylvania," *Ecological Engineering*, vol. 14, no. 3, pp. 221–231, 2000.

[24] D. W. Desrochers, J. C. Keagy, and D. A. Cristol, "Created versus natural wetlands: avian communities in Virginia salt marshes," *Ecoscience*, vol. 15, no. 1, pp. 36–43, 2008.

[25] D. Hartzell, J. R. Bidwell, and C. A. Davis, "A comparison of natural and created depressional wetlands in Central Oklahoma using metrics from indices of biological integrity," *Wetlands*, vol. 27, no. 4, pp. 794–805, 2007.

[26] E. C. Snell-Rood and D. A. Cristol, "Avian communities of created and natural wetlands: bottomland forests in Virginia," *The Condor*, vol. 105, no. 2, pp. 303–315, 2003.

[27] West Virginia Division of Natural Resources, *2015 West Virginia State Wildlife Action Plan*, West Virginia Division of Natural Resources, South Charleston, WVa, USA, 2015.

[28] J. A. Kushlan, "Feeding ecology of wading birds," in *Wading Birds*, S. Winckler, Ed., National Audubon Society Research Report no. 7, pp. 249–297, National Audubon Society, New York, NY, USA, 1978.

[29] C. L. White and M. B. Main, "Waterbird use of created wetlands in golf-course landscapes," *Wildlife Society Bulletin*, vol. 33, no. 2, pp. 411–421, 2005.

[30] M. W. Weller, *Wetland Birds, Habitat Resources and Conservation*, Cambridge University Press, Cambridge, UK, 1999.

[31] W. R. Rice, "Analyzing tables of statistical tests," *Evolution*, vol. 43, no. 1, pp. 223–225, 1989.

[32] G. J. Robertson and F. Cooke, "Winter philopatry in migratory waterfowl," *The Auk*, vol. 116, no. 1, pp. 20–34, 1999.

[33] M. Paracuellos and J. L. Tellería, "Factors affecting the distribution of a waterbird community: the role of habitat configuration and bird abundance," *Waterbirds*, vol. 27, no. 4, pp. 446–453, 2004.

[34] J. A. Sánchez-Zapata, J. D. Anadón, M. Carrete et al., "Breeding waterbirds in relation to artificial pond attributes: implications for the design of irrigation facilities," *Biodiversity and Conservation*, vol. 14, no. 7, pp. 1627–1639, 2005.

[35] M. Paracuellos, "How can habitat selection affect the use of a wetland complex by waterbirds?" *Biodiversity and Conservation*, vol. 15, no. 14, pp. 4569–4582, 2006.

[36] A. Froneman, M. J. Mangnall, R. M. Little, and T. M. Crowe, "Waterbird assemblages and associated habitat characteristics of farm ponds in the Western Cape, South Africa," *Biodiversity and Conservation*, vol. 10, no. 2, pp. 251–270, 2001.

[37] N. Warnock, G. W. Page, T. D. Ruhlen, N. Nur, J. Y. Takekawa, and J. T. Hanson, "Management and conservation of San Francisco Bay salt ponds: effects of pond salinity, area, tide, and season on pacific flyway waterbirds," *Waterbirds*, vol. 25, no. 2, pp. 79–92, 2002.

[38] C. S. Elphick and L. W. Oring, "Winter management of Californian rice fields for waterbirds," *Journal of Applied Ecology*, vol. 35, no. 1, pp. 95–108, 1998.

[39] C. R. Isola, M. A. Colwell, O. W. Taft, and R. J. Safran, "Interspecific differences in habitat use of shorebirds and waterfowl foraging in managed wetlands of California's San Joaquin Valley," *Waterbirds*, vol. 23, no. 2, pp. 196–203, 2000.

[40] T. M. Darnell and E. H. Smith, "Avian use of natural and created salt marsh in Texas, USA," *Waterbirds*, vol. 27, no. 3, pp. 355–361, 2004.

[41] Z. Ma, Y. Cai, B. Li, and J. Chen, "Managing wetland habitats for waterbirds: an international perspective," *Wetlands*, vol. 30, no. 1, pp. 15–27, 2010.

[42] R. J. Craig and K. G. Beal, "The influence of habitat variables on marsh bird communities of the Connecticut River estuary," *The Wilson Bulletin*, vol. 104, no. 1, pp. 295–311, 1992.

[43] K. J. Havens, L. M. Varnell, and J. G. Bradshaw, "An assessment of ecological conditions in a constructed tidal marsh and two natural reference tidal marshes in coastal Virginia," *Ecological Engineering*, vol. 4, no. 2, pp. 117–141, 1995.

[44] S. C. Brown and C. R. Smith, "Breeding season bird use of recently restored versus natural wetlands in New York," *Journal of Wildlife Management*, vol. 62, no. 4, pp. 1480–1491, 1998.

[45] S. R. Confer and W. A. Niering, "Comparison of created and natural freshwater emergent wetlands in Connecticut (USA),"

Wetlands Ecology and Management, vol. 2, no. 3, pp. 143–156, 1992.

[46] J. A. Reinartz and E. L. Warne, "Development of vegetation in small created wetlands in southeastern Wisconsin," *Wetlands*, vol. 13, no. 3, pp. 153–164, 1993.

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Research Article

Improving Urban Water Environment in Eastern China by Blending Traditional with Modern Landscape Planning

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As a fundamental part of greenspace, urban water landscape contributes greatly to the ecological system and at the same time supplies a leisure area for residents. The paper did an analysis on the number of aquatic plant communities, the form of water spaces, and water quality condition by investigating 135 quadrats (90 at amphibious boundary and the land, 45 in the water) in 45 transects of 15 urban and suburban parks. We found that water spaces had monotonous forms with low biodiversity and poor water quality. In addition, urban water landscapes hardly provided ecological functions given excessive construction. Accordingly, a proposition to connect tradition with modernism in the improvement and innovation of urban water landscape planning was put forward, and further, the way to achieve it was explored. By taking Qinhu Wetland Park as a case, the principles and specific planning methods on macro- and microperspectives were discussed to guide the development of urban landscape in eastern China.

1. Background: The Condition of Urban Water Landscape in China

According to "2015 Report on the State of the Environment of China," the result of surface water quality monitoring in 967 sites distributed in 423 main rivers and 62 lakes and reservoirs showed that water quality in I~III category, IV~V category, and V category accounted for 64.5%, 26.7%, and 8.8% respectively. Since the 13th five year plan, Chinese government has been investing heavily on water environment treatment. For example, Wuxi city plans to put 6 billion RMB on more than 26 rivers including the ancient canal, Huancheng river, the grand canal, and Liangxi river to build urban water tourism landscape; Baotou city invested over 18 billion RMB in the improvement and utilization of urban water ecosystem; Shantou city invested around 10.8 million RMB to treat black and foul water. However, serious conditions of water environment still remain although great achievements of water environment treatment have been made compared with the past.

Located in the middle and lower reaches of the Yangtze, Nanjing is a city with rich water resources. Four water systems can be divided as the Yangtze, Chu River, Qinhuai River, and Shuiyang River, including 564 rivers in total among which 57 rivers cover more than 50 km^2. In the 1950s, the amount of lakes and ponds in urban area reached over 300. However, it decreased dramatically since then: Xuanwu lake, Yanque lake, and Mochou lake have degenerated into inland lakes after rounds of landfill; few water landscapes were left and lots of ponds and lakes became none-penetrated surface destructive to water systems as well as urban geological landscape at the end of 1990s. Nowadays, most water landscapes in Nanjing belong to closed or illiquid small systems that get easily polluted with low water environmental capacity and poor self-purification capability [1].

At present, two aspects are worth considering in water environment treatment. Firstly, the construction and reform of urban water system in close touch with people need to be taken as seriously as the treatment of large rivers and lakes. Secondly, how to construct water landscape in a scientific way should be considered given the shortage of relative researches and standards [2]. In this case, this paper investigated means to build attractive landscape with sustainability in urban areas based on the perspective of ecological restoration and aesthetic functions of landscape.

2. Conditions of Urban Water Landscape in Nanjing

2.1. Research Location. Located in the middle and lower reaches of the Yangtze, Nanjing is a land mainly with low mountains and gradual hills. Rich in the water resources (learned from "Nanjing water introduction"); it owns altogether 564 rivers with average volume of 1.8 billion m^3, among which surface water resources reach up to 22.1 billion m^3.

We collected data on 15 wetland parks in Nanjing (Figure 1, 8 parks in the urban area and 7 parks in the suburbs). Three urban wetland parks as Qinhuai River, Xuanwu lake, and Mochou lake belong to main river systems in Nanjing. 15 plots are as follows: Xuanwu lake (Xuanwu men), the Couple park, Xianlin lake, Yueya lake, Qinhuai lakeegret island park, Yangshan park, Nanjing seven-bridge urn ecological wetland park, Jiulong lake, Foshou lake, Mochou lake, Xiuqiu park, Dongshuiguan heritage (Qinhuai lake), and Qianhu.

2.2. Methods and Results. We used typical sample method to collect data. Three transects (1 meter wide and 4 meters long) perpendicular to the amphibious boundaries were arranged in each location. In each transect, three quadrats at 1 m × 1 m were set up at the water, amphibious boundary and the land, respectively (Figure 2). The interval distance between each quadrat varied from 0.5 to 1 m depending on the specific landform; for example, quadrats at the two ends of a clear amphibious boundary could connect. Quadrats in the water extended to the threshold covered by water vascular plants, while those in amphibious boundary and the land were arranged on moisture gradient until the appearance of massive typical xerophytia or road edge. Altogether 135 quadrats (90 at amphibious boundary and the land, 45 in the water) were arranged in 45 transects of 15 urban and suburban parks.

Survey in position was used to make records on the number of aquatic plant species, their frequency, and plant coverage (the proportion of area covered by crown geometry shade to that of the quadrats) sorted by importance calculations (relative plant coverage as the coverage of one species to the area covered by all species). After 15 days, data in 45 transects were collected. Table 1 showed the longitude and latitude of one transect as an example at each location.

The research showed that sixty species were usually seen in Nanjing wetland parks: 3 as submerged plants, 4 as floating-leaved plants, 12 as emerging plants, and 41 as hygrophytes. In the suburbs, we collected 23 aquatic species in total, among which 2 were submerged plants, 2 floating-leaved plants, 14 hygrophytes, and 5 emerging plants (Table 2). Deficiency in species accompanied by wide distribution of certain kinds was universal in suburban parks. *Persicaria lapathifolia* reached 81% in frequency and ranked the first in all species. In the species of hygrophytes, *Ranunculus sceleratus* and *Alternanthera philoxeroides* also showed high frequency at 57%. The frequency of *Potamogeton crispus* was up to 43% in submerged plants. *Iris tectorum* and *Arundinella anomala* showed the highest frequency in emerging plants at 38% (the same with Spirogyra in floating-leaved plants). *Potamogeton crispus* ranked the highest in relative coverage at 0.95 in all species, followed by *Spirogyra* at 0.68, *Iris tectorum* at 0.41, *Arundinella anomala* at 0.32, *Lemna minor* at 0.32, *Persicaria lapathifolia* at 0.2, and *Phragmites australis* at 0.19.

Table 3 showed that 26 species could be found in central city parks. *Persicaria lapathifolia* also ranked the highest in frequency at 79.2% in all species, followed by *Alternanthera philoxeroides* at 66.7%, *reeds* at 63.4%, *Potamogeton crispus* at 58.3%, *Spirogyra* at 58.3%, and *Iris tectorum* at 50%. *Potamogeton crispus* ranked the highest in relative coverage at 0.93 in all species, followed by *Iris tectorum* at 0.48, *Spirogyra* at 0.43, *Lemna minor* at 0.38, and *Phragmites australis* at 0.34; *Persicaria lapathifolia* at 0.22, *Nymphaea tetragona*,and *Alternanthera philoxeroides* at 0.19.

As it is shown in Figure 3, both urban and suburban parks had a tendency that hygrophytes and emerging plants were widely distributed, but much less submerged and floating-leaved plants appeared. In addition, aquatic plants were concentrated on certain types as *Persicaria lapathifolia*, *Alternanthera philoxeroides*, *Spirogyra*, *Potamogeton crispus*, and *Iris tectorum*.

Comparatively speaking, plants showed higher Simpson diversity in urban parks than those in the suburbs (Figure 4). Comparing with urban parks, suburban parks show a decrease in the proportion of hygrophytes with growing number of submerged and floating-leaved plants (Figure 3). Specifically, Simpson diversity was the lowest in Huashen lake and Foshou lake parks in the suburbs, while it reached the highest in Dongshuiguan heritage at 0.65 in central city.

2.3. Analysis and Suggestions. In this paper, we made a survey on the plants of urban and suburban wetland parks in Nanjing. Main findings are as follows.

Firstly, diversity of aquatic plants needs to be enhanced through human intervention. Generally speaking, the results showed that Simpson diversity in urban parks was higher than those in the suburbs. It may greatly be attributed to the government policies advocating plant configuration and maintenance in urban parks. However, limited choices of aquatic plants in landscape building are a problem. High frequency of certain species brings negative effects to ecological functions. In addition, the weak vertical stratification resulting from unbalanced proportion of hygrophytes, emerging plants, submerged plants, and floating-leaved plants can hardly bring aesthetic experiences to the audience.

Second, considering the ecological and aesthetic functions of wetlands, frequency of plant species needs to be taken into account with plant coverage in wetland management. The research showed that although *Potamogeton crispus* did not show most frequently, it ranked the first in relative coverage both in urban and in suburban parks. On the contrary, the most frequently shown plant, *Persicaria lapathifolia* lagged far behind in relative coverage (0.22 in urban parks and 0.2 in suburban parks). On the aspect of ecological functions, high relative coverage of some species may violate the habitats of other plants. As for wetland landscape planning, large scale

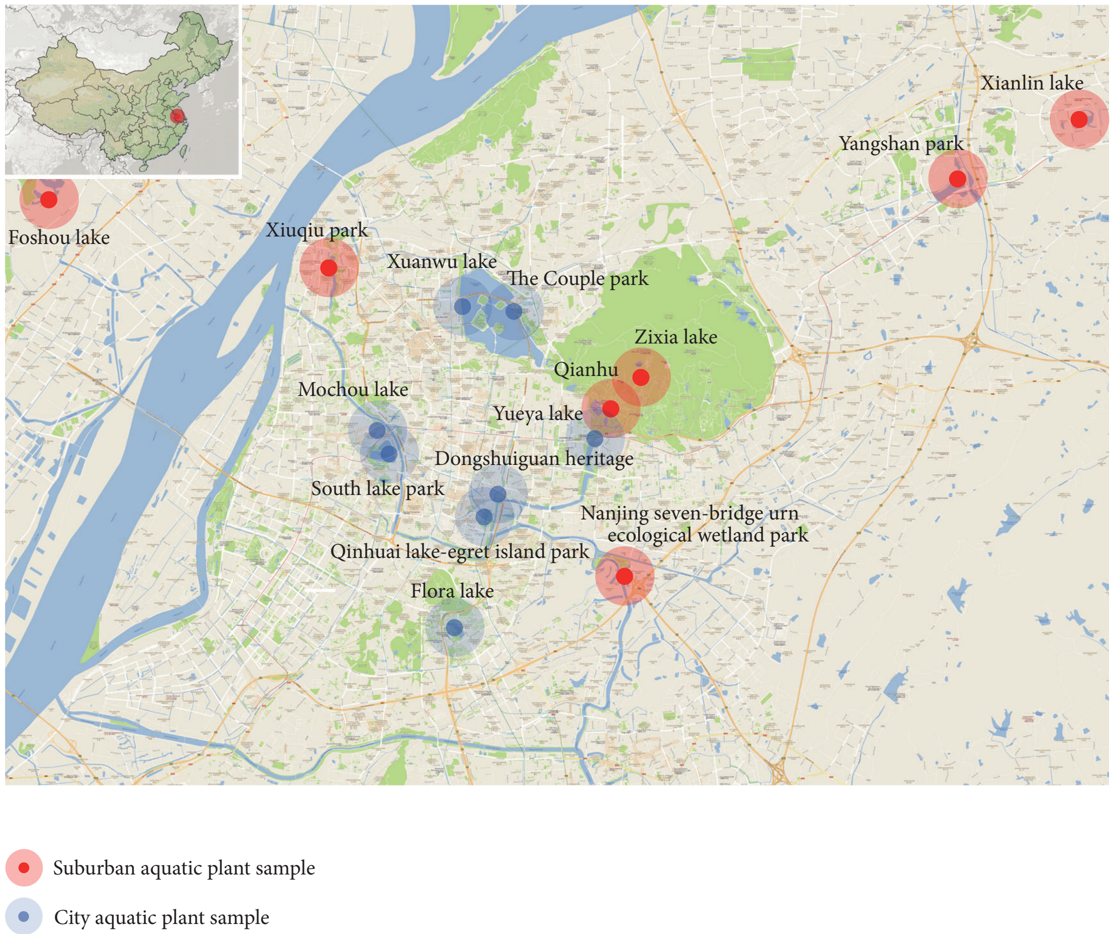

FIGURE 1: Fifteen research sites in Nanjing.

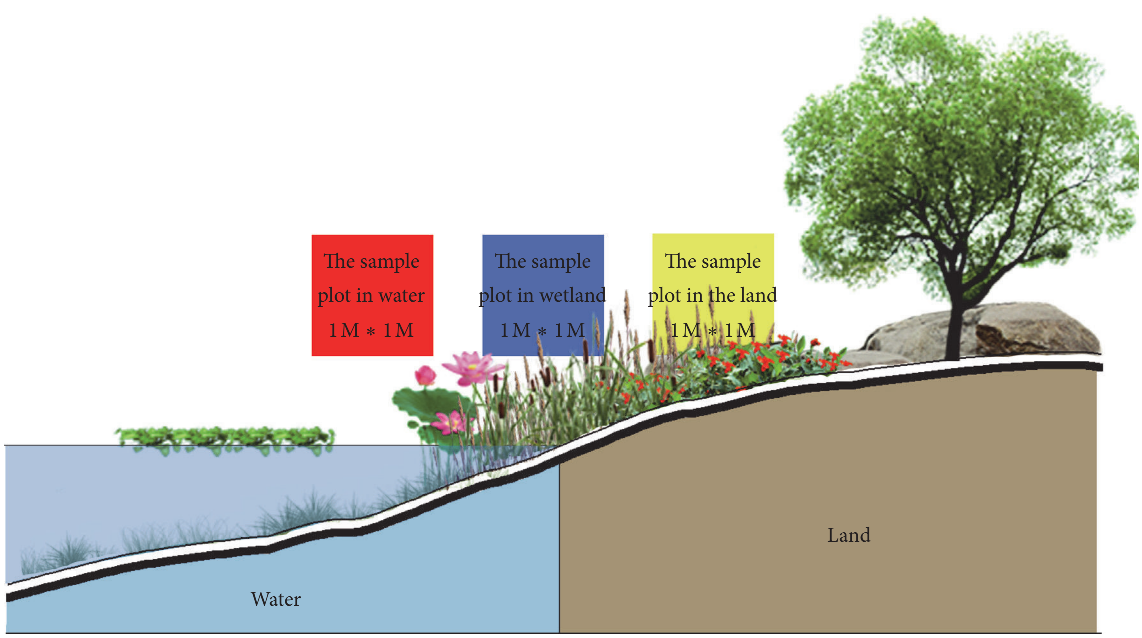

FIGURE 2: Nanjing seven-bridge urn ecological wetland park aquatic plant status (local).

Table 1: Sampling site conditions.

Locale	Longitude	Dimensionality	Area/ha	Dept/m	Lake and revetment form	Pollution level
Flora lake	31°59′23.84N	118°46′48.85E	5.6	20	The lake is rectangular in shape, with a narrow distance in E-W direction. On average, its width is 200 m. Most of the revetments are in natural way.	IV
The Couple park	32°04′16.98N	118°48′22.69E	10	1	The revetments are in natural way. The way the grassy slope extends into the water to make the revetments is used.	III
Xuanwu lake (Xuanwu men)	32°04′18.38N	118°47′25.84E	378	1.4	The lake is rhombus in shape, with 15 km in circumference, 24 km in S-W direction and 2.0 km in E-W direction. The revetments are hard on the periphery and natural in the central island.	IV
Mochou lake	32°02′05.91N	118°45′19.36E	0.37	1	The lake is triangle in shape, with 6 km in circumference. The revetments are mostly made of hard materials, such as rocks.	III
South lake park	32°01′51.08N	118°45′22.44E	10	1.5	The lake is ellipse in shape. Its revetments are mostly made of soil or hard materials and natural revetments are very few.	IV
Yueya lake	32°01′57.66N	118°49′29.71E	17.2	2	The lake is of North-South trend, with the water from Waiqinhuai river. The revetments, which are single in form, are made of hard materials.	III
Qinhuai lake-egret island park	32°01′16.09N	118°47′06.95E	3.8	1.5	The lake is scattered, and the size of the lake is quite significant. The revetments consist of hard form mainly, with some natural forms such as the grass and rock.	IV
Dongshuiguan heritage (Qinhuai lake)	32°01′28.68N	118°47′37.57E	1.55	1.1	The lake attaches to the Qinhuai river and its waterfront is hard revetment.	IV
Xianlin Lake	32°07′44.82N	118°58′41.28E	18	2	The lake is just like a goose web, and the waterfront is in natural form which extends the grassland into water.	II
Yangshan park	32°06′24.76N	118°56′05.23E	31	1.5	The lake makes a water circle by linking with Xihu park and Jiuxianghe wetland reserve. The revetment consists of hard form mostly.	III
Nanjing seven-bridge urn ecological wetland park	32°00′27.97N	118°50′08.58E	1.2	1.7	The lake is rectangle, and the revetment is hardening by gravel and cement.	IV
Qianhu	32°03′00.63N	118°49′39.75E	5.33	2	The size of the lake is just like a tadpole; the revetment was done in a natural way. Some of it is grassland-into-water, and some of it is in rock form.	IV
Zixia lake	32°03′41.50N	118°50′23.10E	4.7	1.2	The surface is just like a block. Most of revetment is in artificial hard form with a little natural form.	III

Table 1: Continued.

Locale	Longitude	Dimensionality	Area/ha	Dept/m	Lake and revetment form	Pollution level
Xiuqiu park	32°05′32.24N	118°44′20.71E	3.6	2	The lake is trapezoid with plenty of hardening revetment and little rock form.	III
Foshou lake	32°06′18.00N	118°38′35.97E	35	1.5	The lake is just like a hand, with hard and natural revetment combination.	II

Table 2: Suburban aquatic plants quantitative data.

Life style	Plant name	Frequency/%	Coverage/%	Important value	Serial number
Submerged plants	*Potamogeton crispus*	4	4.45	0.95	1
	Ceratophyllum demersum	5	0.21	0.05	2
Floating-leaved plants	*Spirogyra*	38	1.18	0.68	1
	Lemna minor	19	0.56	0.32	2
Hygrophytes	*Polygonum lapathifolium*	81	3.62	0.20	1
	Ceratium glomeratum Thuill	29	2.71	0.15	2
	Ranunculus sceleratus	57	2.36	0.13	3
	Alternanthera Philoxeroides (Mart.) Griseb	57	1.95	0.11	4
	Veronica persica	33	1.57	0.09	5
	Lamium amplexicaule L.	33	1.56	0.09	6
	Ranunculus cuneifolius Maxim.	43	0.93	0.05	7
	Herba Ranunculi Japonici	24	0.89	0.05	8
	Cymbidium nanulum	19	0.79	0.04	9
	Be-gonia pedatifida Levl.	14	0.52	0.03	10
	Rorippa indica	14	0.44	0.03	11
	Mazus japonicus	5	0.25	0.01	12
	Poaannua	5	0.13	0.01	13
	Capsella bursa-pastoris (Linn.) Medic.	10	0.07	0.01	14
Emerging plants	*Iris tectorum Maxim.*	38	3.37	0.41	1
	Arundinella anomala Steud.	38	2.69	0.32	2
	Phragmites australis	19	1.55	0.19	3
	Typha orientalis	5	0.43	0.05	4
	Canna indica	11	0.31	0.04	5

coverage of certain plants would definitely cause fatigue for the audience by their excessive appearances.

2.4. Urban Water Landscape Planning: Tradition and Developments. Researches have been supporting the great influences of greenspace on ecological performance [3], among which water landscape works as a fundamental part. However, overcommercialization lack of scientific planning and maintenance is destroying the quality of urban water landscape and further weakening the ecological and economic benefits brought by water systems. Traditional landscape planning deals with such problems mainly in the following parts.

Landscape Layouts. Scholars proposed topography and geomorphy plans adapted to natural forms and spatial patterns in a specific ecosystem. For example, a streamline shape is promoted for curved waterline contributes to protect ecological diversity by providing habitats with rich resources [4].

Landscape Architecture. Architectures in a landscape are necessary to meet people's needs of sightseeing and recreation. Generally speaking, large scale buildings should be placed away from the location of the best view, while those in small scale can be built in the scenic beauty spots in a style coordinated with the whole water landscape. Architectures submerged in the water can cover a small space in "bottom elevated" shape so as to reduce the disturbing effect to ecosystems. It is preferable to use natural materials or concrete construction of trunks and stones in landscape buildings [5–7].

Revetments. Ecological revetments, especially those varied with shore side situations, are preferred, such as ecological friendly planted revetments or lifted natural matrix soil instead of hard bulkhead to fulfil transition from land to lakes. Landlake ecozone should be in a stretched natural curve so as to embrace various habitats for animals [8, 9].

Plants. Generally speaking, submerged plants, floating and floating-leaved plants, emerging plants, and hygrophyte (preferably local species) are used in water landscape. Choices should be adjusted to meet plants' living habits in a consideration of biodiversity [10–12].

However, scholars have proposed critics towards these commonly used methods for their unsatisfactory management of plant diversity and the richness of layouts. For example, Freedman et al. (2006) pointed out that limited kinds of aquatic plants were applied in a monotonous layouts in urban wetland landscape. As it was shown in the findings of Nanjing urban wetland research, diversity both in species and in the spatial form of wetlands was in urgent need to be enhanced. Considering this, we introduce a successful case of urban water landscape planning in Eastern China to explore possible ways in wetland management innovation.

Table 3: Urban aquatic plants quantitative data.

Life style	Plant name	Frequency/%	Coverage/%	Important value	Serial number
Submerged plants	*Potamogeton crispus*	58.3	5.22	0.93	1
	Ceratophyllum demersum	12.5	0.40	0.07	2
Floating-leaved plants	*Spirogyra*	58.3	4.23	0.43	1
	Lemna minor	41.7	3.79	0.38	2
	Nymphaea tetragona	12.5	1.85	0.19	3
Hygrophytes	*Polygonum lapathifolium*	79.2	4.97	0.22	1
	Alternanthera Philoxeroides (Mart.) Griseb	66.7	4.57	0.19	2
	Herba Ranunculi Japonici	37.5	2.58	0.09	3
	Ranunculus cuneifolius Maxim.	29.2	2.25	0.07	4
	Mimulus bodinieri Vant.	25.0	1.46	0.04	5
	Arundinella anomala Steud.	12.5	1.23	0.04	6
	Ranunculus natans	20.8	1.08	0.05	7
	Ranunculus sceleratus	12.5	0.86	0.04	8
	Marsilea L.	4.2	0.83	0.04	9
	Mazus japonicus	12.5	0.79	0.02	10
	Poaannua	4.2	0.75	0.02	11
	Capsella bursa-pastoris (Linn.) Medic.	8.3	0.49	0.02	12
	Veronica persica	8.3	0.38	0.01	13
	Cymbidium nanulum	4.2	0.23	0.01	14
	Trifolium repens	4.2	0.20	0.01	15
	Rorippa indica	4.2	0.04	0.00	16
Emerging plants	*Iris tectorum Maxim.*	50.0	4.90	0.48	1
	Phragmites australis	63.4	4.79	0.34	2
	Canna indica	29.2	3.18	0.13	3
	Typha orientalis	12.5	0.99	0.06	4
	Scirpus validus Vahl	8.3	0.37	0.03	5

3. Case of Qinhu National Wetland Park: A Connection of Tradition and Modernism

In a northern subtropical monsoon climate, Qianhu National Wetland Park is located in the south of Lixia river basin, one of the most three famous basins in China. As a national 5A level urban wetland park, it owns significant wetland resources in the middle and lower reaches of the Yangtze and in central Jiangsu province (Figure 5). The fresh water wetland of Qinhu is rarely seen in China, among which natural lake wetlands account for 10.68 sq.km and man-made wetlands occupy 32.61 sq.km with various types as moor, lakes, rivers, and man-made wetlands [13].

Since the mid-20th century, due to land scarcity brought by population growth, excessive utilization on Qinhu wetland to reclaim land from lakes has greatly destroyed wetland ecology by turning large scale of wetlands to farms. With the awareness of wetland protection these days, in 2007, Qinhu National Wetland Park was listed in the protection network of the middle and lower reaches of the Yangtze with a conservation area of 26 sq.km (core area as 6 sq.km) and has been under continuous construction since then [14]. It provided the following enlightenments (Figure 6).

(1) Enriching Spatial Forms in Water Landscape. A multiscale spatial form was created to bring diverse wetland types: various landscape forms can be seen in Qinhu wetland, such as lakes, rivers, moors, mudflats, ponds, and pools, by making full use of existing situation.

In the lake wetlands occupying a large area, a flat terrestrial-aquatic transverse zone was designed to enlarge the contact area of land and water so that plants and animals would enjoy more living spaces. In the river wetlands, a terrestrial-aquatic transverse zone with varied illumination intensity could be seen to satisfy the demands of plant growth in a consideration of changing coastlines. In the moor and mudflat wetlands, plants instead of artificial factors were used to enrich wetland functions. In the pond and pool wetlands, a unique regional environment was created by embracing local plants, culture, and heritage into the planning.

(2) Enriching the Cross Section Structure of Terrestrial-Aquatic Transverse Zones. Transiting from water to land ecosystem, the terrestrial-aquatic transverse zone is significant in wetland landscape. Different from traditional ecological revetments design, diversified structures of cross section were created in Qinhu Wetland Park by constructing microtopography in the transverse zone. Meanwhile, varied kinds of plants were adopted to enrich spatial forms so as to build manifold habitats as well as coastline landscapes (Figure 7).

As a result, the ecology of Qinhu wetland has been greatly improved. With growing self-adjustment capacity, the

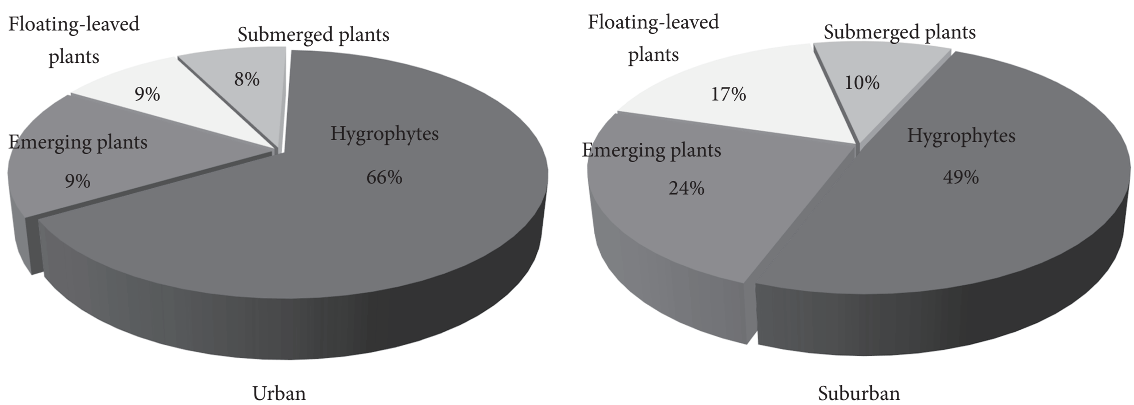

Figure 3: Percentages of aquatic plant life form urban and suburban parks.

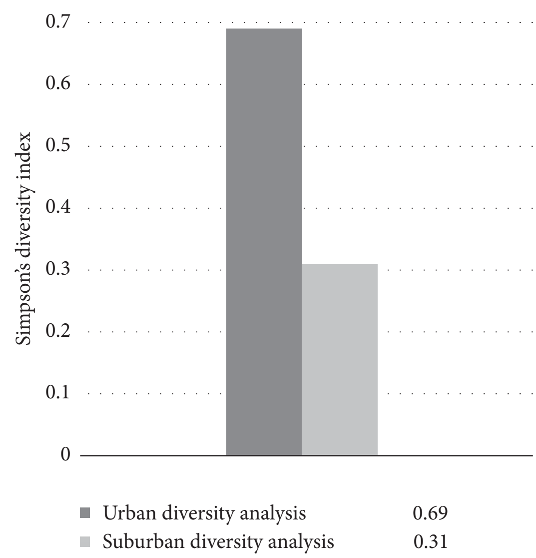

Figure 4: Urban and suburban diversity analysis.

biocenosis remained stable where biodiversity increased the preliminary statistics showing that there were 153 kinds of plants, 97 kinds of birds, 21 kinds of beasts, 38 kinds of fishes, and 21 kinds of floating animals. The main plants were as follows: *fluitantes as diatom*, *Cyanophyta*, and *chlorella*, and so forth; floating plants as *Eichhornia crassipes*, *Nymphaeales*, *Floral Aquatic*, *Salvinia natans*, and so forth; submerged plants as *Ceratophyllum demersum*, *Potamogeton distinctus*; water woody plants as *Taxodium ascendens*, *Metasequoia glyptostroboides*, *dryland willow*, *Pterocarya stenoptera*, *italian popular*, and so forth; water herbage as *Iris wilsonii*, *Typha orientalis*, *reed*, *Erigeron annuus*, *Indian Kalimeris Herb*, *Common hogfenneI root*, and so forth.

4. Rethinking about the Case of Qinhu National Wetland Park

Traditional principals of urban water landscape planning do have limits nowadays considering the idea of orderly revetments, monotonous landscape forms, and plant arrangements. Only through breaking these limitations, can urban water landscape be innovated in terms of construction and transformation. It can be achieved through the following two aspects.

4.1. Macroaspects

(1) Promotion and Education about the Value of Urban Water System. Ignorance of urban water system for a long time has led to dramatic decrease in the number of water resources and poor water quality. Superficial understanding still exists in recent years, although growing consciousness has been seen on the protection of urban water system. Constructions with damage to water systems usually happen, especially in the process of building urban water landscape. Present community structure is the result of natural events and human intervention years by years [15]. Accordingly, it is necessary to inform and educate the public about their value and how to utilize them in a sustainable way. "Mass awareness

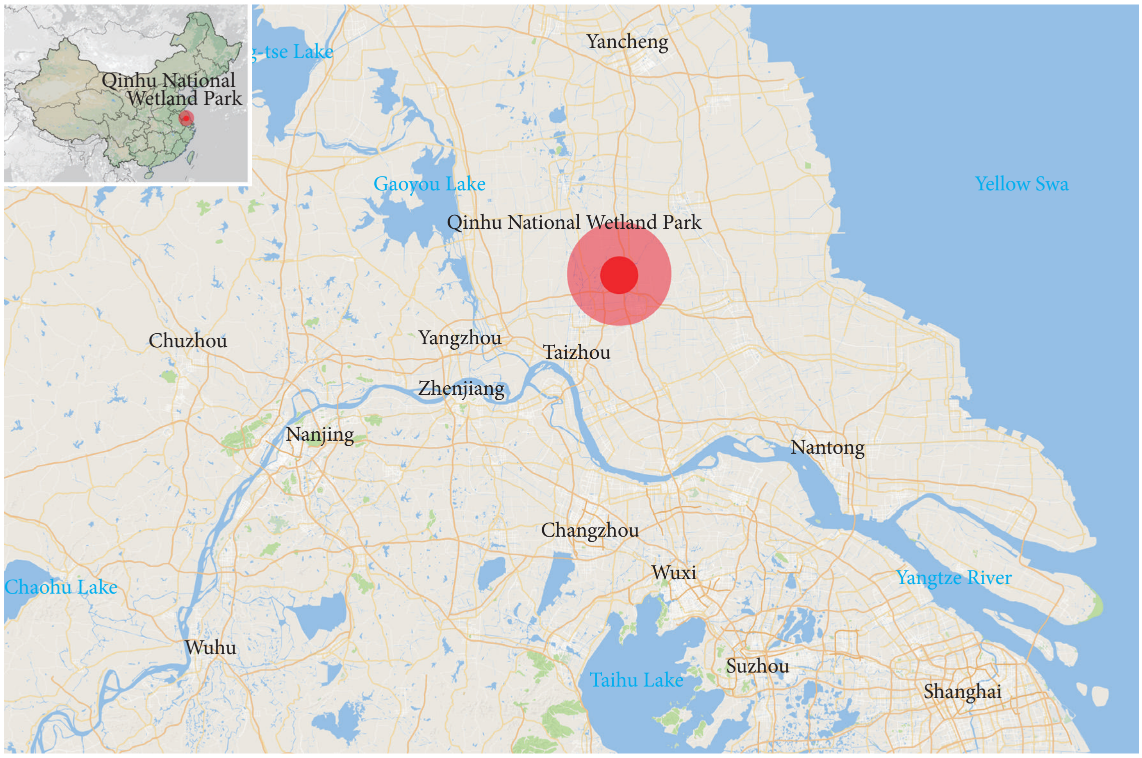

Figure 5: The location map of Qinhu National Wetland Park.

programme" carried out by the government and NGOs may work on this aspect [16].

(2) Coordinating the Utilization Plan of Urban Water System. As an ecosystem, urban water system should be utilized in an integrated way in which different departments cooperate closely. It is an initial step to clarify how to use varied types of urban water resources, especially those located on the edge of plan areas so that they can be listed in green belt systems. People need to define the functions, attributes, and protecting scope of urban water and further compile protection and utilization plans in the guidance of city greenbelt plans.

(3) Pollution Control to Protect Urban Water Ecology. The sustainable utilization of urban water greatly relies on the improvements of water ecology. Therefore, it is a precondition to reduce and eradicate diversified sources of pollution which may bring destructive disaster for water ecology. At the same time, sewage closure is needed to avoid secondary contamination by separating rains and sewers.

(4) Gradually Enhancing the Coverage and Quality of Urban Water. If it is possible, artificial water landscapes can be built in cities accompanied with the reconstruction of water ecosystem. Through ecological techniques and engineering, degenerated or disappearing water system can be restored and reconstructed to provide the ecological functions of wetlands, as well as recreation areas for residents.

4.2. Microaspects. To some extent, the case of Qinhu wetland planning can be used for reference in urban water management in eastern China. According to the planning goals based on value and significance of city water resources, researches and production, environmental protection, leisure, and recreation should be combined so as to enrich users' experiences in water landscape and meanwhile enhance education effects. This evolutionary system needs to be in accordance with city development, the construction of ecological environment, and residents' demands in a sustainable principal. Urban water landscape planning should stress the following two parts.

(1) Spatial Forms of Water Landscape. Landscape ecology is focused on the relationship of spatial forms and ecological processes [17]. Researchers have been advocating the natural side of landscape, that is, to integrate "the spatially heterogeneous area" with its ecological functions [18]; for example, land-use activity is found to influence aquatic diversity [19], Ward et al. [18] proposed that patch size and shape contributed greatly on biodiversity, the research of Robinson et al. (2002) showed that the level of biodiversity decreased together with less ecotones, and Camacho-Valdez et al. (2013) and Chaikumbung et al. (2016) suggested that wetland characteristics, distribution, and contexts all had effect on their values. Generally speaking, structural complexity plays a positive role in ecosystem engineering [20]. A nonequilibrium state is common in ecosystems

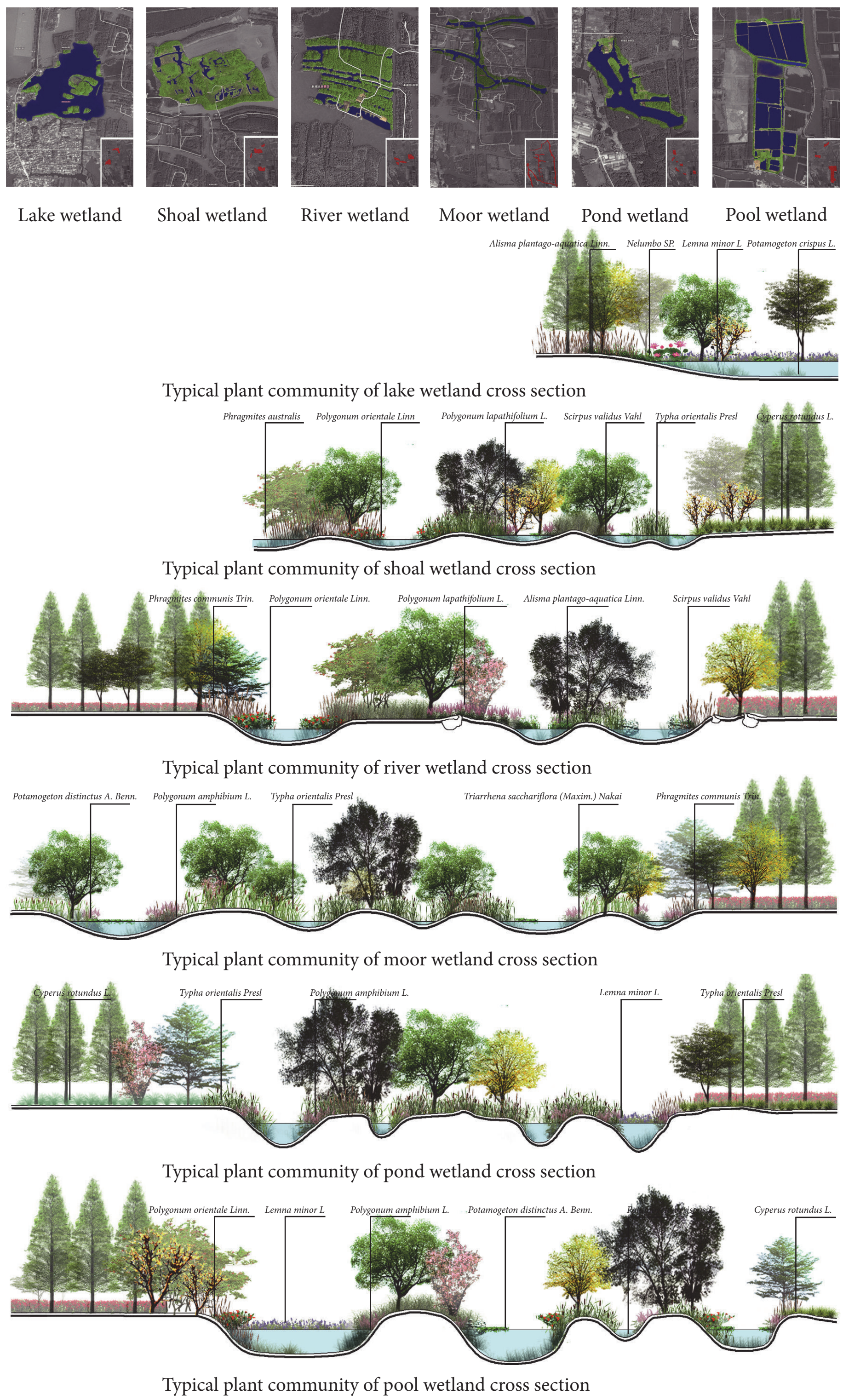

FIGURE 6: Plant community of wetland cross section.

for their heterogeneity in time and space. Different water spatial forms can provide a plant friendly environment by increasing heterogeneity. It helps build a steady biome in the harmonious ecosystem. For example, natural biofilm can be made from large amounts of gravels in shoals and streams to purify suspended matters, nitrogen, phosphorus and heavy metals, and so forth. Plant roots, as well as the complex structure of soil, roots, and microorganism in the water, help degrade organic pollutants. Drops and waterfalls increase dissolved oxygen through the contact area of water and the

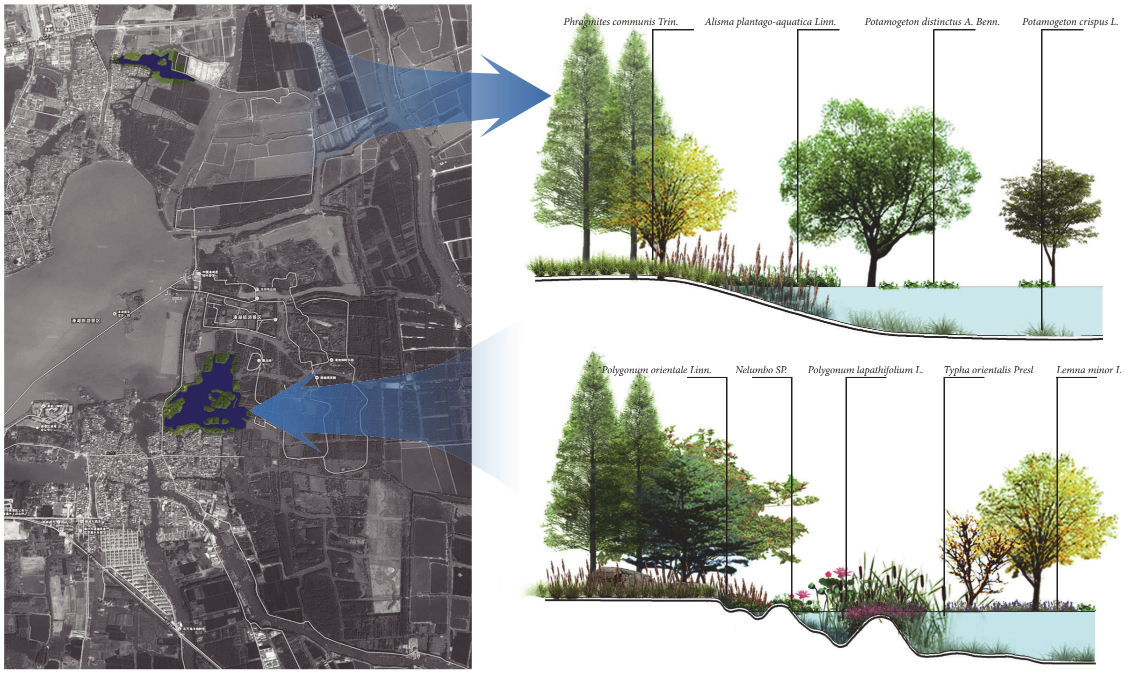

Figure 7: The structural comparison of revetments cross section.

air. However, urban water systems lack structural complexity and heterogeneity for “a smooth, engineered, almost vertical shoreline and bare substratum” [21]. In this case, diversified spatial forms of terrains and water should be fully utilized in water landscape planning.

(2) Ecological Revetments. Revetments play an important role in the material exchange between water and land. Therefore, water, land, and revetments should be taken as a whole to facilitate their interaction in revetment planning. Ecological planting is a primary part in revetments: waterfront plants and aquatic plants are connected to provide a sustainable natural environment for creatures through flat coastlines, natural transition from land to water, and coastline microtopography. Meanwhile, it leads the audience to change attention with the shape of coastlines. Varying sceneries with changing viewpoints can be achieved due to zigzagging natural banks. The functions of permeation and purification are given full play in ecological revetments in which the wetland increases its role of self-adjustment to bring a virtuous circle for the whole system, and at the same time, a vibrant landscape is enjoyed by the audience.

To sum up, in the improvement and innovation of urban water environment in eastern China, ecological landscape planning is needed to blend tradition with modernism. We should continually bring the latest scientific findings to existing methods to advance the development of water landscape planning in eastern China.

Competing Interests

The authors declare that they have no competing interests.

Acknowledgments

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References

[1] Y. Q. Guo, “Pollution control and remediation technology for urban landscape water,” *Environmental Science and Technology*, vol. 28, supplement 1, pp. 148–150, 2005.

[2] L. Ding, Y. L. Shen, and Y. Huang, “The present situations of technology and development of restoration for park water,” *Journal of Suzhou University of Science and Technology (Engineering and Technology)*, vol. 18, no. 2, pp. 48–52, 2005.

[3] V. Whitford, A. R. Ennos, and J. F. Handley, “‘City form and natural process’—indicators for the ecological performance of urban areas and their application to Merseyside, UK,” *Landscape and Urban Planning*, vol. 57, no. 2, pp. 91–103, 2001.

[4] B. Hong, S. Liu, and S. H. Li, “Ecological landscape planning and design of an urban landscape fringe area: a case study of Yang’an District of Jiande City,” *Procedia Engineering*, vol. 21, pp. 414–420, 2011.

[5] Y. Xia, “Design on urban landscape facilities based on mass behavior,” *Advances in Social Science Education and Humanities Research*, vol. 37, pp. 457–460, 2016.

[6] J. D. Zhou, "Research on the ecological planning and design theory framework of City Wetland Park," *Journal of Anhui Agricultural Sciences*, vol. 35, no. 36, pp. 11818–11821, 2007.

[7] K. J. Yu, D. H. Li, and L. M. Chao, "Ten landscape strategies to build urban ecological infrastructure," *Planners*, vol. 17, no. 6, pp. 9–13, 2001.

[8] Y. Tian, Z. Q. Guo, W. Zhong, Y. C. Qiao, and J. X. Qin, "A design of ecological restoration and eco-revetment construction for the riparian zone of Xianghe Segment of China's Grand Canal," *International Journal of Sustainable Development and World Ecology*, vol. 23, no. 4, pp. 333–342, 2016.

[9] J. Zhao, X. Liu, R. Dong, and G. Shao, "Landsenses ecology and ecological planning toward sustainable development," *International Journal of Sustainable Development and World Ecology*, vol. 23, no. 4, pp. 293–297, 2016.

[10] C.-B. Hsu, H.-L. Hsieh, L. Yang et al., "Biodiversity of constructed wetlands for wastewater treatment," *Ecological Engineering*, vol. 37, no. 10, pp. 1533–1545, 2011.

[11] B. Ma and P. Z. Li, "Primary theory and discussion on the ecological design of urban wetland scenery," *Journal of Shanxi Agricultural Science*, vol. 36, no. 2, pp. 84–86, 2008.

[12] Y. Z. Zou, "Ecological design method on urban wetland landscape," *Heilongjiang Science and Technology of Water Conservancy*, vol. 40, no. 10, pp. 96–100, 2012.

[13] Y. P. Zhou, "Evaluation of wetland ecosystem service value in Qinhu Lake," *Journal of Meteorology and Environment*, vol. 26, no. 2, pp. 16–20, 2010.

[14] X. Zhang, Q. F. You, F. Ling, and D. Z. Chen, "Present status and restoration measure of Qin Lake wetland," *Journal of Jiangsu Forestry Science & Technology*, vol. 39, no. 6, pp. 22–25, 2012.

[15] J. Adis and W. J. Junk, "Terrestrial invertebrates inhabiting lowland river floodplains of Central Amazonia and Central Europe: a review," *Freshwater Biology*, vol. 47, no. 4, pp. 711–731, 2002.

[16] T. V. Ramachandra, R. Rajinikanth, and V. G. Ranjini, "Economic valuation of wetlands," *Journal of Environmental Biology*, vol. 26, no. 2, pp. 439–447, 2005.

[17] J. A. Wiens, "Riverine landscapes: taking landscape ecology into the water," *Freshwater Biology*, vol. 47, no. 4, pp. 501–515, 2002.

[18] J. V. Ward, F. Malard, and K. Tockner, "Landscape ecology: a framework for integrating pattern and process in river corridors," *Landscape Ecology*, vol. 17, pp. 35–45, 2002.

[19] J. S. Harding, E. F. Benfield, P. V. Bolstad, G. S. Helfman, and E. B. D. Jones III, "Stream biodiversity: the ghost of land use past," *Proceedings of the National Academy of Sciences of the United States of America*, vol. 95, no. 25, pp. 14843–14847, 1998.

[20] C. T. Robinson, K. Tockner, and J. V. Ward, "The fauna of dynamic riverine landscapes," *Freshwater Biology*, vol. 47, no. 4, pp. 661–677, 2002.

[21] N. J. Waltham, A. Reichelt-Brushett, D. McCann, and B. D. Eyre, "Water and sediment quality, nutrient biochemistry and pollution loads in an urban freshwater lake: balancing human and ecological services," *Environmental Sciences: Processes and Impacts*, vol. 16, no. 12, pp. 2804–2813, 2014.

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Research Article

The Effect of Artificial Mowing on the Competition of *Phragmites australis* and *Spartina alterniflora* in the Yangtze Estuary

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Spartina alterniflora Loisel. is one of the most invasive species in the world. However, little is known about the role of artificial mowing in its invasiveness and competiveness. In this work, we studied the effect of mowing on its interspecific interactions with native species *Phragmites australis* (Cav.) Trin ex Steud of the Yangtze Estuary, China. We calculated their relative neighbor effect (RNE) index, effect of relative crowding (D_r) index, and interaction strength (I) index. The results showed that the RNE of *Phragmites australis* and *Spartina alterniflora* was 0.354 and 0.619, respectively, and they have competitive interactions. The mowing treatments can significantly influence the RNE of *Phragmites australis* and *Spartina alterniflora* on each other. Concretely, the RNE of *Spartina alterniflora* in the removal treatments was significantly higher than the value in the controls. But the RNE of *Phragmites australis* in the removal treatments was significantly lower than the value in the controls. Meanwhile, D_r of the two species on the targets was higher in the removal treatments than that in the controls, and the opposite was for I. We concluded that artificial mowing could promote the invasion of *Spartina alterniflora* by increasing its competitive performance compared with native species.

1. Introduction

The problem of invasive species and their control has become one of the most pressing applied issues in ecology today [1]. More and more attention was paid to invasive species and the research on the interactions between invasive and native species was thought to be important for biological control [2]. Plant-plant interactions are key processes that strongly influence the composition and structure of plant communities, and both biotic and abiotic factors can influence the outcome of interspecific interactions [3]. Artificial mowing can influence the interactions between invasive species and native species through increasing the availability of both light and nutrients and further to affect species richness, composition, and the competition of species [4, 5].

Salt marsh is susceptible to biological invasion and the invasion has dramatic consequences that include local extinctions of native species, genetic modifications, species displacements, and habitat degradation [6]. *Spartina alterniflora* Loisel. (smooth cordgrass) was widely recognized as one of the most aggressive invaders of estuaries and salt marshes around the world [7]. A large number of studies have been carried out on its growth and reproduction and control managements [8, 9]. The ecological measures of controlling *Spartina alterniflora* expansion include chemical herbicide, physical measures, and biological control [10].

The well-known invasive *Spartina alterniflora* was intentionally introduced to China in 2001 for the purpose of land reclamation and the prevention of soil erosion [11, 12]. Its introduction has caused severe ecological consequences to the native ecosystems including conversion of mudflats to *Spartina* meadows, decrease in abundance of native species, considerable loss of shorebirds' foraging habitats, and degradation of native ecosystems. Dongtan wetland of Chongming, East China, which is a typical tidal marsh with an environmental gradient, has also introduced the species in the 1970s and 1980s [13]. The introduction and spread of the alien species resulted in a decrease in the bird biodiversity and abundance of some native species such as *Scirpus triqueter* [11, 14, 15]. *Phragmites australis* (Cav.) Trin. ex Steud (common reed), a native species in the Dongtan wetland, coexisted with invasive *Spartina alterniflora*, and they have very similar competitive abilities [16]. Therefore, the biological control of *Spartina alterniflora* by *Phragmites australis* has been a focus of increasing management concern of the government and the ecological scholars.

Some studies have compared the competitive abilities of *Phragmites australis* and *Spartina alterniflora* [17, 18]. And we have known that the invasion of *Spartina alterniflora* and its competition with native species is dramatically influenced by abiotic factors such as salt and water [19]. It seems that no consistent conclusion on their interactions of the two species has been reached. Most of them indicated that the relative competitive ability of *Spartina alterniflora* was significantly greater than that of *Phragmites australis*, but some studies indicated that their interactions may be competitive or facilitative owing to different study conditions [15, 20–22]. We thought the plant interactions of the two species in our research site were competitive according to our early works [16]. However, the role of artificial mowing in their interspecific interactions has not yet been clear.

Herbivores and artificial mowing are very common in the Dongtan wetland of Chongming, East China, and these activities influenced the growth and breed of the species in the area. Mowing could respond in a density-dependent fashion and make plants less able to compete with either conspecifics or surrounding vegetation [5]. Therefore, artificial mowing may influence the control effects of *Spartina alterniflora* by *Phragmites australis*. In this study, we conducted a mowing experiment to study the effect of artificial mowing on the interspecific interactions of *Phragmites australis* and *Spartina alterniflora*.

We evaluated the effects of artificial mowing on the interspecific interactions of *Phragmites australis* and *Spartina alterniflora* by comparing their competitive performance and competitive intensities in different artificial mowing intensity. To evaluate competitive intensities, we chose three measuring indices: relative neighbor effect (RNE), relative crowding (D_r), and interaction strength (I). We attempted to address the following questions: what is the interspecific interaction of *Phragmites australis* and *Spartina alterniflora* in the research site? How can artificial mowing influence their interactions?

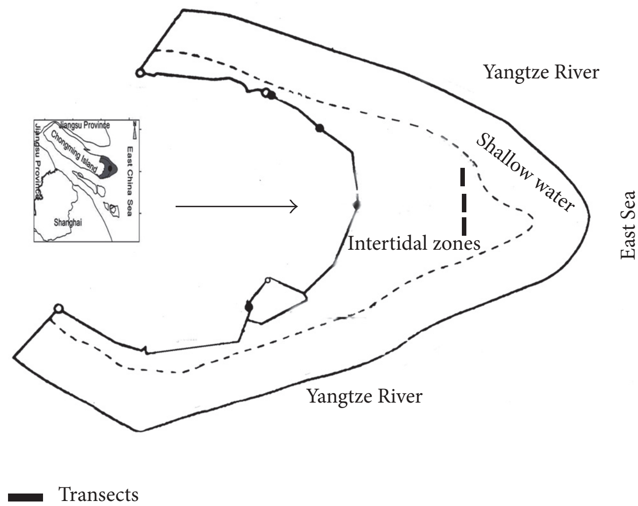

Figure 1: The transects setting in the middle-low tidal zone of Chongming Dongtan wetland, China.

2. Materials and Methods

2.1. Study Site. The field studies were conducted at the 32,600 ha Dongtan wetland (31°25′–31°38′N, 121°50′–122°05′E), which is located at the eastern end of Chongming island in the Yangtze River estuary [23]. Dongtan wetland includes both natural and artificial wetlands. The natural wetlands include the intertidal zones and the coastal shallow-water zones below the mean-low-water lines (Figure 1) [19]. The Dongtan wetland was very productive and affected by the semidiurnal tides. Due to the repeated flooding, the intertidal zones were divided into high tidal zone, middle tidal zone, and low tidal zone. The wetlands were 8 km wide at its maximum width in the intertidal zones, with the uppermost 2.5 km covered by marsh vegetation [16].

The average tidal range of Dongtan wetland was between 2.4 and 3.0 m. The high tidal zone was located between the high tidal level of springs and neaps and its altitude was above 2.5 m. The middle tidal zone was located between the high tidal level of neaps and the low tidal level of neaps and its altitude was between 2.5 m and 1 m. The low tidal zone was located between the low tidal level of springs and neaps and its altitude was below 1 m (Figure 2) [24].

With an elevation increase in the intertidal zones, the water content and the content of NaCl in the soil were significantly different. In the high tidal zone, the water content was approximately 34% to 35%, and the NaCl content was approximately 14 to 25 ppt. In the middle tidal zone, the water content was approximately 27%–32%, and the NaCl content was approximately 25 to 34 ppt. In the low tidal zone, the water content was approximately 33% to 39%, and the NaCl content was approximately 11 to 21 ppt [17].

Before the invasion, the plant communities of Dongtan wetland showed distinct zonation patterns, which were dominated by *Phragmites australis* in the high tidal zone and *Scirpus triqueter* in the low tidal zone [25]. After the invasion, *Scirpus triqueter* in the low tidal zone was completely replaced

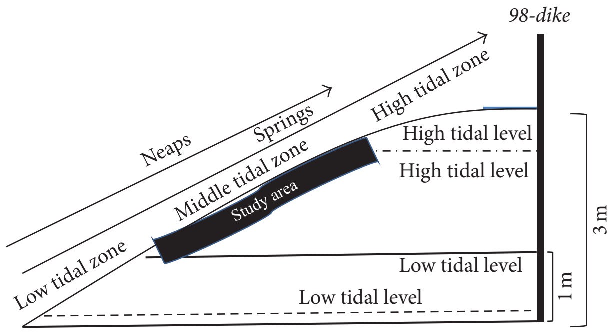

FIGURE 2: A sketch map on the tidal range and different tidal zones of Chongming Dongtan wetland, China.

by *Spartina alterniflora* [26], and *Spartina alterniflora* and *Phragmites australis* coexisted in the wide range of intertidal zones.

2.2. Mowing Treatments. We eliminated the aboveground parts of *Spartina alterniflora* and *Phragmites australis* from the plots by a sharp scissors for the artificial mowing treatments. Two intensities were set up: total-removal (TR) in which all of the individuals in plots were eliminated and half-removal (HR) in which half density of the individuals in plots was eliminated.

In July 2010, one single *Phragmites australis* transect, one single *Spartina alterniflora* transect, and one *Phragmites-Spartina* mixtures transect in the middle-low tidal zone of the Dongtan wetlands were set subjectively along the same horizontal line perpendicular to the flooding gradient. All of the three transects were 10 m in length and 3 m in width. Then the three transects were all divided into 30 plots and each plot was 1 m × 1 m. Among that, 10 plots in each transects were chosen randomly as control plots in which no plant individuals were eliminated, 10 plots in each transects were chosen randomly as HR plots, and 10 plots in each transects were chosen randomly as TR plots.

2.3. Competitive Intensity. We calculated the competitive intensity for the two species by comparing the performance of the plant in the single species plots and mixture species plots. We chose three measuring indices: relative neighbor effect (RNE), relative crowding (D_r), and interaction strength (I).

RNE is one of the most widely used indices for measuring the outcomes of competition on organisms [21, 27], which compares the performance of plants growing in neighbor-absence with neighbor-presence conditions. However, increased neighbor crowding may influence the performance of targets and the traditional RNE index does not consider the influence of different neighbor densities on the competitive intensity. Therefore, the RNE cannot distinguish between pure crowding and the actual strength of plant-plant interactions. Considering this limitation, we further calculated D_r and I of the two species, which were proposed by Wilson, to distinguish between the influence of pure crowding and the actual strength of plant-plant interactions on RNE. The RNE is equal to the product of the D_r by I. For a detailed modeling process, refer to [28].

RNE is calculated as follows:

$$\text{RNE} = \frac{(y_{\text{iso}} - y_{\text{mix}})}{\max |y_{\text{iso}}| \text{ or } |y_{\text{mix}}|}, \quad (1)$$

where y_{iso} is the performance of the target species in the single species plots. y_{mix} is the performance of the target species in the mixture species plots [27]. In our experiment, the performance of the target species was defined as the relative growth rate (RGR) of the highest individuals per day per plot.

D_r and I are calculated as follows:

$$D_r = \frac{z_{\text{mix}}}{\max (y_{\text{iso}}, y_{\text{mix}})}$$
$$I = \frac{(y_{\text{iso}} - y_{\text{mix}})}{z_{\text{mix}}}, \quad (2)$$

where z_{mix} is the abundance of neighbors surrounding the target plants. y_{iso} is the performance of a target plant grown in the single species plots, and y_{mix} is the performance of a target plant grown in the mixture species plots [28]. Similarly, the performance of the target species was defined as the RGR of the highest individuals per day per plot.

The RGR of the highest individuals per day per plot was calculated using

$$\text{RGR} = \frac{[\ln (M_2) - \ln (M_1)]}{(t_2 - t_1)}, \quad (3)$$

where M_2 and M_1, respectively, represent the aboveground biomass of the highest individuals in plots before and after the treatments and $t_2 - t_1$ is the number of days of the experiment. Here our experiment lasted for 120 days.

To calculate the RGR, we needed to know the target biomass both before and after the treatments. Therefore, six higher individuals per plot were selected as the highest individuals. They were marked and distributed randomly and evenly for the measurement of the target biomass both before and after the treatments. Therefore, the target biomass before and after the treatments here was the average value of the three higher individuals biomass. We mowed the aboveground parts of the targets and oven-dried them to a constant weight at 70°C to estimate the target biomass.

2.4. Statistics. Through above calculations, we compared the mean competitive intensities of *Phragmites australis* and *Spartina alterniflora* among different mowing treatments. Meanwhile, we also compared the mean height of highest individuals and mean number of *Phragmites australis* and *Spartina alterniflora* individuals per plot among different mowing treatments.

In our research, all statistical analyses were conducted using SAS 8.1 (SAS Institute Inc., USA). We analyzed the experimental results using an analysis of variance (ANOVA). Normality and homoscedasticity of all data were tested first. Data that violated these assumptions were ln-transformed to improve normality and homoscedasticity. The statistical tests were considered significant at the 0.05 probability level.

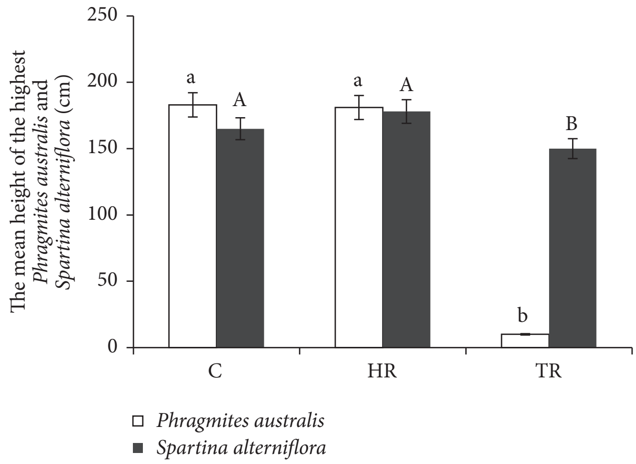

Figure 3: The height of the highest *Phragmites australis* and *Spartina alterniflora* (mean ± SE) in different mowing treatments: control (C), HR (half-removal), and TR (total-removal). Different letters indicate a significant difference ($P < 0.05$) among treatments.

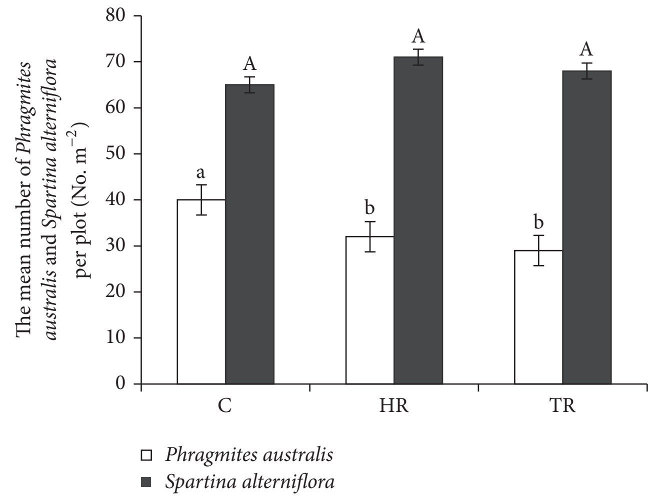

Figure 4: The number of *Phragmites australis* and *Spartina alterniflora* individuals per plot (mean ± SE) in different mowing treatments: control (C), HR (half-removal), and TR (total-removal). Different letters indicate a significant difference ($P < 0.05$) among treatments.

3. Results

3.1. The Effect of Different Mowing Intensities on Phragmites australis and Spartina alterniflora

3.1.1. The Height of Highest Individuals. The analysis of variance indicated that TR treatments significantly affected the height of the highest individuals of *Phragmites australis* and *Spartina alterniflora*. At the end of the treatments, the mean height of the highest individuals of *Phragmites australis* and *Spartina alterniflora* in the TR treatments was significantly lower than that in the controls ($P < 0.05$), but the mean height of the highest individuals of *Phragmites australis* and *Spartina alterniflora* in the HR treatments was similar to that in the control treatments (Figure 3). The result showed that only higher mowing intensity could affect the growth performance of *Phragmites australis* and *Spartina alterniflora*.

3.1.2. The Number of Individuals per Plot. The analysis of variance indicated that mowing treatments significantly affected the number of *Phragmites australis* individuals per plot ($P < 0.05$). In detail, the mean number of *Phragmites australis* individuals per plot was higher in the control treatments than that in the TR and HR treatments at the end of the treatments, but the mean number of *Phragmites australis* individuals per plot in the HR treatments was similar to that in the TR treatments. There were no significant differences in the number of *Spartina alterniflora* individuals per plot between the different mowing treatments and the controls (Figure 4).

3.2. Competitive Intensities of Phragmites australis and Spartina alterniflora on Each Other in Different Mowing Treatments

3.2.1. RNE. A negative competitive effect was found in the mixed community for *Phragmites australis* and *Spartina alterniflora*, and the effect of *Spartina alterniflora* on *Phragmites australis* was even more pronounced. The mean RNE of *Phragmites australis* and *Spartina alterniflora* on each other in the controls was 0.354 and 0.619, respectively.

The relationship between RNE and mowing treatments was not the same for *Spartina alterniflora* and *Phragmites australis*. The mean RNE of *Spartina alterniflora* on the targets in the HR and TR treatments was 0.963 and 1.000, respectively, and expected to be more intense than that in the controls. However, the mean RNE of *Phragmites australis* on the targets in the HR and TR treatments was 0.116 and 0.102, respectively, and expected to be more alleviative than that in the controls (Table 1).

3.2.2. D_r. The results showed that D_r of *Spartina alterniflora* on *Phragmites australis* was more pronounced than that of *Phragmites australis* on *Spartina alterniflora* in the controls. The mean D_r of *Phragmites australis* and *Spartina alterniflora* on each other in the controls was 1220 and 6881, respectively. Moreover, the relationship between D_r and mowing treatments was the same for *Spartina alterniflora* and *Phragmites australis*. That is, D_r of both *Spartina alterniflora* and *Phragmites australis* on the targets was expected to be more intense when there was a mowing disturbance.

3.2.3. I. The results showed that I of *Phragmites australis* on *Spartina alterniflora* was more pronounced than that of *Spartina alterniflora* on *Phragmites australis* in the controls. The mean I of *Phragmites australis* and *Spartina alterniflora* was 0.00029 and 0.00009, respectively, in the controls. Moreover, the relationship between I and mowing treatments was the same for *Spartina alterniflora* and *Phragmites australis*. That is, I of both *Spartina alterniflora* and *Phragmites australis* on the targets was expected to be more intense when there was no mowing disturbance.

Table 1: The RNE, D_r, and I of *Phragmites australis* and *Spartina alterniflora* in different mowing treatments ($n = 10$ plots) (mean ± SE).

	RNE	D_r	I
Spartina alterniflora			
Control	0.619 ± 0.018	6881 ± 239.145	0.00009 ± 0.000
HR	0.963 ± 0.041*	12400 ± 998.395**	0.00007 ± 0.000**
TR	1.000 ± 0.000**	15926 ± 1203.326**	0.00006 ± 0.000**
Phragmites australis			
Control	0.354 ± 0.046	1220 ± 101.444	0.00029 ± 0.000
HR	0.116 ± 0.039*	2318 ± 265.353**	0.00005 ± 0.0000**
TR	0.102 ± 0.026**	2547 ± 278.355**	0.00004 ± 0.000**

HR means half-removal treatments. TR means total-removal treatments.
$^{*}P < 0.05$; $^{**}P < 0.01$.

4. Discussions

In this study we investigated the response of plant-plant interactions between invasive and native species towards artificial mowing in a typical salt marsh of China. Our results found that artificial mowing had a positive influence on the competitive effect of *Spartina alterniflora* on *Phragmites australis* and *a* negative influence on the competitive effect of *Phragmites australis* on *Spartina alterniflora*.

4.1. Affecting Factors of Plant Competitive Abilities. One species may have specific traits that allow them to outcompete other species such as fast growth, rapid reproduction, high dispersal ability, phenotypic plasticity, and association with humans [29]. For *Spartina alterniflora* and *Phragmites* australis, difference of photosynthesis type may result in different plant growth abilities of the two species. *Phragmites australis* is a C_3 plant, which has the obvious photosynthesis midday depression phenomenon. While *Spartina alterniflora* is a C_4 plant with higher apparent quanta efficiency, carboxylic efficiency, and net photosynthetic rate [22]. In addition, *Phragmites australis* and *Spartina alterniflora* have different absorptive capacity for nitrogen [30, 31]. With the increase of the content of soil nitrogen, *Phragmites australis* is growing better. Therefore, the relative performance and competitive ability of plants also depend on their growing conditions [17].

Competitive advantage species may have strong genetic differentiation and phenotypic plasticity. Some researchers reported genetic diversity of *Spartina alterniflora* was very high and its Shannon genetic diversity index can reach 0.703. Among that, its intraspecific genetic diversity index was greater than that of interspecific genetic diversity, but there were also differentiation among populations [32]. In addition, some studies indicated that the phenotypic plasticity indices of *Spartina alterniflora* were higher than that of *Phragmites australis* for traits related to the morphology, growth, and biomass allocation in response to nitrogen and culm density [33, 34].

4.2. The Influence of Mowing on the Interspecific Interactions between Invasive and Native Species. In the researches of invasive mechanism, both species and ecosystem factors should be considered. Many hypotheses have been proposed to explain the success of plant invasions. For example, a common explanation is that invasive species outcompete their cooccurring natives [15, 35, 36]. However, some studies also found invasive species tended to have many similar traits compared with native species in some physical conditions [37].

Disturbance may have an influence on the competitive abilities of different species. Shifts in the relative availability of canopy resources versus soil resources might modulate interspecific competition and, therefore, the outcome of mowing on community structure [38]. Tissue loss and modified light profiles may be major causes of changes in establishment, growth, competitive success, and longevity [39]. Mowing hardly causes the instantaneous death of individuals but rather removes aboveground biomass and often acts selectively towards certain plant trait attributes, such as large height and palatable leaf tissue. Mowing is also supposed to change the relative importance of aboveground versus below-ground competition and thus potentially the competitive hierarchy among species [40]. Humans promoted or inhibited invaders through exerting their influence on the interspecific interactions of native and invasive species.

Disturbance has a long and recurring role as a potential explanation for the coexistence of species and the maintenance of patterns of species diversity. The human release hypothesis stated that the abundance of invasive species is different between different regions because population expansion is reduced in some regions through continuous land management and associated cutting of the invasive species [41–46]. Invaded ecosystems may have experienced disturbance, typically human-induced. Such a disturbance may give invasive species a chance to establish themselves with less competition from natives less able to adapt to a disturbed ecosystem.

4.3. Control Suggestions for Invasive Species. Concretely, some studies indicated controlled water-logging was an effective measure to the invasive plant *Spartina alterniflora* [47], and some studies indicated clipping vegetation at the early florescence stage and the integrated technique of cutting plus water-logging were more efficient for controlling the invasive plant *Spartina alterniflora* [48]. Moreover, 3S (GPS, RS, and GIS) technology, mathematical models of population growth and dispersal, and long-term monitoring systems have also been employed in *Spartina alterniflora* monitoring and controlling [49].

Spartina alterniflora may invade new places by quickly occupying more space and inhibiting the growth of *Phragmites australis*. Meanwhile, *Phragmites australis* might have a genetic competitive dominance over *Spartina alterniflora* because of its strong I. Therefore, the control of *Spartina alterniflora* by *Phragmites australis* is feasible. In addition, the competitive intensity of *Spartina alterniflora* would be increased, and the competitive intensity of *Phragmites australis* would be decreased if there was a disturbance. Artificial mowing disturbance may promote the spread of *Spartina alterniflora* in the Dongtan wetland. Meanwhile, grazing and human mowing are very common in this area, and these activities have important influences on the growth and colonize of *Spartina alterniflora* [50]. Therefore, we suggested that wetland managers should plan to establish nature reserves and prohibit the harvesting behavior of local residents for the better control of invasive *Spartina alterniflora*.

5. Conclusions

Our results found that *Spartina alterniflora* has a competitive dominance over *Phragmites australis* in the middle-low tidal zone of Dongtan wetland on the eastern coast of China. Our results also showed that RNE of *Spartina alterniflora* on *Phragmites australis* increased with human mowing intensities, but the RNE of *Phragmites australis* on *Spartina alterniflora* decreased with human mowing intensities.

In our study, D_r was higher for both *Spartina alterniflora* and *Phragmites australis* in the mowing treatments than that in the controls, but I was lower for both *Spartina alterniflora* and *Phragmites australis* in the mowing treatments than that in the controls. The change of RNE of *Spartina alterniflora* with the mowing treatments was the same as the change of D_r of *Spartina alterniflora* with the mowing treatments. And the change of RNE of *Phragmites australis* with the mowing treatments was the same as the change of I of *Phragmites australis* with the mowing treatments. RNE of *Spartina alterniflora* may be mainly determined by its D_r, and RNE of *Phragmites australis* may be mainly determined by its I.

Competing Interests

The authors declare that they have no competing interests.

Acknowledgments

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References

[1] A. Hastings, R. J. Hall, and C. M. Taylor, "A simple approach to optimal control of invasive species," *Theoretical Population Biology*, vol. 70, no. 4, pp. 431–435, 2006.

[2] D. F. M. Hammecher, "Classical biological control of the invasive coconut mite in Africa and Asia: integrating biological, molecular and socioeconomic studies in the search and introduction of effective natural enemies," 2008.

[3] M. A. Bowker, S. Soliveres, and F. T. Maestre, "Competition increases with abiotic stress and regulates the diversity of biological soil crusts," *Journal of Ecology*, vol. 98, no. 3, pp. 551–560, 2010.

[4] A. Fenesi, J. Geréd, S. J. Meiners, B. Tóthmérész, P. Török, and E. Ruprecht, "Does disturbance enhance the competitive effect of the invasive Solidago canadensis on the performance of two native grasses?" *Biological Invasions*, vol. 17, no. 11, pp. 3303–3315, 2015.

[5] A. A. M. MacDonald and P. M. Kotanen, "The effects of disturbance and enemy exclusion on performance of an invasive species, common ragweed, in its native range," *Oecologia*, vol. 162, no. 4, pp. 977–986, 2010.

[6] C. R. Whitcraft, L. A. Levin, D. Talley, and J. A. Crooks, "Utilization of invasive tamarisk by salt marsh consumers," *Oecologia*, vol. 158, no. 2, pp. 259–272, 2008.

[7] M. W. Hester, I. A. Mendelssohn, and K. L. McKee, "Species and population variation to salinity stress in Panicum hemitomon, Spartina patens, and Spartina alterniflora: morphological and physiological constraints," *Environmental and Experimental Botany*, vol. 46, no. 3, pp. 277–297, 2001.

[8] J. Adams, E. van Wyk, and T. Riddin, "First record of Spartina alterniflora in southern Africa indicates adaptive potential of this saline grass," *Biological Invasions*, vol. 18, no. 8, pp. 2153–2158, 2016.

[9] Q. Wang, S.-Q. An, Z.-J. Ma, B. Zhao, J.-K. Chen, and B. Li, "Invasive *Spartina alterniflora*: biology, ecology and management," *Acta Phytotaxonomica Sinica*, vol. 44, no. 5, pp. 559–588, 2006.

[10] J. Chen, L. Wang, Y. Li, W. Zhang, X. Fu, and Y. Le, "Effect of Spartina alterniflora invasion and its controlling technologies on soil microbial respiration of a tidal wetland in Chongming Dongtan, China," *Ecological Engineering*, vol. 41, pp. 52–59, 2012.

[11] H. Li and L. Zhang, "An experimental study on physical controls of an exotic plant Spartina alterniflora in Shanghai, China," *Ecological Engineering*, vol. 32, no. 1, pp. 11–21, 2008.

[12] D. Xiao, L. Zhang, and Z. Zhu, "A study on seed characteristics and seed bank of Spartina alterniflora at saltmarshes in the Yangtze Estuary, China," *Estuarine, Coastal and Shelf Science*, vol. 83, no. 1, pp. 105–110, 2009.

[13] Y. Gao, L. Tang, J. Wang et al., "Clipping at early florescence is more efficient for controlling the invasive plant Spartina alterniflora," *Ecological Research*, vol. 24, no. 5, pp. 1033–1041, 2009.

[14] R. M. Chambers, D. T. Osgood, and N. Kalapasev, "Hydrologic and chemical control of Phragmites growth in tidal marshes of SW Connecticut, USA," *Marine Ecology Progress Series*, vol. 239, pp. 83–91, 2002.

[15] B. Li, C.-H. Liao, X.-D. Zhang et al., "Spartina alterniflora invasions in the Yangtze River estuary, China: an overview of current status and ecosystem effects," *Ecological Engineering*, vol. 35, no. 4, pp. 511–520, 2009.

[16] Y. Yuan, K. Wang, D. Li et al., "Interspecific interactions between phragmites australis and spartina alterniflora along a tidal gradient in the Dongtan wetland, Eastern China," *PLoS ONE*, vol. 8, no. 1, Article ID e53843, 2013.

[17] Q. Wang, C. H. Wang, B. Zhao et al., "Effects of growing conditions on the growth of and interactions between salt marsh plants: implications for invasibility of habitats," *Biological Invasions*, vol. 8, no. 7, pp. 1547–1560, 2006.

[18] C. Zhao, Z. Deng, C. Zhou, B. Guan, S. An, and L. Chen, "Effects of nitrogen availability and competition of leaf characteristics of *Spartina alterniflora* and *Phragmites australis*," *Journal of Plant Ecology (Chinese Version)*, vol. 32, pp. 392–401, 2008.

[19] Z. Ma, Y. Wang, X. Gan, B. Li, Y. Cai, and J. Chen, "Waterbird population changes in the wetlands at chongming dongtan in the yangtze river estuary, China," *Environmental Management*, vol. 43, no. 6, pp. 1187–1200, 2009.

[20] J. Chen, B. Zhao, W. Ren et al., "Invasive Spartina and reduced sediments: Shanghai's dangerous silver bullet," *Journal of Plant Ecology*, vol. 1, no. 2, pp. 79–84, 2008.

[21] J. M. Huckle, J. A. Potter, and R. H. Marrs, "Influence of environmental factors on the growth and interactions between salt marsh plants: effects of salinity, sediment and waterlogging," *Journal of Ecology*, vol. 88, no. 3, pp. 492–505, 2000.

[22] G.-Q. Zhao, L.-Q. Zhang, and X. Liang, "A comparison of photosynthetic characteristics between an invasive plant Spartina alterniflora and an indigenous plant Phragmites australis," *Acta Ecologica Sinica*, vol. 25, no. 7, pp. 1604–1611, 2005.

[23] Q. Yan, J.-J. Lu, and W.-S. He, "Succession character of salt marsh vegetations in Chongming Dongtan wetland," *Chinese Journal of Applied Ecology*, vol. 18, no. 5, pp. 1097–1101, 2007.

[24] C. Zhao, *Evolution Analysis of Chongming Dongtan, North Section and Eastern Hengsha Shoal in the Mouth of the Yangtze River*, East China Normal University, Shanghai, China, 2006.

[25] Y. He, X. Li, Z. Ma, Y. Sun, and Y. Jia, "Vegetation zonation related to the edaphic factors in the East headland of Chongming Island," *Acta Ecologica Sinica*, vol. 30, no. 18, pp. 4919–4927, 2010.

[26] B. Zhao, Y. Yan, H. Guo, M. He, Y. Gu, and B. Li, "Monitoring rapid vegetation succession in estuarine wetland using time series MODIS-based indicators: an application in the Yangtze River Delta area," *Ecological Indicators*, vol. 9, no. 2, pp. 346–356, 2009.

[27] J. H. Markham and C. P. Chanway, "Measuring plant neighbour effects," *Functional Ecology*, vol. 10, no. 4, pp. 548–549, 1996.

[28] M. V. Wilson, "Measuring the components of competition along productivity gradients," *Journal of Ecology*, vol. 95, no. 2, pp. 301–308, 2007.

[29] C. P. Mudzengi, C. Murungweni, E. Dahwa, X. Poshiwa, S. Kativu, and S. M. D. And, "Woody species composition and structure in a semi-arid environment invaded by *Dichrostachys cinerea* (I.) Wight and Arn (Fabaceae)," *International Journal of Scientific and Research Publications*, vol. 3, no. 9, pp. 1–10, 2013.

[30] M. A. Rickey and R. C. Anderson, "Effects of nitrogen addition on the invasive grass Phragmites australis and a native competitor Spartina pectinata," *Journal of Applied Ecology*, vol. 41, no. 5, pp. 888–896, 2004.

[31] A. C. Tyler, J. G. Lambrinos, and E. D. Grosholz, "Nitrogen inputs promote the spread of an invasive marsh grass," *Ecological Applications*, vol. 17, no. 7, pp. 1886–1898, 2007.

[32] C. E. Proffitt, S. E. Travis, and K. R. Edwards, "Genotype and elevation influence *Spartina alterniflora* colonization and growth in a created salt marsh," *Ecological Applications*, vol. 13, no. 1, pp. 180–192, 2003.

[33] Y. J. Zhao, H. Qing, C. J. Zhao et al., "Phenotypic plasticity of Spartina alterniflora and Phragmites australis in response to nitrogen addition and intraspecific competition," *Hydrobiologia*, vol. 637, pp. 143–155, 2010.

[34] Y. Yuan, D.-Z. Li, and K.-Y. Wang, "Research progress in mutual invasion of Phragmites australis and Spartina alterniflora communities," *Wetland Science*, vol. 12, no. 4, pp. 533–538, 2014.

[35] M. Williamson and A. Fitter, "The varying success of invaders," *Ecology*, vol. 77, no. 6, pp. 1661–1666, 1996.

[36] Z. Deng, S. An, Y. Zhi et al., "Preliminary studies on invasive model and outbreak mechanism of exotic species, Spartiona alterniflora Loisel," *Acta Ecologica Sinica*, vol. 26, pp. 2678–2686, 2006.

[37] M. D. Smith and A. K. Knapp, "Physiological and morphological traits of exotic, invasive exotic, and native plant species in tallgrass prairie," *International Journal of Plant Sciences*, vol. 162, no. 4, pp. 785–792, 2001.

[38] S. Škornik, M. Vidrih, and M. Kaligarič, "The effect of grazing pressure on species richness, composition and productivity in North Adriatic Karst pastures," *Plant Biosystems*, vol. 144, no. 2, pp. 355–364, 2010.

[39] J. Bokdam and J. M. Gleichman, "Effects of grazing by free-ranging cattle on vegetation dynamics in a continental north-west European heathland," *Journal of Applied Ecology*, vol. 37, no. 3, pp. 415–431, 2000.

[40] F. May, V. Grimm, and F. Jeltsch, "Reversed effects of grazing on plant diversity: the role of below-ground competition and size symmetry," *Oikos*, vol. 118, no. 12, pp. 1830–1843, 2009.

[41] S. A. Levin and R. T. Paine, "Disturbance, patch formation, and community structure," *Proceedings of the National Academy of Sciences of the United States of America*, vol. 71, no. 7, pp. 2744–2747, 1974.

[42] M. Slatkin, "Competition and regional coexistence," *Ecology*, vol. 55, no. 1, pp. 128–134, 1974.

[43] P. S. Petraitis, R. E. Latham, and R. A. Niesenbaum, "The maintenance of species diversity by disturbance," *Quarterly Review of Biology*, vol. 64, no. 4, pp. 393–418, 1989.

[44] D. Tilman, "Constraints and tradeoffs: toward a predictive theory of competition and succession," *Oikos*, vol. 58, no. 1, pp. 3–15, 1990.

[45] P. Chesson and N. Huntly, "The roles of harsh and fluctuating conditions in the dynamics of ecological communities," *American Naturalist*, vol. 150, no. 5, pp. 519–553, 1997.

[46] K. Shea, S. H. Roxburgh, and E. S. J. Rauschert, "Moving from pattern to process: coexistence mechanisms under intermediate disturbance regimes," *Ecology Letters*, vol. 7, no. 6, pp. 491–508, 2004.

[47] L. Yuan and L. Zhang, "Effects of waterlogging on the physiology of *Spartina alterniflora*," *Oceanologia et Limnologia Sinica*, vol. 41, pp. 175–179, 2010.

[48] L. Tang, Y. Gao, J. Wang et al., "Designing an effective clipping regime for controlling the invasive plant Spartina alterniflora in an estuarine salt marsh," *Ecological Engineering*, vol. 35, no. 5, pp. 874–881, 2009.

[49] S. Sun, X. Zhu, and C. Lue, "Function assessment and ecological controls on alien species cordgrass," *Chinese Journal of Ecology*, vol. 23, pp. 93–98, 2004.

[50] T. Zhang, *Analysis on the Landscape Pattern and Change of Chongming Dongtan*, East China Normal University, Shanghai, China, 2004.

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Research Article

Simulation of the Effect of Artificial Water Transfer on Carbon Stock of *Phragmites australis* in the Baiyangdian Wetland, China

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How to explain the effect of seasonal water transfer on the carbon stocks of Baiyangdian wetland is studied. The ecological model of the relationship between the carbon stocks and water depth fluctuation of the reed was established by using STELLA software. For the first time the Michaelis-Menten equation (1) introduced the relation function between the water depth and reed environmental carrying capacity, (2) introduced the concept of suitable growth water depth, and (3) simulated the variation rules of water and reed carbon stocks of artificial adjustment. The model could be used to carry out the research on the optimization design of the ecological service function of the damaged wetland.

1. Introduction

Water has an important role in the growth of wetland plants [1–4]. Changes in wetland water content influence surface vegetation type and individual form, which in turn affect both the demand for and the utilization efficiency of water. Because water is involved in processes such as plant photosynthesis and respiration, it can affect directly or indirectly the fixation and release of CO_2 by vegetation [5–7]. Wetland plant biomass reflects the fixed shape, size, and productivity of ecosystem carbon [5]. Thus, changes in artificial or ecological water, or other human activities that alter the hydrological situation of wetlands, can influence wetland photosynthetic processes and directly or indirectly affect vegetation carbon reserves.

Artificial water transfer has become one of the most direct and effective approaches for the management of water quantity in wetlands, especially in temperate regions and in areas with scant water resources, for example, the Baiyangdian wetland, Zhalong wetland, and Lake Wuliangsuhai in China. Each year, water in these areas is regulated artificially to ensure the appropriate water depth to maintain wetland ecological function [8]. Previous research of vegetation growth in wetland ecological systems has elucidated many aspects of the carbon cycle [9–14]. However, most studies have considered natural wetlands rather than those affected by artificial water transfer. Therefore, it is necessary to investigate the effect of the artificial regulation of water depth on wetland plant growth and to consider the relationship between plant carbon stock and water depth fluctuation, to establish the consequences on the carbon cycle and carbon flux of wetlands.

Usually, the process of photosynthesis is expressed using the Michaelis-Menten equation [2, 4, 13, 14]. Based on previous research, this study introduced into the Michaelis-Menten equation the relation between water depth and the environmental carrying capacity of reeds, to elucidate the effect of environmental capacity on the carbon stock of reeds (biomass carbon content) in terms of water depth fluctuation.

Figure 1: Typical landscape pattern of Baiyangdian wetland, phragmites platform.

In this paper, the most important hydrological conditions are studied, which are also the most easily controlled and influenced by human factors, which are one of the most effective means to restore the damaged wetland ecosystem services. The Baiyangdian *Phragmites australis* (reed) wetland under the influence of artificial water transfer is studied, and the ecological model of the relationship between the carbon stocks and water depth fluctuation of the reed was established by using STELLA software. We try to understand how to explain the effect of seasonal water transfer on the carbon stocks of Baiyangdian wetland and how to control the water transfer time and water transfer quantity (water depth) in Baiyangdian.

2. Study Area and Method

2.1. Study Area. Baiyangdian wetland is located in the middle of Hebei Province in China. It is the largest freshwater lake wetland of the North China Plain (38°43′–39°02′N, 115°38′–116°07′E), covering an area of 366 km^2 (Dagu water level 10.5 m [15], which is zero in 1902 in Tanggu of Tianjin setting up a tide gauge as the basis). This area has a temperate continental monsoon climate that is cold and dry in winter but hot and wet in summer. The wetland area has more than 3700 ditches between connected platforms of *Phragmites australis* and 143 lakes (Figure 1). Its elevation is about 7.5–8.5 m and the average water depth is 1.0–2.0 m. Apart from a small number of poplar and peach trees planted on the edge of the Baiyangdian wetland, the majority of vegetation of this wetland ecosystem is *P. australis*, and this comprises the main resource for the local community.

Since the 1990s, due to the impact of environmental changes and human factors, the water depth in the Baiyangdian wetland has exhibited annual and seasonal fluctuations. The monthly fluctuation in the range of water depth in drought and wet years is 0.93–1.04 m and 1.35–1.56 m, respectively [16]. When the groundwater depth is <6.5 m (Dagu), it is called semidry and when it is <5.5 m (Dagu), it is called dry [17] (where level 6.5 m in Baiyangdian wetland means the water depth is about zero meters). In order to maintain the minimum ecological water level (7.3 m) [17] necessary to preserve ecological services in the Baiyangdian wetland, the local government implements an ecological water replenishment project. According to statistics (2001–2011), average annual replenishment with 80 million cubic meter of water [8] was required to maintain the Baiyangdian wetland water level at 6.6–8.6 m [18] (Dagu 8.6 m). To a certain extent, artificial water transfer modifies the hydrological situation of the Baiyangdian wetland, and it affects both the structure of the wetland and the growth status of the reeds. Therefore, it is necessary to determine the optimum time to implement artificial water transfer to maintain the water level within the prescribed design parameters and to have maximum effect on the carbon stock capacity of the reeds, in order to restore and maintain wetland ecological service function.

2.2. Data Sources. Carbon stock validation data was assessed in reeds sampled from June to 10th month of 2009 from near Yuanyang Island in the Baiyangdian wetland that was published by Li et al. [19]. The initial values of the aboveground and underground carbon stock of the reeds and the carbon stock of the litter were obtained by analogy [15]. The relationship between the aboveground carbon stock and the average water depth in the growing season was verified in 2010 by Zhao [18]. (In 2009 and 2010, the growth of the Baiyangdian wetland reeds reflected water level.)

This study collated ground vegetation carbon stock data from the Baiyangdian wetland (Shihoudian) [20], Zhalong wetland [21], and Sanjiang source *angustifolia* meadow wetland [22] to test the model.

Environmental temperature, solar radiation, and other parameters were obtained from the local meteorological bureau of Anxin County [15]. Details of the hydrological conditions and water transfer data were obtained from the Baoding Municipal Water Affairs Bureau.

3. Ecological Model Description

Plants absorb atmospheric carbon dioxide (CO_2) via photosynthesis, which is then transformed and fixed in the plant. During life, the plant body transfers carbon from the rhizome of the root to the ground, and plant respiration releases carbon into the atmosphere [14, 23]. The decay of plant litter and of the root and stem after the death of the plant releases carbon into the soil [24]. In this study, based on the carbon cycle process, the STELLA software (version 9.1.3) was used both to build the relationship model between the underground carbon stock and water level fluctuation and to simulate the changes of carbon stock. The model was established with two state variables: aboveground tissue carbon stock (PaC) and underground carbon stock (PbC) (unit: gC/m^2). The unit of the modeled process was gC/m/d, and the period of simulation was April 1 to October 31.

The forcing function of the model was CO_2 concentration in ambient air, ambient temperature, solar radiation, and water depth. Abbreviations summarizes the symbols used in

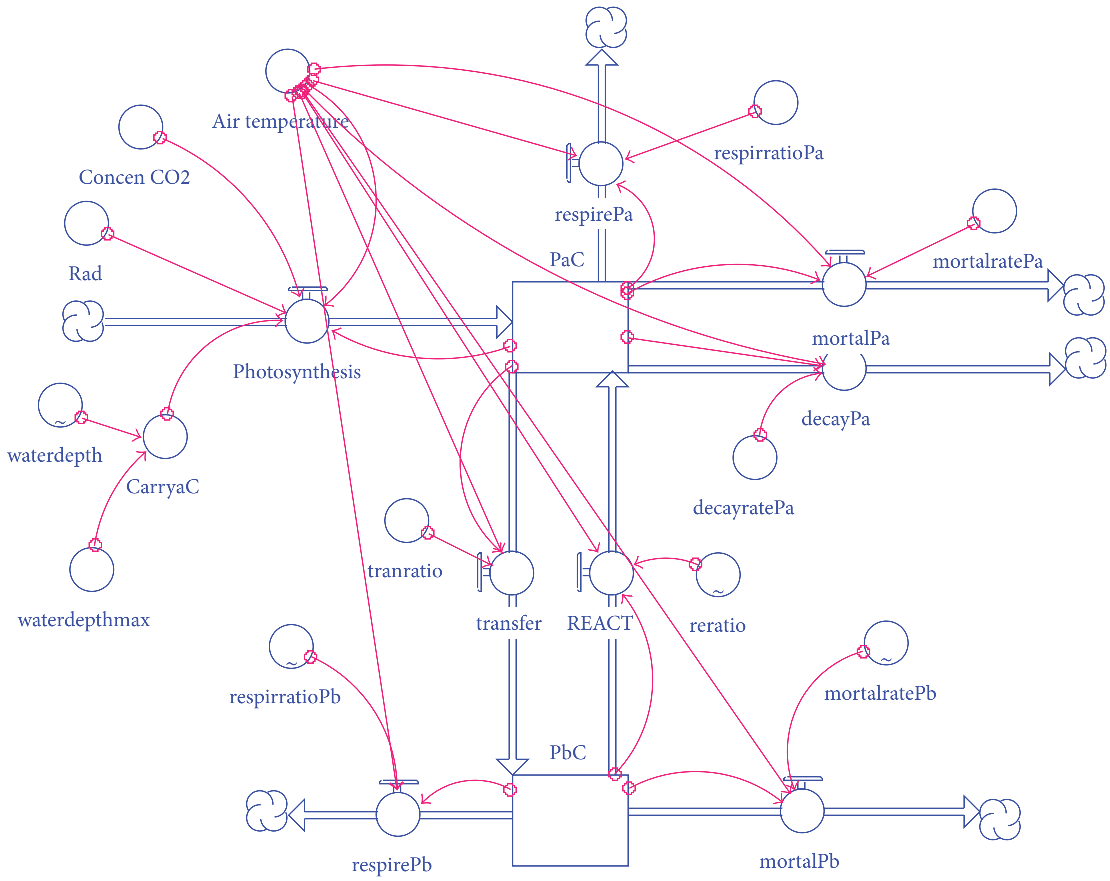


Figure 2: Conceptual diagram of STELLA model.

the model, their meanings, and their units. All processes in the model were affected and controlled by temperature. The temperature in this study was expressed using the Arrhenius equation. Temperature, water depth, rate of CO_2 release, rate of plant litter organization on the ground, underground death rate, reed carbon reserves from the earth to the underground transport rate, and reed root on tissue like reactivation rate parameters are all time-dependent dynamic parameters on which the STELLA model was formulated. Figure 2 shows the conceptual schematic of the STELLA model (see Abbreviations for explanation of the symbols).

3.1. Ecological Model Process and Equations

3.1.1. Photosynthesis Process. The earth has only green plants (and photosynthetic bacteria) through photosynthesis and light interception energy directly from the sun, and they will use inorganic material (carbon dioxide) reduction of organic matter, as their own food. Reed leaf photosynthesis, that is, the use of solar radiation for CO_2 assimilation of carbohydrates, is the most important source of energy in the ecosystem and the most important process in the model, which is expressed using the Michaelis-Menten equation [13, 25]. The Michaelis-Menten equation can change effects of solar radiation and air CO_2 concentration on photosynthetic process; the rate of plant photosynthesis changes with atmospheric CO_2 concentration and thus a constant semisaturated concentration of 300 ppm was introduced here [25], and it was considered that the vegetation would capture about 6% of the solar radiation available to the plant biomass [26].

The photosynthesis process of reeds was expressed by logistic growth, and the concept of environmental carrying capacity (CarryaC) was introduced, where we use the logistic equation to limit growth. That is, the aboveground carbon stocks of the reed are not of exponential growth but rise to the maximum smoothly. Because the growth of reeds is restricted by space and nutrient resources, the ultimate reed growth reaching the maximum capacity of the environment will slow down; this is the ecological significance of logistic equation. According to relevant research [4, 14], the process should be applied to the first-order equation and the influence of factors (1−PaC/CarryaC) on the organization of carbon stock/environmental carrying capacity. When the aboveground carbon stock of the reed reaches the limit of the environmental carrying capacity, the reed will stop growing. The model assumed that the maximum value of organic carbon in the reed was 2450 gCm^{-2} [18, 27, 28].

In this study, the relationship between water depth and environmental carrying capacity was introduced for the first time, that is, CarryaC is a function of water depth. The suitable growth water depth was introduced for the water depth of the wetland, that is, the minimum ecological water depth.

Monitoring data of average water depth during the growth season of the reeds in the Baiyangdian wetland [18], other related research (showing the minimum ecological water level of 7.3 m, corresponding to the water depth of 0.8 m) [16, 17], and the special characteristics of the Baiyangdian reeds indicate that the water depth of 0.8 m is suitable for reed growth.

3.1.2. Reactivation of the Underground Root of the Reed by the Aboveground Tissue. The reactivation process (REACT) of the aboveground tissue on the underground tissue of the reed rhizome begins during the initial stage of the growth season. It is a function of the carbon stock, reactivation rate, time, and temperature of the underground structure of the reed.

3.1.3. Transmission Process of Aboveground Organic Matter of the Reed to Underground Tissue. As plant leaves begin to grow, the transmission process (transfer) of organic matter produced by photosynthesis to underground tissue occurs continuously throughout the growing season. After the cessation of photosynthesis, the energy of the leaf and stem of the reed [14] is redistributed underground. The process of energy transfer from aboveground to underground is a function of the carbon stock, transfer rate, time, and temperature of the plant, which is expressed in a first-order reaction equation in the model.

3.1.4. Death Process of the Aboveground Tissue of the Reed, Decay of Litter, and Death of Underground Tissue of the Reed. The processes of mortalPa, decayPa, and mortalPb are all functions of the carbon stock, death rate, apoptosis rate, time, and temperature, which are expressed by first-order reaction equations.

3.1.5. Respiration Process of Aboveground and Underground Tissue of the Reed. The processes respirPb and respirPa are determined by temperature, respiration rate, and carbon stock, which are expressed in the model with firs-order reaction equations.

The basic equations of the ecological model are shown as follows.

Basic equation

$$d\mathrm{PaC}/dt = \mathrm{PaC}(t - dt) + (\mathrm{Photosynthesis} + \mathrm{react} - \mathrm{transfer} - \mathrm{respirPa} - \mathrm{decayPa} - \mathrm{mortalPa}) * dt$$

$$d\mathrm{PbC}/dt = \mathrm{PbC}(t - dt) + (\mathrm{transfer} - \mathrm{react} - \mathrm{respirPb} - \mathrm{mortalPb}) * dt$$

Photosynthesis process

$$\mathrm{Photosynthesis} = \mathrm{Concen\_CO_2}/(\mathrm{Concen\_CO_2} + 300) * (\mathrm{Rad}/(\mathrm{Rad} + 6)) * \theta\hat{}(\mathrm{Air\_temperature} - 20) * \mathrm{PaC} * (1 - \mathrm{PaC}/\mathrm{CarryaC})$$

CarryaC and water depth relation function

If water depth = 0.8 then CarryaC = 2450 else CarryaC = 2450 ∗ (waterdepthmax − ABS((waterdepth − 0.8)/2))/waterdepthmax

Retransmission process

$$\mathrm{Transfer} = \mathrm{tranratio} * \mathrm{PaC} * \theta\hat{}(\mathrm{Air\_temperature} - 20)$$

Reactivation process:

$$\mathrm{REACT} = \mathrm{reratio} * \mathrm{PbC} * \theta\hat{}(\mathrm{Air\_temperature} - 20)$$

Respiration process

$$\mathrm{respirPa} = \mathrm{mrespiratioPa} * \mathrm{PaC} * \theta\hat{}(\mathrm{Air\_temperature} - 20)$$

$$\mathrm{respirPb} = \mathrm{mrespiratioPb} * \mathrm{PbC} * \theta\hat{}(\mathrm{Air\_temperature} - 20)$$

Litter and death process

$$\mathrm{mortalPa} = \mathrm{mortalatePa} * \mathrm{PaC} * \theta\hat{}(\mathrm{Air\_temperature} - 20)$$

$$\mathrm{decayPa} = \mathrm{mortalratePa} * \mathrm{PaC} * \theta\hat{}(\mathrm{Air\_temperature} - 20)$$

$$\mathrm{mortalPb} = \mathrm{mortalatePb} * \mathrm{PbC} * \theta\hat{}(\mathrm{Air\_temperature} - 20)$$

4. Results

4.1. Model Parameters. Research on the dynamic changes of aboveground and underground reed carbon stock provides a large amount of data for the calibration of the ecological model. Other parameters used in the model calibration were derived from literature [14, 24, 29–32]. The model was calibrated by repeated testing until the parameters demonstrated good agreement with observed values. Table 1 summarizes the parameter values of the model after calibration.

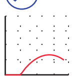
4.2. Model Verification. Based on the model calibration results, the observed and simulated values were compared and analyzed. Figures 3(a) and 3(b) show the results of the verification of the carbon stock in the aboveground and underground tissues of Baiyangdian reeds. The modeled value for 2009 is in good agreement with the observed value, reflecting the change of carbon stock of the reed during the growing season [20, 28]. The reeds grew steadily during the first three to four months of the growing season and then grew rapidly during months four and five, after which growth appeared to stabilize. The aboveground carbon stock reached its peak in August and underground carbon stock peaked in October.

Figure 4 shows that the relationships of the simulated and observed aboveground carbon stock of reeds with water depth in the Baiyangdian wetland are in good agreement. Furthermore, it can be seen that an increase or decrease of depth relative to the suitable growth water depth of the reed caused a reduction of carbon stock. This could reflect the tissue carbon stock related to the growth of the reed platform is associated with seasonal changes of the average

Table 1: Basic parameters used in the model.

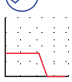
Parameters	Range	Unit	Source
INIT Pac = 103	100~150	gCm^{-2}	Libo 2012;
INIT Pbc = 1650	1500~2000	gCm^{-2}	Libo 2012;
INIT Concen CO_2 = 400	280~404.83	ppm per year	http://www.carbonify.com/carbon-dioxide-levels.htm
θ^a = 1.05~1.09			Calibrate and Calibrate;
CarryaC = 2450	1000~5000	gCm^{-2}	Libo 2012; Zhao 2012; Guo 2012; Zhang et al. 2014;
waterdepthmax = 1.8	0.0~2.0	m	Zhao et al. 2005; Cui et al. 2010; Zhao 2012;
Rad = 14.67	14.67	MJ m^{-2} per year	Collect;
decayratePa = 0.005	0.0~0.18	gg^{-1} per day	Calibrate;
mortalratePa = 0.001	0.0~0.15	gg^{-1} per day	Soetaert et al. 2004, Eid et al. 2012;
mortalratePb = 0.001	0.0~1.0	gg^{-1} per day	
transratio = 0.20	0.0~0.35	gg^{-1} per day	Zhang et al. 2014;
carbratio = 0.4	0.0~0.5	gg^{-1} per day	Calibrate;
mathratio = 0.006	0.0~0.3	gg^{-1} per day	Calibrate;

mortalratePb = GRAPH (TIME)



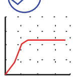
(1.00, 0.00), (30.9, 0.00), (60.7, 0.00), (90.6, 0.000325), (120, 0.0006), (150, 0.00069), (180, 0.000645), (210, 0.00051)

reratio = GRAPH (TIME)



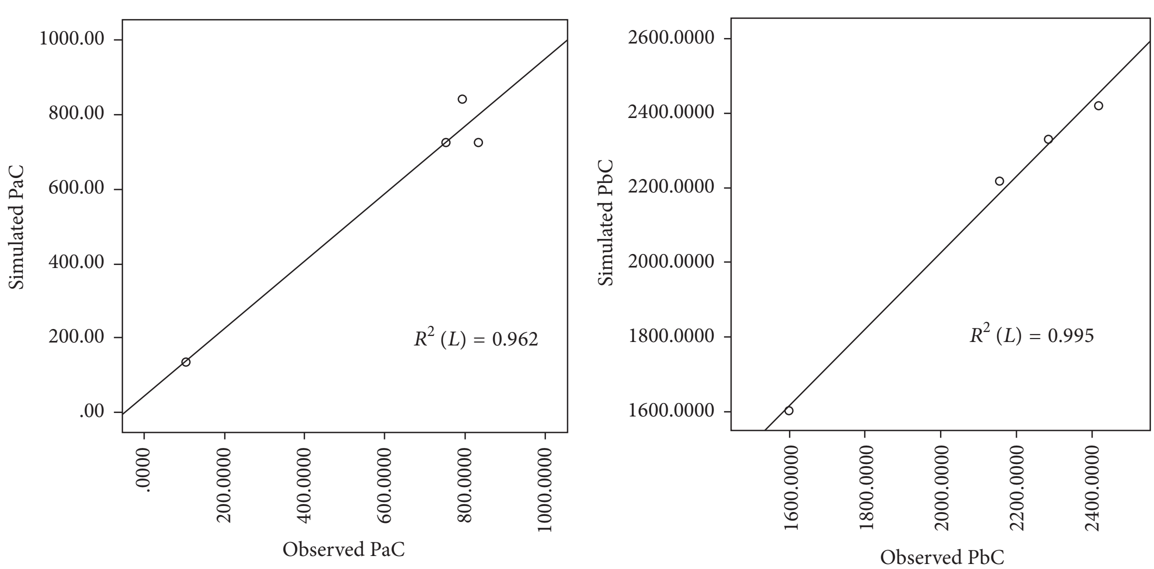
(1.00, 0.0008), (30.9, 0.00079), (60.7, 0.00079), (90.6, 0.00079), (120, 0.00078), (150, 0.00), (180, 0.00), (210, 0.00)

respirratioPb = GRAPH (TIME)



(1.00, 0.003), (30.9, 0.019), (60.7, 0.054), (90.6, 0.0585), (120, 0.059), (150, 0.059), (180, 0.059), (210, 0.058)

[a] θ is the Arrhenius constant, Soetaert et al. 2004, Zhang et al. 2014, and Jørgensen and Nielsen 2015; θ is 1.09 in photosynthesis and respirePb; θ is 1.07 in respirePa, transfer, and REACT; θ is 1.05 in mortalPa, mortalPb, decayPa, methane emission, and carbon emission.



(a) Aboveground carbon stocks (2009a) (b) Underground carbon stocks (2009a)

Figure 3: Comparison of observed and simulated carbon stock in Baiyangdian wetland in 2009: (a) aboveground plant tissue and (b) belowground plant tissue.

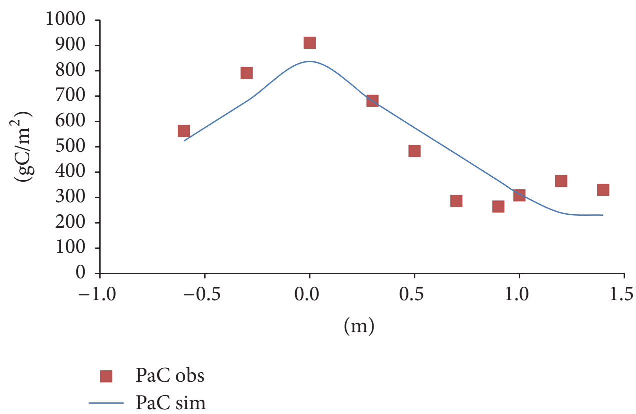


Figure 4: Comparison of the relationships of the simulated and observed carbon stock of aboveground plant tissue of reeds and water depth of the Baiyangdian wetland in 2009.

water depth. The 0 m relative water depth in the Baiyangdian wetland is defined as the suitable growth water depth of the reed [18] (corresponding water depth of 0.8 m), which is consistent with the largest carbon stock of the reeds.

4.3. Model Validation. The model was applied in the Zhalong wetland (Figure 5(a), 2010), Sanjiang Plain wetland (Figure 5(b), 2012), and Baiyangdian wetland (Figure 5(c), 2012). Analysis established that with suitable adjustment of the temperature, respiration, and release rate parameters of the model, the change trends of the simulated and observed values aboveground carbon stock were consistent. Thus, this model could be applied in equivalent calculations for similar wetlands.

4.4. Simulation Prediction. Based on the model validation and verification, the water depth conditions of different seasons (within the range of 1.0 m) were set, and the conditions of artificial water transfer in the Baiyangdian wetland were simulated. The change trend of reed carbon stock was predicted for the entire growing season under different water depths (i.e., 0.1, 0.3, 0.5, and 0.8 m) and in different seasons (spring, summer, fall, winter harvest, and treatment) with different water depths (0.1, 0.3, 0.5, and 0.8 m).

4.4.1. Entire Growing Season. Under conditions of suitable growth water depth, fluctuation of water depth caused by water transfer will not be conducive to reed growth and there will be a decrease in carbon stock (Figure 6). For water depth change of ±0.1 m, the corresponding changes of carbon stock of aboveground and underground are from −1.8% to −9.4% and from 0.2% to −9.8%, respectively. For water depth change of ±0.3 m, the corresponding changes of carbon stock of aboveground and underground tissue are from −1.8% to −29.5% and from 0.2% to −29.8%, respectively. For water depth change of ±0.5 m, the corresponding changes of carbon stock of aboveground and underground tissue are from −1.8% to −49.6% and from 0.1% to −49.9%, respectively. For water depth change of ±0.8 m, the corresponding changes of carbon stock of aboveground and underground tissue are from −1.8% to −50.0% and from 0.0% to −50.0%.

The simulation results indicate that artificial regulation of water has considerable effect on carbon content during the growth season of the reeds. Therefore, water depth increases due to artificial water transfer which should be limited to within 0.5 m (preferably to within 0.3 m) in order to reduce the influence of water depth fluctuation on the carbon stock of the reeds.

4.4.2. Different Seasons. Figure 7(a) shows that artificial water transfer in spring caused a reduction in the carbon stock of the reeds during the same period (the largest reduction in aboveground and underground carbon stock was from −5.9% to −48.3% and from −5.0% to −42.1%, resp.), consistent with other research results [1, 4, 35, 36]. However, toward the end of the flood season, or when the water depth reverted to the suitable growth water depth, the ability of the reeds to store carbon was gradually restored by summer. Obviously, depth increases due to artificial water transfer in spring which should be controlled to within 0.5 m (preferably to within 0.3 m).

Figure 7(b) shows that artificial water transfer in summer also caused a reduction of reed carbon stock (the largest reduction in aboveground and underground carbon stock was from −6.2% to −50.0% and from −6.2% to −49.9%, resp.). However, toward the end of the flood season, or when the water depth reverted to the suitable growth water depth, the ability of the reeds to store carbon was gradually restored by the beginning of fall. However, the degree of flooding dictates whether it can return to the value. Thus, depth increases due to artificial water transfer in summer which should be controlled to within 0.5 m (preferably to within 0.3 m).

Figure 7(c) shows that artificial water transfer in fall also caused a reduction in reed carbon stock until the end of season (the largest reduction in aboveground and underground carbon stock was from −6.2% to −50.0% and from −6.1% to −49.5%, resp.). However, subsequently, reed carbon stock was not restored. Again, the conclusion can be drawn that depth increases due to artificial water transfer in fall which should be controlled to within 0.5 m (preferably to within 0.3 m).

Based on the above simulation, Figure 7(d) illustrates the overall change trend of reed carbon stock in the growing season when the water depth fluctuated by 0.3 m. The results show that water depth fluctuations caused by artificial water transfer resulted in a decrease of carbon stock compared with the same period when water depth was regulated to the suitable growth water depth. Artificial water transfer in spring was relatively less; however, in the latter stage with the return of the water depth, the growth of the reeds might be gradually restored. Therefore, artificial water transfer is important for maintaining the suitable growth water depth in the Baiyangdian wetland, which enables reed growth and the increase of reed carbon stock.

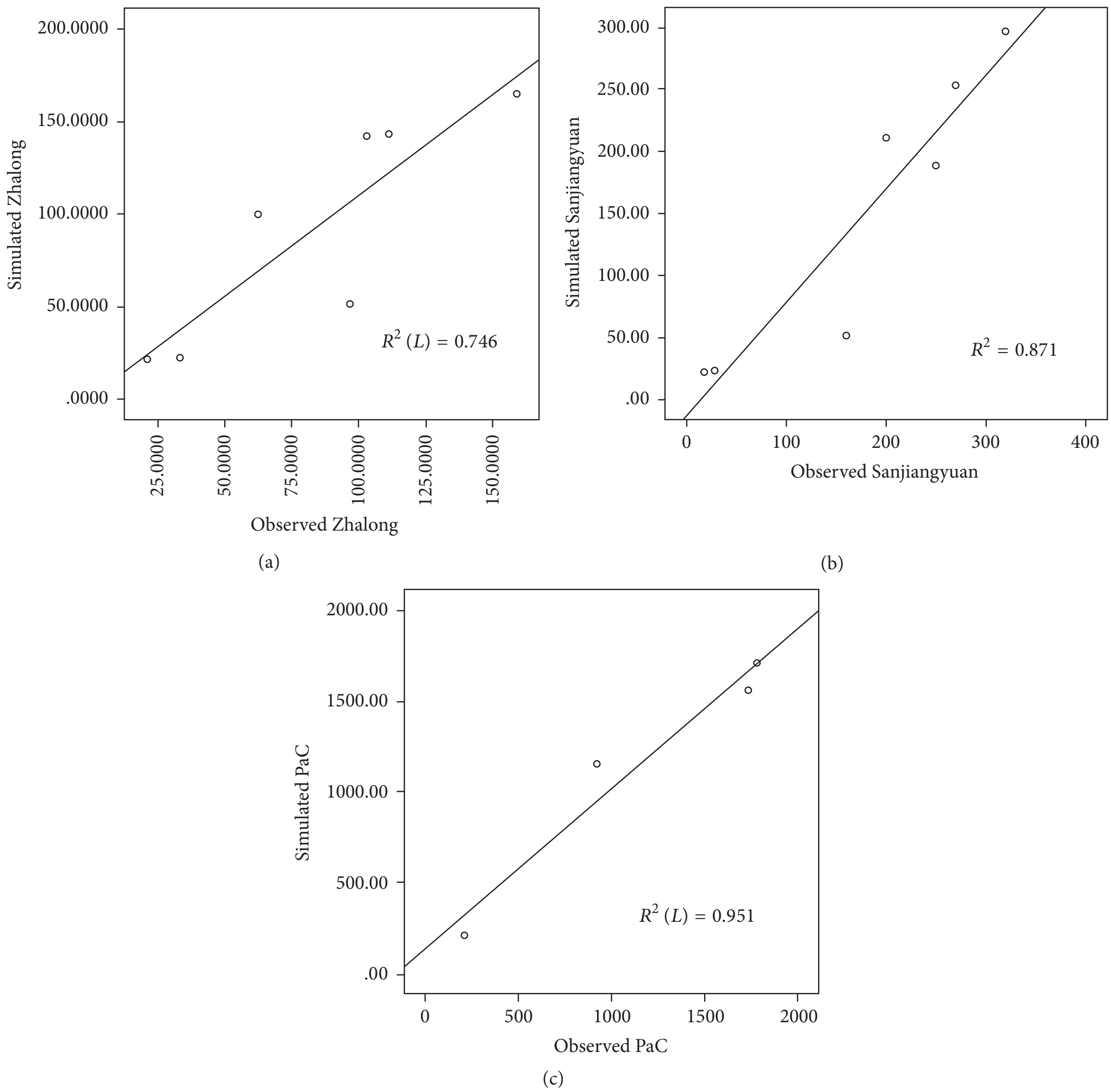


FIGURE 5: Comparison of observed and simulated aboveground carbon stock: (a) Zhalong wetland in 2010, (b) Sanjiang Plain wetland in 2012, and (c) Baiyangdian wetland in 2012.

5. Discussion

(1) Usually, the process of photosynthesis is expressed using the Michaelis-Menten equation [2, 3, 12–14]. Based on previous research, this study introduced into the Michaelis-Menten equation a relation between the environmental carrying capacity of reeds and water depth, as well as the concept of suitable growth water depth. A model of the relationship between the carbon stock capacity of reeds and water depth fluctuation was constructed, and simulations of the Baiyangdian wetland were performed. The results showed good agreement between the predicted and observed values, reflecting the variation of reed carbon stock with water depth. Thus, this model provides a new approach to the simulation of the growth of water plants in different water depths.

(2) This study established that the influence of water depth fluctuation on the reed carbon stock is seasonal (spring, summer, and fall); however, a general decrease of the carbon stock was found. Furthermore, it was revealed that water flooding affects the photosynthesis process, which inhibits/delays reed growth [4, 33–35]. Similarly, in summer and fall, water diversion was found to cause a decrease in reed carbon stock, confirming the results of many other studies [21, 36, 37]. The results of this study showed that an appropriate volume of artificial water transfer is required to maintain the suitable growth water

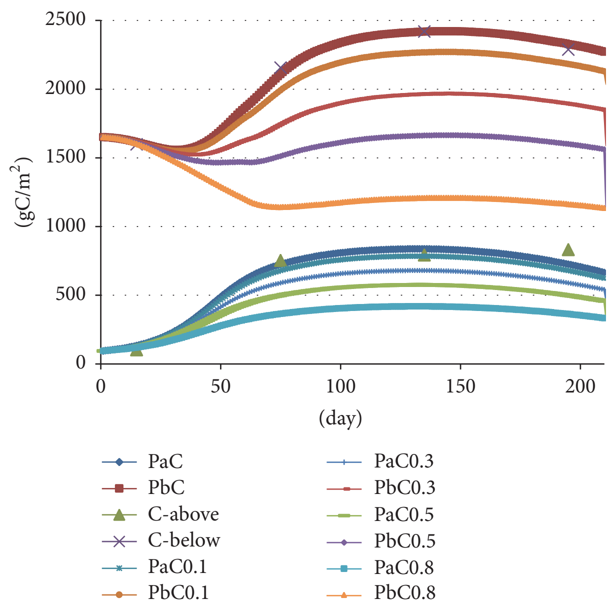


Figure 6: Change trend of carbon stock in aboveground/underground tissue of reeds during the entire growing season.

depth, which is conducive to reed growth and to the increase of carbon stock.

(3) Based on the findings of this study, in order to maintain or increase the carbon stock of the Baiyangdian wetland reed, it is suggested that spring is the optimum time for artificial water transfer. Fall is not an appropriate season for water transfer, and if conducted in summer, it should be performed late in the season (i.e., the end of August). The range of water depth fluctuation due to artificial water regulation should be limited to within 0.3 m.

(4) The growth and carbon cycles of wetland plants are extremely complex and they are influenced by many factors. In this study, the most important hydrological conditions were selected, which also constitute those factors most vulnerable to human control and influence. Thus, improved understanding of these factors is most effective for the restoration of damaged wetland ecosystem service function. This study focused on the artificial regulation of water, within the context of seasonal fluctuations of water depth and the carbon stock capacity of the Baiyangdian wetland, to draw the relevant conclusions; however, the effects of spatiotemporal scale and environmental and human factors should be considered in future research.

6. Conclusions

This study considered the influence of seasonal fluctuations of water depth and artificial water transfer on the carbon stock of the reeds in the Baiyangdian wetland. Based on this analysis, a relational model was established, which was used to simulate the effects of the artificial regulation of water on the growth of reed carbon stock. It is of considerable importance to estimate accurately the cycle and flux of carbon under the influence of artificial control. It was demonstrated that the model could be used in the design of the ecological service function of damaged wetlands and that it could be applied to other similar wetland environments.

There are three main conclusions to this research. (1) For the first time, a relational function between the environmental carrying capacity of reeds and water depth, as well as the concept of suitable growth water depth, has been introduced into the Michaelis-Menten equation. Thus, this provides a new approach to the simulation of the growth of water plants in different water depths. (2) The results showed that an appropriate volume of artificial water transfer is necessary to maintain the suitable growth water depth, enabling reed growth and the increase of carbon stock. (3) The optimum time for artificial water transfer in the Baiyangdian wetland is spring. The range of water depth fluctuation due to artificial water regulation should be limited to within 0.3 m.

Abbreviations

Summary of Symbols Used in the STELLA Model

PaC: Aboveground carbon stocks of the reed (gCm^{-2})
PbC: Underground carbon stocks of reed (gCm^{-2})
Photosynthesis: Photosynthesis (gCm^{-2} per day)
decayPa: Litter carbon stocks in aboveground tissues of the reed (gCm^{-2} per day)
mortalPa: Aboveground tissue death carbon stocks (gCm^{-2} per day)

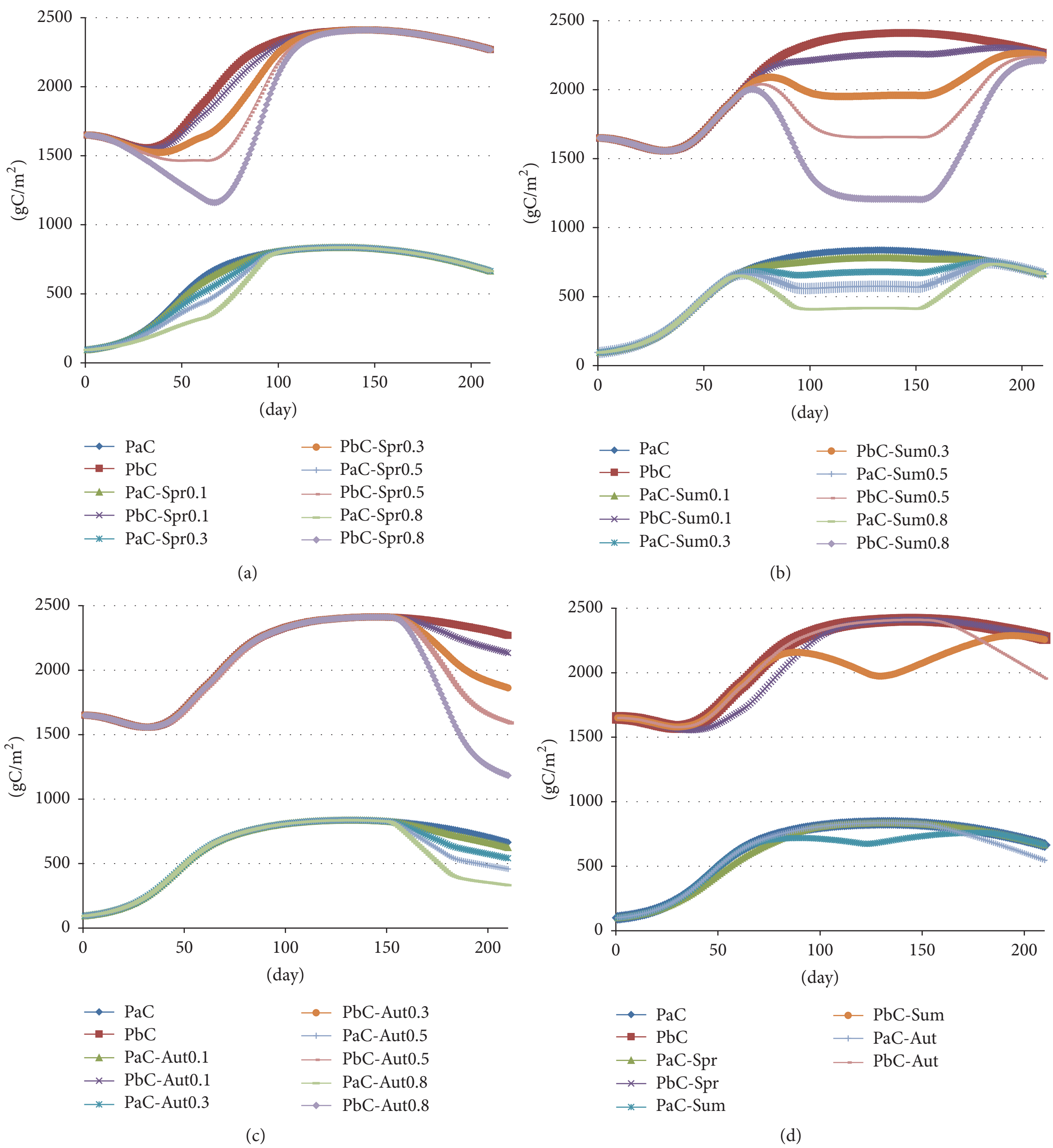


FIGURE 7: Change trend of carbon stock in aboveground/underground plant tissue: (a) spring, (b) summer, (c) fall, and (d) entire growing season.

mortalPb: Underground tissue death carbon stocks (gCm^{-2} per day)

respirePa: Aboveground tissue respiration of reed (gCm^{-2} per day)

respirePb: Underground tissue respiration (gCm^{-2} per day)

REACT: Remobilization (gCm^{-2} per day)

transfer: Translocation (gCm^{-2} per day)

Air temperature: Air temperature (°C)

Concen CO_2: Air carbon dioxide concentration (ppm per year)

CarryaC: Maximum environmental carrying capacity of reed (gCm^{-2})

Rad: Solar radiation ($MJ\,m^{-2}$ per year)

(Suitable) water depth: Suitable water depth/average depth (m)

waterdepthmax: Maximum water depth (m)

decayratePa: Tissue litter rate on the ground of reed (gg^{-1} per day)
mortalratePa: Death rate of the aboveground reed (gg^{-1} per day)
mortalratePb: Death rate of the underground reed (gg^{-1} per day)
respirratioPa: Respiration rate of the aboveground reed (gg^{-1} per day)
respirratioPb: Respiration rate of the underground reed (gg^{-1} per day)
reratio: Remobilization rate (gg^{-1} per day)
transratio: Translocation rate (gg^{-1} per day).

Competing Interests

The authors declare that they have no competing interests.

Acknowledgments

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References

[1] A. E. Altor and W. J. Mitsch, "Methane and carbon dioxide dynamics in wetland mesocosms: effects of hydrology and soils," *Ecological Applications*, vol. 18, no. 5, pp. 1307–1320, 2008.

[2] E. M. Eid, K. H. Shaltout, and T. Asaeda, "Modeling growth dynamics of *Typha domingensis* (Pers.) Poir. ex Steud. in Lake Burullus, Egypt," *Ecological Modelling*, vol. 243, pp. 63–72, 2012.

[3] J. Zhang, S. E. Jørgensen, J. J. Lu, S. N. Nielsen, and Q. Wang, "A model for the contribution of macrophyte-derived organic carbon in harvested tidal freshwater marshes to surrounding estuarine and oceanic ecosystems and its response to global warming," *Ecological Modelling*, vol. 294, no. 1, pp. 105–116, 2014.

[4] C. M. Li, X. Q. Ye, M. Wu, and X. X. Shao, "Effects of water depth and coexistence on growth characteristics of *Phragmitesaustralis* and *Typhadomingensis*," *Wetland Science*, vol. 13, no. 5, pp. 609–615, 2015.

[5] T. R. Moore, J. L. Bubier, S. E. Frolking, P. M. Lafleur, and N. T. Roulet, "Plant biomass and production and CO_2 exchange in an ombrotrophic bog," *Journal of Ecology*, vol. 90, no. 1, pp. 25–36, 2002.

[6] D. J. Cooper, J. S. Sanderson, D. I. Stannard, and D. P. Groeneveld, "Effects of long-term water table drawdown on evapotranspiration and vegetation in an arid region phreatophyte community," *Journal of Hydrology*, vol. 325, no. 1-4, pp. 21–34, 2006.

[7] H. W. Qin, Y. F. Liu, Z. X. Liu et al., "On effects of simulate submerged test in the Three Gorges Reservoir hydro-fluctuation area on growth of 2 species of Herbs: Salix variegate and Ficus tikoua," *Journal of Southwest China Normal University*, vol. 37, no. 10, pp. 426–434, 2012.

[8] W. Yang, Z. Yang, and Y. Qin, "An optimization approach for sustainable release of e-flows for lake restoration and preservation: model development and a case study of Baiyangdian Lake, China," *Ecological Modelling*, vol. 222, no. 14, pp. 2448–2455, 2011.

[9] H. C. J. Wang and Z. Jin, "Influence of water level gradient on plant growth, reproduction and biomass of plant species," *Acta Phytoecologica Sinica*, no. 3, pp. 269–274, 1999.

[10] X. Bai, K. Chen, K. Ren, W. Huang, X. A. Chen, and H. Yang, "Physiological response of Phragmites communis seedling to two types of sediment under submergence and its recovering growth," *Journal of Lake Sciences*, vol. 24, no. 4, pp. 562–570, 2012.

[11] M. J. Saunders, M. B. Jones, and F. Kansiime, "Carbon and water cycles in tropical papyrus wetlands," *Wetlands Ecology and Management*, vol. 15, no. 6, pp. 489–498, 2007.

[12] J. Z. Coletti, C. Hinz, R. Vogwill, and M. R. Hipsey, "Hydrological controls on carbon metabolism in wetlands," *Ecological Modelling*, vol. 249, pp. 3–18, 2013.

[13] M. Michela, R. M. Pulselli, N. Marchettini, F. M. Pulselli, and S. Bastianoni, "Carbon dioxide sequestration model of a vertical greenery system," *Ecological Modelling*, vol. 306, pp. 46–56, 2015.

[14] S. E. Jørgensen and S. N. Nielsen, "A carbon cycling model developed for the renewable Energy Danish Island, Samsø," *Ecological Modelling*, vol. 306, pp. 106–120, 2015.

[15] Anxin County Annals, *Compilation Committee of Local Chronicles of Anxin County (CCLCAC)*, Xinhua Publishing House, Beijing, China, 2000.

[16] B. Cui, X. Li, and K. Zhang, "Classification of hydrological conditions to assess water allocation schemes for Lake Baiyangdian in North China," *Journal of Hydrology*, vol. 385, no. 1, pp. 247–256, 2010.

[17] X. Zhao, B. Cui, and Z. F. Yang, "A study of the lowest ecological water level of Baiyangdian lake," *Acta Ecologica Sinica*, vol. 25, no. 5, pp. 1033–1041, 2005.

[18] Z. X. Zhao, *Coupling Mechanism of Eco-Hydrological Processes and Integrated Regulation in Baiyangdian Wetland*, Tianjin University, Tianjin, China, 2012.

[19] B. Li, S. Liu, Y. Niu et al., "Altered expression of the TCR signaling related genes CD3 and FcεRIγ in patients with aplastic anemia," *Journal of Hematology and Oncology*, vol. 5, article 6, 2012.

[20] A. J. Guo, *Photosynthetic Characteristics of Phragmitesaustralis and Wetland Soil CO_2 Flux in Baiyangdian Lake*, Hebei University, Baoding, China, 2013.

[21] P. W. Huang, *Study on CO_2 and CH_4 Fluxes from Zhalong Wetland*, Northeast Forestry University, Harbin, China, 2010.

[22] C. C. Hou, *Effects of Hydrological Changes on Soil Carbon Sequestration of Marsh in the Sanjiang Plain*, Graduate School of Chinese Academy of Sciences, 2012.

[23] K. L. Webster, J. W. McLaughlin, Y. Kim, M. S. Packalen, and C. S. Li, "Modelling carbon dynamics and response to environmental change along a boreal fen nutrient gradient," *Ecological Modelling*, vol. 248, no. 1751, pp. 148–164, 2013.

[24] K. Soetaert, M. Hoffmann, P. Meire et al., "Modeling growth and carbon allocation in two reed beds (Phragmites australis) in the Scheldt estuary," *Aquatic Botany*, vol. 79, no. 3, pp. 211–234, 2004.

[25] S. E. Jørgensen and B. D. Fath, "Applications in environmental management and research," *Developments in Environmental Modelling*, vol. 23, pp. 1–413, 2011.

[26] D. P. Aikman, "Potential increase in photosynthetic efficiency from the redistribution of solar radiation in a crop," *Journal of Experimental Botany*, vol. 40, no. 8, pp. 855–864, 1989.

[27] J.-R. Zhang, H.-Y. Zhang, and J.-J. Lu, "Comparisons of monthly biomass dynamics between aboveground and belowground parts of Pkragmites australis in freshwater tidal flat in Yangtze Estuary," *Wetland Science*, vol. 11, no. 1, pp. 7–12, 2013.

[28] L. Bo, C. Q. Liu, P. B. Yan, J. X. Wang, Y. J. Zhang, and Z. F. Tian, "Linear analysis of reed growth parameters in Baiyangdian Lake," *Hubei Agricultural Sciences*, vol. 49, no. 2, pp. 316–318, 2010.

[29] T. Asaeda and S. Karunaratne, "Dynamic modeling of the growth of Phragmites australis: model description," *Aquatic Botany*, vol. 67, no. 4, pp. 301–318, 2000.

[30] S. E. Jørgensen, S. N. Nielsen, and S. N. Ecotox, *Ecological Modelling and Ecotoxicology*, Elsevier Science, 2000.

[31] T. Asaeda, L. H. Nam, P. Hietz, N. Tanaka, and S. Karunaratne, "Seasonal fluctuations in live and dead biomass of *Phragmites australis* as described by a growth and decomposition model: implications of duration of aerobic conditions for litter mineralization and sedimentation," *Aquatic Botany*, vol. 73, no. 3, pp. 223–239, 2002.

[32] M. Anna and J. J. Lu, "Net ecosystem exchange of carbon and tidal effects in Chongxi Wetland, Yangtze Estuary," *Research of Environmental Sciences*, vol. 24, no. 7, pp. 716–721, 2011.

[33] Y. Li, C. Zhang, Y. Xie, and F. Liu, "Germination of *Deyeuxia angustifolia* as affected by soil type, burial depth, water depth and oxygen level," *Mitigation and Adaptation Strategies for Global Change*, vol. 14, no. 6, pp. 537–545, 2009.

[34] J. Yu, X. Wang, K. Ning et al., "Effects of salinity and water depth on germination of phragmites australis in coastal wetland of the yellow river delta," *Clean—Soil, Air, Water*, vol. 40, no. 10, pp. 1154–1158, 2012.

[35] B. Liu, L. Xian-Guo, M. Jiang, W. G. Zhang, and W. U. Hai-Tao, "Effects of light and water depth on seed germination of *Phragmites australis* in the wetlands of Songnen Plain," *Chinese Journal of Plant Ecology*, vol. 39, no. 6, pp. 616–620, 2015.

[36] M. Hirota, Y. Tang, Q. Hu et al., "Carbon dioxide dynamics and controls in a deep-water wetland on the Qinghai-Tibetan Plateau," *Ecosystems*, vol. 9, no. 4, pp. 673–688, 2006.

[37] X. Xu, Q. Zhang, Y. Li, Z. Tan, and X. Wang, "Reed population features and relationships between feature indicators and flooding depth, groundwater depth in an islet in Poyang Lake," *Wetland Science*, vol. 12, no. 6, pp. 714–722, 2014.

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Research Article

Phragmites australis + *Typha latifolia* Community Enhanced the Enrichment of Nitrogen and Phosphorus in the Soil of Qin Lake Wetland

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Aquatic plants play an essential role and are effective in mitigating lake eutrophication by forming complex plant-soil system and retaining total nitrogen (TN) and phosphorus (TP) in soils to ultimately reduce their quantities in aquatic systems. Two main vegetation types (*Phragmites australis* community and *P. australis* + *Typha latifolia* community) of Qin Lake wetland were sampled in this study for the analysis of TN and TP contents and reserves in the wetland soils. The results showed that (1) the consumption effect of Qin Lake wetland on soluble N was much more significant than on soluble P. (2) The efficiency of TN enrichment in wetland soil was enhanced by vegetation covering of *P. australis* and *T. latifolia*. (3) Wetland soil P was consumed by *P. australis* community and this pattern was relieved with the introduction of *T. latifolia*. (4) According to the grey relativity analysis, the most intensive interaction between plants and soil occurred in summer. In addition, the exchange of N in soil-vegetation system primarily occurred in the 0–15 cm soil layer. Our results indicated that vegetation covering was essential to the enrichment of TN and TP, referring to the biology-related fixation in the wetland soil.

1. Introduction

Eutrophication has now been an increasing problem in many countries. The Water Wheel reports in its recent issue that 54% of the lakes/reservoirs in Asia are impaired by eutrophication [1]. Comparatively, the percentage reaches 66% in China [2]. Total nitrogen (TN) and total phosphorus (TP) are the two major plant nutrients involved in the process of eutrophication [3, 4]. Extensive studies at the site scale have indicated that aquatic plants are effective in mitigating lake eutrophication, and many initiatives have been undertaken to improve water quality, such as introducing aquatic plants in the ecosystem and restoring or even creating wetland for this particular purpose [5]. The aquatic plants are able to form complex system with soils, where TN, TP, and some heavy metal elements cycle through various processes, such as litter decomposition, root absorption, and acid excretion, and ultimately retain N and P in soils to reduce their quantities in aquatic systems [6]. While various aquatic plants species have been adopted to treat eutrophication [7], *Phragmites australis* and *Typha latifolia* are the most commonly used aquatic plants in wetlands for the enhancement of water quality [8]. They have relatively high nutrients enrichment efficiencies compared to other plants, such as yellow flag (*Iris pseudacorus* L.) [7], contributing to effective nutrients removal from water in wetlands. However, their enrichment processes and the mechanisms are still unclear.

P. australis substantially improves the TN and TP removal efficiency in wetland ecosystem, due to its high growth rate and great capacity for nutrient accumulation in its stem, roots, and rhizomes [8]. It has a shallow-root system that often results in the nutrients enrichment occurring in surface soils through roots and rhizomes absorptions [9–11]. In addition, *P. australis* root biomass showed a positive correlation with the N content in aquatic system [12]. Add a few sentences of describing *P. australis* and *T. latifolia* and their abilities to enrich nutrients and their functions/mechanisms to remove N and P from water. By contrast, the enrichment efficiency of P is higher than that of N in *T. latifolia* [13]. It has a highly developed fine root system by which it can absorb the TN and TP in deep soil solutions and effectively increases the nutrient retention efficiency in wetland ecosystem [14]. Interestingly, *P. australis* and *T. latifolia* exhibit different tolerance to TN and TP deficiency, where *P. australis* is more tolerant to P limitation, while *T. latifolia* to N limitation [15, 16]. While a fair amount is known about the TN and TP retention efficiencies of *P. australis* and *T. latifolia* community, the nutrient retention efficiency of the mixed communities, such as *P. australis* and *T. latifolia*, is still unknown.

Qin Lake National Wetland Park is the second national wetland park in China approved by the State Forestry Administration [17]. In recent years, the water quality of the Qin Lake Wetland is dramatically decreasing due to the wastewater discharge, various pollutions, and dredged material disposal. The water N and P content has been increased by 4- and 17-fold, respectively, since the year from 1989 to 2010, which resulted in severe eutrophication [18]. Understanding the roles of the various aquatic plants in Qin Lake wetland, as well as their mechanisms by which they retain nutrients and purify water, provides theoretical support to the wetland management and the policy-making process. Here we select two typical riverfront communities (*P. australis* community, *P. australis* + *T. latifolia* community) in Qin Lake wetland to study the TN and TP stoichiometry in plant organs and the total N and P content in wetland soils and waters, as well as their seasonal variations, and to clarify how the two species differ in their contribution to wetland TN and TP enrichment, in order to improve understanding of the soil-vegetation nutrient cycle in wetland ecosystem and help aquatic plants management for Qin Lake.

2. Materials and Methods

2.1. Study Site. Qin Lake National Wetland Park is located in the middle of Jiangsu Province, Taizhou City, China (120°5′29.90″E~120°6′14.70″E, 32°37′2.70″N~32°37′33.70″N), 1.4 km in width (east-west) and 1.5 km in length (south-north), with an area of about 233.3 ha around. It has a humid subtropical climate with mild temperature and four distinct seasons.

Mean annual temperature is 16°C, respectively, 3.3°C in winter and 26.2°C in summer. Mean annual precipitation and relative humidity are 1031.8 mm and 80%. The average frost-free season is 220 days per year, and the annual leading wind directions are southeast wind. Present vegetation consists mainly of *P. australis*, *T. latifolia*, and so on. *P. australis* and *T. latifolia* community are the dominant plants, widely distributed in this area.

2.2. Experimental Design and Field Sampling. Two undisturbed sampling sites (A: *P. australis* dominated site and B: *P. australis* + *T. latifolia* dominated site) within the Qin Lake wetland were selected. 20×20 m control plot was established in both sites, where all the plants were removed once a month to prevent plant growth. Nine well-grown *P. australis* individuals were randomly collected in the two sites (outside the control plots) twice a season during February 2012–February 2013. Meanwhile, three additional plots (1×1 m) were selected in each site for harvesting the aboveground plant biomass at the end of growing season (late October), where plant stems, leaves, and spikes were collected separately in paper bags for analysing of plant biomass allocation. Roots were excavated and kept in polyethylene zip-top bags after removing dead roots and washing at the sampling sites. All plant materials were labeled and sent to lab for further analysis. Soil samples were collected from 0–15 cm, 15–30 cm, 30–45 cm, and 45–60 cm layers using multipoint mixing method. Flowing-water samples were collected at the locations 200 m up- and downstream of the channel in both sites. A transect was set up at each location and divided equally into 6 sections. Water sample at the 20 cm below the flow surface was collected for each section, separately kept in polyethylene bottle on ice, sealed tightly, and sent for lab analysis.

2.3. Determination of TN and TP in Plants, Soil, and Water. All plant samples were divided into roots, stems, leaves, and spikes, oven dried at 105°C for 15 min, and followed by 65°C for 24 hours to constant weight to measure the biomass. Dried plant samples were crushed, screened to a maximum particle size of 0.25 mm, and digested in H_2SO_4-H_2O_2 solution. TN was measured using the kjeldahl analysis methods and TP by the colorimetric molybdenum blue methods [19, 20]. Plant TN or TP storage (g/m^2) was estimated by multiplying the biomass of each plant organs (g/m^2) by total N or total P content (g/kg). Soil samples were air dried at room temperature, ground, and screened to a maximum particle size of 0.15 mm. TN and TP contents were measured, respectively, by the kjeldahl analysis methods and the colorimetric molybdenum blue methods. TN and TP of water samples were determined by ultraviolet spectrophotometric methods and spectrophotometric molybdate methods [21].

2.4. Data Analysis. TN and TP Concentrations were presented as mean values of at least three replicates. One-way analysis of variance (ANOVA) was performed using SPSS 19.0, and multiple comparisons were made by Dunnett's tests at a significant level of 0.05.

Since *P. australis* is the dominant species in Qin Lake wetland, a Grey Correlation Analysis was performed to study the relationship between TN and TP contents in the *P. australis* organs and in the soil [22].

3. Result

3.1. The N and P Contents in Upstream and Downstream Waters. TN and TP contents were both low in the downstream water relative to the upstream water ($P < 0.05$, Table 1).

Table 1: Annual average concentrations of N and P in the water of upstream and downstream of experimental sites ($n = 12$).

TN (mg/L)			TP (mg/L)		
Upstream	Downstream	P value	Upstream	Downstream	P value
1.26 ± 0.20	0.79 ± 0.08	0.05	0.11 ± 0.01	0.08 ± 0.003	0.02

The reduction rate of TN content (37.3%) in the downstream water was significantly lower than that of P content (27.3%).

3.2. TN and TP Contents in Soils. In contrast to *P. australis* community, *P. australis* + *T. latifolia* community was higher in TN content in all soil layers ($P < 0.05$), while the TN content of *P. australis* community was higher than that of the control sites with plants harvested ($0.05 < P < 0.08$, Figure 1). The soil total P content was lowest in *P. australis* community ($P < 0.01$). No significant difference in total P content was observed between the soils in the mixed communities and the control sites ($P > 0.05$), except the 0–15 cm layer showing a significantly low total P content compared to the control site ($P < 0.01$). However, the TP content of *P. australis* community was significant lower than that of the control sites ($P < 0.01$).

3.3. TN and TP Contents in Plant Organs. TN content was highest in *P. australis* leaves ($P < 0.01$), while little difference was observed between *P. australis* communities in sites A and B ($P > 0.05$, Figure 2). The total N content of *T. latifolia* leaves was lower than that of *P. australis* ($P < 0.05$), whereas other organs did not show significant difference in the total N content between the two species ($P > 0.05$). The total P content in both *P. australis* and *T. latifolia* varied significantly with the plant organ (Figure 2). It is 1.8- to 3.4-fold higher in *T. latifolia* spikes than in other organs. Conversely, TP content in the *P. australis* leaves was significantly higher than that of the spikes in the same community, which is consistent with the profile of TN allocation. Interestingly, TP was significantly increased in *P. australis* spikes in the *P. australis* + *T. latifolia* community, showing little difference to the leaves ($P > 0.05$). The P content in *P. australis* spikes of the mixed community is 2.5-fold higher than that of the *P. australis* community ($P < 0.01$).

The allocation of the nitrogen and phosphorus storage across organs in *P. australis* showed the same profile as roots > stems > leaves > spikes, while *T. latifolia* in the mixed community showed roots > spikes > stems > leaves (Table 2). The N and P storages in *P. australis* roots were 66.7% and 71.4% of the total N and P storage, which is higher than that in *T. latifolia* roots. By contrast, the N and P storages in spikes were significantly higher in *T. latifolia* ($P < 0.05$). The N and P storages of *P. australis* in *P. australis* community were only 4.2% and 2.6% of the TN and TP storage. Nonetheless, TP storage of *P. australis* in *P. australis* + *T. latifolia* community was significantly increased, taking up to 7.2% of the total P storage. This is 2.8-fold higher than that in *P. australis* community. Due to the notable difference in plant biomass between *P. australis* and *T. latifolia*, the N and P storage were both high in *P. australis*. However, the high P content in

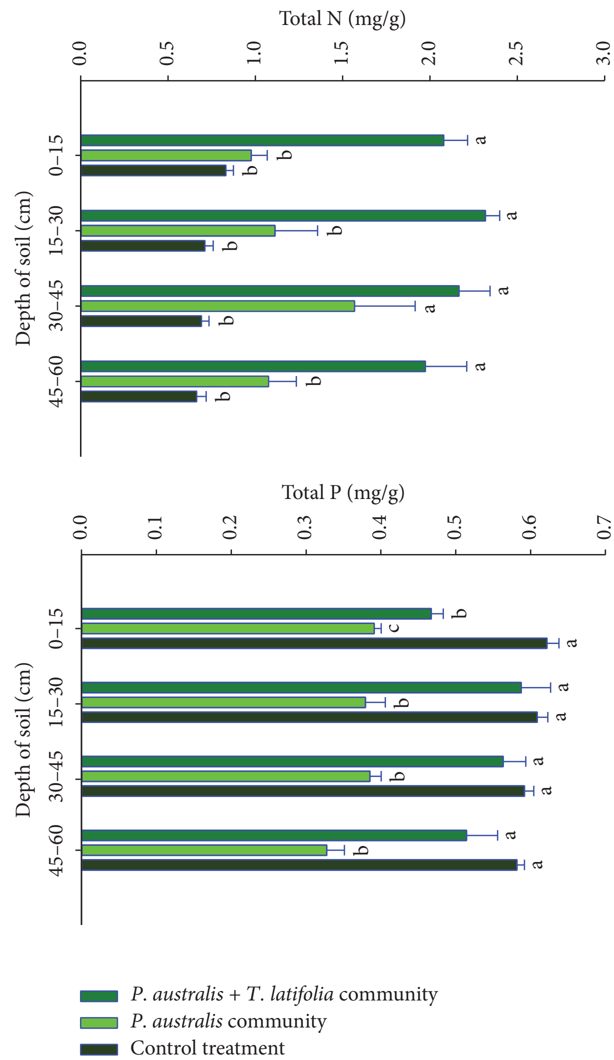

Figure 1: Concentrations of TN and TP in different depths of soil in experimental site. Different lowercase letters indicate significant difference among TN and TP contents of soil ($P < 0.05$). Values are mean + 1 sem.

T. latifolia spikes results in the total P storage in *T. latifolia* community 2-fold higher than that in *P. australis* community.

3.4. The Seasonal Variations of TN and TP Contents in Plant Organs. No data was collected on TN and TP content in *T. latifolia* spikes in late autumn and winter, during which the spikes disappear in response to the climate condition (Figure 3). The total N and P contents in *P. australis* stems and leaves were highest in summer, which is particularly true for

Table 2: Organs biomass and N and P reserves of *P. australis* and *T. latifolia* ($n = 8$).

	P. australis from plot A					*P. australis* from plot B					*T. latifolia* from Plot B				
	Root	Stem	Leaf	Spike	Total	Root	Stem	Leaf	Spike	Total	Root	Stem	Leaf	Spike	Total
Biomass (g/m^2)	4512.0 ± 1308.1	1047.9 ± 991.70	343.5 ± 305.5	269.2 ± 229.4	6172.6 ± 1819.3	3965.5 ± 1822.6	1123.4 ± 887.02	389.8 ± 223.7	305.8 ± 189.1	5784.5 ± 2230.3	436.1 ± 110.0	75.8 ± 32.4	59.60 ± 15.70	124.2 ± 42.00	603.80 ± 152.90
N reserves (g/m^2)	55.2	11.6	9.4	3.3	79.5	39.9	9.9	9.2	3.4	62.4	6.3	0.9	0.8	2.2	10.2
P reserves (g/m^2)	1.1	0.3	0.1	0.04	1.54	1.0	0.2	0.1	0.1	1.4	0.2	0.03	0.01	0.08	0.32

Plot A is *P. australis* community; plot B is *P. australis* + *T. latifolia* community.

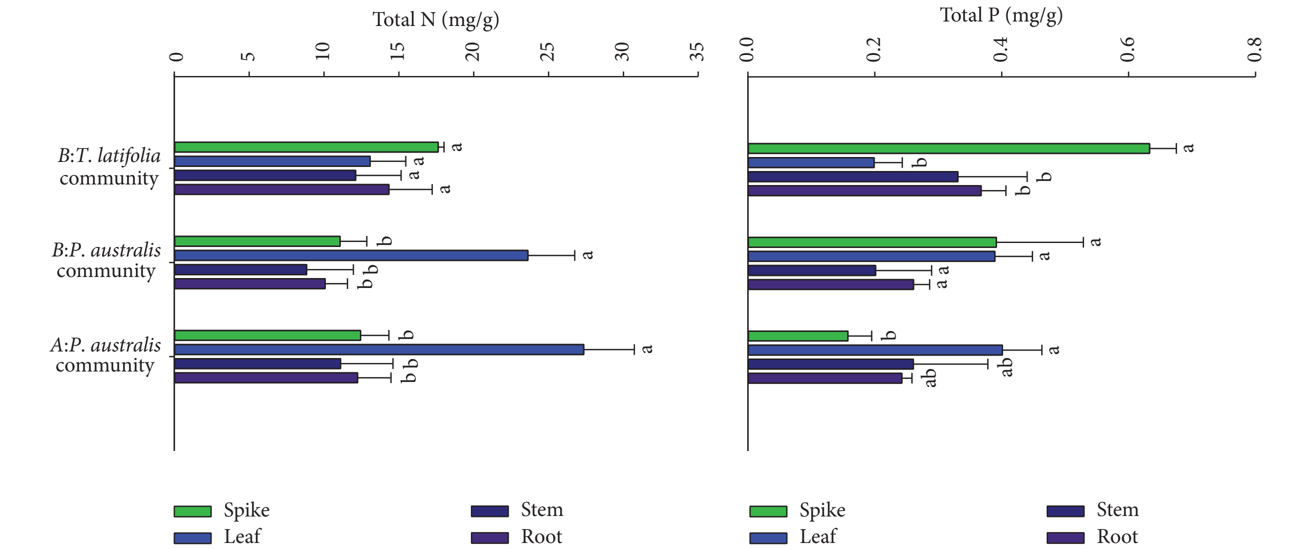

FIGURE 2: Concentration of TN and TP in the organs of *P. australis* and *T. latifolia* in experimental sites. Different lowercase letters indicate significant difference among TN and TP contents of organs ($P < 0.05$). Values are mean + 1 sem.

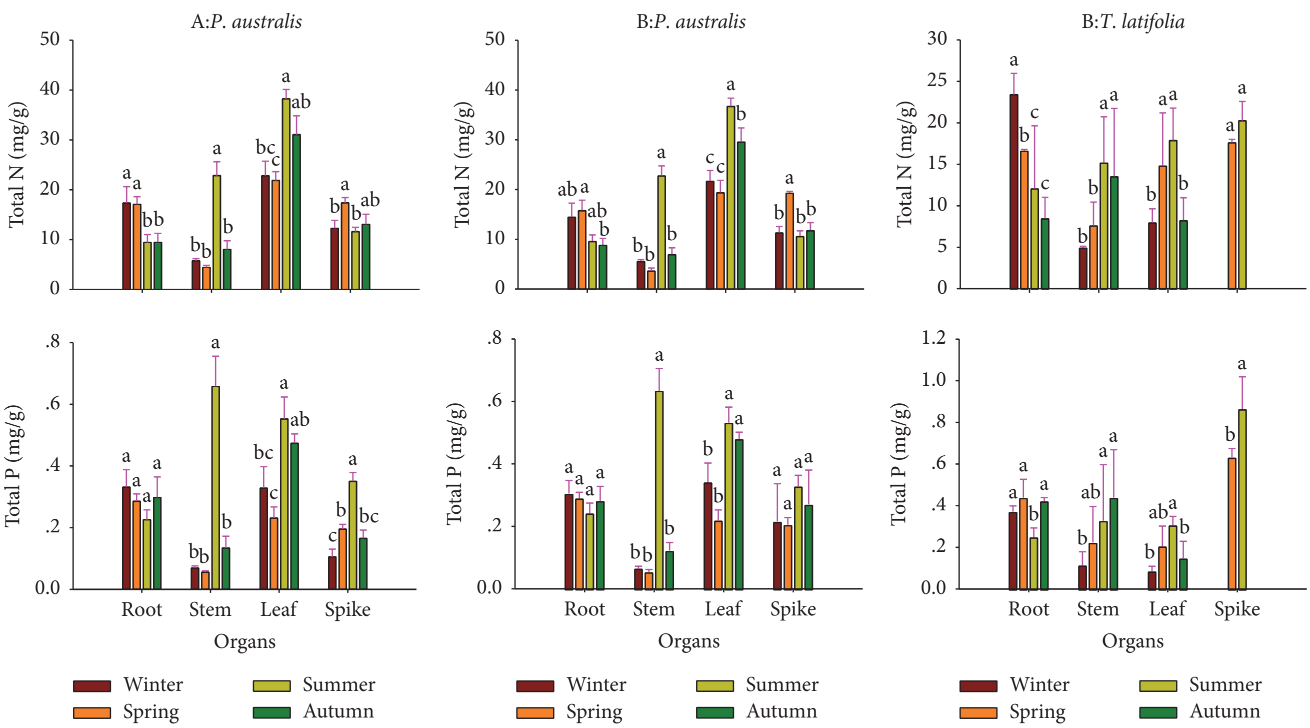

FIGURE 3: Seasonal dynamic concentration of TN and TP in the organs of *P. australis* and *T. latifolia* in experimental sites. The first and second columns of figure are the information of *P. australis* from sites A and B, and the third column is *T. latifolia* from site B. Different lowercase letters indicate significant difference among TN and TP contents of seasons ($P < 0.05$). Values are mean + 1 sem.

TN. TN content in *P. australis* stems was higher in summer than other seasons up to 70–84%, while the P content up to 81–92%. TP content in *P. australis* roots did not vary with the season, which was likely due to the stable biomass and TP demands throughout the seasons, whereas, the high TP content in *P. australis* stems was likely due to the vigorous growth in summer that needs more TP than usual to promote plant growth. Similarly, TN contents in *T. latifolia* stems and leaves varied with season, both highest in summer, when the TN content among organs showed a pattern of spikes > leaves > stems > roots, while the P content was spikes > stems > leaves > roots. The high TN and TP contents in *T. latifolia*

Table 3: Grey Correlation Analysis on contents of N and P between organs of *P. australis* and different depth of soil.

Depth of soil	Association index								Association index of N and P between the whole plant and different depth of soil	
	Winter		Spring		Summer		Autumn			
	N	P	N	P	N	P	N	P	N	P
0–15	0.30	0.40	0.30	0.54	0.51	0.72	0.25	0.25	0.71	0.65
≥15–30	0.35	0.40	0.25	0.45	0.45	0.66	0.35	0.25	0.68	0.66
≥30–45	0.43	0.36	0.25	0.45	0.52	0.64	0.33	0.25	0.67	0.65
≥45–60	0.41	0.33	0.26	0.62	0.41	0.70	0.25	0.25	0.64	0.65
Seasonal association index of N and P between the whole plant and soil	0.37	0.37	0.27	0.52	0.47	0.68	0.30	0.25		

spikes were likely due to the fast growth and development in summer, which demands more TN and TP supply, compared to other seasons.

3.5. The Grey Correlation Analysis between Soil and the Contents of TN and TP about P. australis Organs. The association index of N content in *P. australis* and soil in different depth has negative correlation with soil depth, while the largest value of association index about P is the soil layer ≥15–30 cm (Table 3). *P. australis* needs N in the process of growth of different organs and obtains it mainly from the shallow soil, whereas the nutrient elements in the deep soil layer play a relatively small role in the growth of *P. australis*. There is no significant difference in various soil layers with the association index of P content, slightly increasing only in the soil layer ≥15–30 cm. It is in summer that the relationship of N and P contents is best in *P. australis* and soil. The plants thrive in summer, and the soil layer of 0–15 cm has large impact on *P. australis* gaining the N and P in summer which is a period of strong growth of plants. At the same time, as a result of the underground water level that rises higher in rain season, medium and lower soil layer may have greatly impact the content of nutrient elements in plants. This phenomenon is also reflected in spring, but the tendency is not significant compared with summer.

4. Discussions

Wetland soils play an important role in improving water quality, nutrient accumulation, and regeneration of nitrogen and phosphorus, which largely depends on the plant community structure [23, 24]. The notable reductions of TN and TP content in the downstream water of the research site indicated that Qin Lake wetland is able to remove the water N and P and that the ability of investigated communities to retain TN (reduced by 37.3%) is high as compared to TP (reduced by 27.3%), while *P. australis* and *T. latifolia* are two dominating aquatic plants, both of which are regularly used in construction wetlands to remove N and P from aquatic systems due to their competence in nutrient enrichment [25–27]. The enhanced N retention in *P. australis* soils, especially in the 30–45 cm soil layer, was mainly attributed to the roots whose 43% were distributed in the below 30 cm soils. Furthermore, 91% of them were fine roots less than 2 mm in diameter, but with a massive surface area and fast turnover rate, which dramatically enhances the N enrichment in below 30 cm soils.

In addition, *P. australis* + *T. latifolia* community significantly increased the wetland soil TN content, demonstrating that plant community structure has significant impact on the wetland soil N retention (Figure 1). It was likely due to the mechanism of interspecific adaptations, where *P. australis* adopted stress tolerance mechanism, while *T. latifolia* adopted stress escape mechanism to accommodate interspecific competition as they coexisted in the same environment [28]. The adaptation was reflected by a series of changes mainly in morphological phenotypes, such as the elongation of *T. latifolia* root, decrease in stem diameter, decrease of the leaf numbers, and the increase of plant height [29]. The competition between *P. australis* and *T. latifolia* improved the plant nitrogen use and uptake efficiencies and promoted the biological N enrichment process, which ultimately contributed to the increase of soil total N through the plant-soil system. Contrarily, the wetland soil P content was significantly decreased in the *P. australis* community and maintained the same concentration in the *P. australis* + *T. latifolia* community (Figure 1), which was mainly attributed to the strategy of these two species referring to the low P condition in the environment. *P. australis* is able to grow in low P condition by adjusting the rhizospheric soil pH to solubilize insoluble P and making it available for plant uptake, which expedites soil P consumption and, on the long run, depletes P in the surface soils [30]. By contrast, *T. latifolia* growing in the P deficient condition relies on increasing its root biomass to acquire P from deep soil solution to support their growth without changing the soil chemical compositions [31], which ultimately retains P through plant-soil system and offsets the soil P consumption by *P. australis* + *T. latifolia* community. The impact of *T. latifolia* on soil P retention is enhanced with the increase in soil depth. The P reduction in the 0–15 cm soil layers in the mixed community was primarily due to the *T. latifolia* roots distributed mainly in the deep soils, which may result from the escape mechanisms adopted during the interspecific competition [32, 33], which makes it unable to offset *P. australis* P consumption in the surface soils.

It had been reported that spike, as a reproductive organ, had comparatively higher N and P content than other organs, because of the high level of mitochondria content. However, the N and P content in *P. australis* spikes were not significantly higher than that of other organs in our study. It is likely due to the Qin Lake wetland eutrophication that changed the ecological strategy of *P. australis* community, where they reduced the cost of reproduction but increased the plant growth input. Similarly, the stem N and P contents were also low in both *P. australis* and *T. latifolia*, which is reasonable because it is mainly used to transport water and nutrients, as well as to provide plants physical support [34]. However, the total N and P contents in stem and leaves were highest in both species. This can be explained by the highest efficiency of photosynthesis and evapotranspiration in summer, when proteins, nucleic acids, and chloroplast are synthesized and transported more efficiently in these organs than any other seasons [12]. On the other hand, a large quantity of P is necessary to meet the need of energy consumed in the photosynthesis and respiration [35].

N : P ratio has been known as an indicator of N or P limitation [36]. $N:P < 14$ suggests N limits plant growth, while P becomes the limiting factor when $N:P > 16$ [37]. The N : P ratios in plant organs in this study are significantly different from the global and national N : P ratios for terrestrial plants. It is noteworthy that the average N : P ratio in leaves was 68.9, which was remarkably higher than that of the global (13) and national (14) averages. High leaf N : P ratio is beneficial to the photosynthesis and further promotes plant growth [38, 39]. The high leaf N : P ratio in this study is mainly due to the fact that the N content (26.87 g/kg) in *P. australis* leaves is well above the average of terrestrial plant leaves in China (18.6 g/kg) and that the P content (0.39 g/kg and 0.19 g/kg) in *P. australis* and *T. latifolia* leaves is, conversely, well below the average of terrestrial plant leaves in China (1.21 g/kg) [40]. The whole evidence indicated that the Qin Lake eutrophication is likely due to the excess of N, because P, relative to N, is still in deficiency, implying that P is the limiting factor of the aquatic plant growth in this region. It also suggested that *P. australis* community not only failed to improve the P retention efficiency, but also expedited the soil P consumption.

Nutrients exchange and enrichment occurred mainly in the surface soils. The root biomass of *P. australis* is mainly distributed in the surface soil (0–30 cm), while its vertical distribution could reach the 60 cm of soil layer. The content of N and P was often higher in the surface soil than that in deep soil due to the fresh soil solution supply from eutrophic runoff [41] and higher concentration of N, P, and organic matters [42], which led to intensive sequestration of nutrient elements in soil-vegetation system [43]. Meanwhile, the utilization efficiency of plant roots for N and P in deep soil was depressed by the lack of oxygen caused by the long-term water saturation [44].

According to the seasonal dynamic of correlation between N and P contents in *P. australis* and soil, the N and P sequestration of the wetland soil benefited from the growth of aquatic plants [45], especially in summer when the plants were at the time of most vigorous growth. The N and P in the runoff would be absorbed by plants, and then feedback to the soil as fine root litters and root exudates [46]. Therefore, the N and P contents in *P. australis* and the surface soil with the densest distribution of roots had the highest correlation.

5. Conclusion

P. australis + *T. latifolia* community enhanced the efficiency of N enrichment in wetland soil and was the same to the P comparing to *P. australis* community. Nutrient enrichment efficiency varies with the season and soil depth. The highest N and P enrichment efficiencies of the plant communities occurred in summer in 0–15 cm soil layer, while the enrichment of P occurred uniformly in all vertical soil layer.

Competing Interests

The authors declare that they have no competing interests.

Acknowledgments

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References

[1] I. Chorus and J. Bartram, *Toxic Cyanobacteria in Water: A Guide to Their Public Health Consequences, Monitoring, and Management*, E & FN Spon, London, UK, 1999.

[2] X.-C. Jin and X.-Z. Hu, "Concept and tactic of clean water runoff generation mechanism restoration in lake watershed," *China Environmental Science*, vol. 30, no. 3, pp. 374–379, 2010.

[3] L. Bengtsson and R. W. Herschy, "Encyclopedia of lakes and reservoirs," *Monographiae Biologicae*, no. 53, pp. 10–26, 2012.

[4] E. Granéli, M. Weberg, and P. S. Salomon, "Harmful algal blooms of allelopathic microalgal species: the role of eutrophication," *Harmful Algae*, vol. 8, no. 1, pp. 94–102, 2008.

[5] S. Wu, P. Kuschk, H. Brix, J. Vymazal, and R. Dong, "Development of constructed wetlands in performance intensifications for wastewater treatment: a nitrogen and organic matter targeted review," *Water Research*, vol. 57, pp. 40–55, 2014.

[6] L. Lyubenova, A. J. Kuhn, A. Höltkemeier, and P. Schröder, "Root exudation pattern of *Typha latifolia* L. plants after copper exposure," *Plant and Soil*, vol. 370, no. 1-2, pp. 187–195, 2013.

[7] N. Korboulewsky, R. Wang, and V. Baldy, "Purification processes involved in sludge treatment by a vertical flow wetland system: focus on the role of the substrate and plants on N and P removal," *Bioresource Technology*, vol. 105, pp. 9–14, 2012.

[8] J. Vymazal, "Horizontal sub-surface flow and hybrid constructed wetlands systems for wastewater treatment," *Ecological Engineering*, vol. 25, no. 5, pp. 478–490, 2005.

[9] J. Vymazal, "Constructed wetlands for wastewater treatment: five decades of experience," *Environmental Science and Technology*, vol. 45, no. 1, pp. 61–69, 2011.

[10] S. G. Kang, Y. S. Kwak, and J. K. Hwangbo, "Management of slag dumping site by artificial floating vegetated island," *RIST Journal of R & D*, vol. 17, no. 2, pp. 131–135, 2003.

[11] R. K. Hubbard, "Floating vegetated mats for improving surface water quality," in *Emerging Environmental Technologies*, vol. 2, pp. 211–244, Springer, Dordrecht, Netherlands, 2010.

[12] L. Li, S. Zerbe, W. Han et al., "Nitrogen and phosphorus stoichiometry of common reed (*Phragmites australis*) and its relationship to nutrient availability in northern China," *Aquatic Botany*, vol. 112, pp. 84–90, 2014.

[13] S. Miao, "Rhizome growth and nutrient resorption: mechanisms underlying the replacement of two clonal species in Florida Everglades," *Aquatic Botany*, vol. 78, no. 1, pp. 55–66, 2004.

[14] L. M. Conley, R. I. Dick, and L. W. Lion, "An assessment of the root zone method of wastewater treatment," *Research Journal of the Water Pollution Control Federation*, vol. 63, no. 3, pp. 239–247, 1991.

[15] L. C. Batty and P. L. Younger, "Effects of external iron concentration upon seedling growth and uptake of fe and phosphate by the common reed, *Phragmites australis* (Cav.) Trin ex. Steudel," *Annals of Botany*, vol. 92, no. 6, pp. 801–806, 2003.

[16] H. Brix, K. Dyhr-Jensen, and B. Lorenzen, "Root-zone acidity and nitrogen source affects *Typha latifolia* L. growth and uptake kinetics of ammonium and nitrate," *Journal of Experimental Botany*, vol. 53, no. 379, pp. 2441–2450, 2002.

[17] Z. Z. Xu and J. Z. Chu, "In-depth development of wetland ecotourism—a case study of Qinhu Lake National Wetland Park," *Wetland Science & Management*, no. 3, pp. 22–25, 2013.

[18] J. M. Feng and H. R. Zhang, "Conservation and measures of Qin Lake wetland in Jiangyan," *Pollution Control Technology*, vol. 24, no. 1, pp. 23–24, 2011.

[19] J. M. Bremner and M. A. Tabatabai, "Use of an ammonia electrode for determination of ammonium in kjeldahl analysis of soils," *Communications in Soil Science & Plant Analysis*, vol. 3, no. 2, pp. 159–165, 1972.

[20] J. Murphy and J. P. Riley, "A modified single solution method for the determination of phosphate in natural waters," *Analytica Chimica Acta*, vol. 27, pp. 31–36, 1962.

[21] J. J. Ameel, R. P. Axler, and C. J. Owen, "Persulfate digestion for determination of total nitrogen and phosphorus in low nutrient water," *American Environmental Laboratory*, vol. 10, no. 93, pp. 1–11, 1993.

[22] Y.-L. Yeh and T.-C. Chen, "Application of grey correlation analysis for evaluating the artificial lake site in Pingtung Plain, Taiwan," *Canadian Journal of Civil Engineering*, vol. 31, no. 1, pp. 56–64, 2004.

[23] H. Brix, "Do macrophytes play a role in constructed treatment wetlands?" *Water Science and Technology*, vol. 35, no. 5, pp. 11–17, 1997.

[24] T. Gumbricht, "Nutrient removal processes in freshwater submersed macrophyte systems," *Ecological Engineering*, vol. 2, no. 1, pp. 1–30, 1993.

[25] R. M. Gersberg, B. V. Elkins, S. R. Lyon, and C. R. Goldman, "Role of aquatic plants in wastewater treatment by artificial wetlands," *Water Research*, vol. 20, no. 3, pp. 363–368, 1986.

[26] N. B. Chang, M. K. Islam, and M. P. Wanielista, "Floating wetland mesocosm assessment of nutrient removal to reduce ecotoxicity in stormwater ponds," *International Journal of Environmental Science and Technology*, vol. 9, no. 3, pp. 453–462, 2012.

[27] R. K. Hubbard, G. J. Gascho, and G. L. Newton, "Use of floating vegetation to remove nutrients from swine lagoon wastewater," *Transactions of the American Society of Agricultural Engineers*, vol. 47, no. 6, pp. 1963–1972, 2004.

[28] T. Y. Yeh, M. H. Wu, C. Y. Cheng et al., "A study and analysis on the physical shading effect of water quality control in constructed wetlands," *Journal of Civil Environmental Engineering*, vol. 4, no. 3, article no. 147, 2014.

[29] L. Squires and A. G. Van Der Valk, "Water-depth tolerances of the dominant emergent macrophytes of the Delta Marsh, Manitoba," *Canadian Journal of Botany*, vol. 70, no. 9, pp. 1860–1867, 1992.

[30] A. Debska, K. Jóźwiakowski, M. Gizińska-Górna et al., "The efficiency of pollution removal from domestic wastewater in constructed wetland systems with vertical flow with common reed and glyceria maxima," *Journal of Ecological Engineering*, vol. 16, no. 5, pp. 110–118, 2015.

[31] O. Q. Hendrickson, L. Chatarpaul, and D. Burgess, "Nutrient cycling following whole-tree and conventional harvest in northern mixed forest," *Canadian Journal of Forest Research*, vol. 19, no. 6, pp. 725–735, 1989.

[32] J. E. Weaver and W. J. Himmel, "Relation of increased water content and decreased aeration to root development in hydrophytes," *Plant Physiology*, vol. 5, no. 1, pp. 69–92, 1930.

[33] T. R. Headley, D. O. Huett, and L. Davison, "Seasonal variation in phosphorus removal processes within reed beds—mass balance investigations," *Water Science and Technology*, vol. 48, no. 5, pp. 59–66, 2003.

[34] Y. Xing, J. Bubier, T. Moore et al., "The fate of ^{15}N-nitrate in a northern peatland impacted by long term experimental nitrogen, phosphorus and potassium fertilization," *Biogeochemistry*, vol. 103, no. 1, pp. 281–296, 2011.

[35] N. Terry and A. Ulrich, "Effects of phosphorus deficiency on the photosynthesis and respiration of leaves of sugar beet," *Plant Physiology*, vol. 51, no. 1, pp. 43–47, 1973.

[36] C. A. Suttle and P. J. Harrison, "Ammonium and phosphate uptake rates, N: P supply ratios, and evidence for N and P limitation in some oligotrophic lakes," *Limnology and Oceanography*, vol. 33, no. 2, pp. 186–202, 1988.

[37] W. Koerselman and A. F. M. Meuleman, "The vegetation N:P ratio: a new tool to detect the nature of nutrient limitation," *Journal of Applied Ecology*, vol. 33, no. 6, pp. 1441–1450, 1996.

[38] P. B. Reich, J. Oleksyn, and I. J. Wright, "Leaf phosphorus influences the photosynthesis-nitrogen relation: a cross-biome analysis of 314 species," *Oecologia*, vol. 160, no. 2, pp. 207–212, 2009.

[39] P. B. Reich and J. Oleksyn, "Global patterns of plant leaf N and P in relation to temperature and latitude," *Proceedings of the National Academy of Sciences of the United States of America*, vol. 101, no. 30, pp. 11001–11006, 2004.

[40] W. Han, J. Fang, D. Guo, and Y. Zhang, "Leaf nitrogen and phosphorus stoichiometry across 753 terrestrial plant species in China," *New Phytologist*, vol. 168, no. 2, pp. 377–385, 2005.

[41] T. Saeed and G. Sun, "A review on nitrogen and organics removal mechanisms in subsurface flow constructed wetlands: dependency on environmental parameters, operating conditions and supporting media," *Journal of Environmental Management*, vol. 112, pp. 429–448, 2012.

[42] J. Bai, Q. Wang, W. Deng, H. Gao, W. Tao, and R. Xiao, "Spatial and seasonal distribution of nitrogen in marsh soils of a typical floodplain wetland in Northeast China," *Environmental Monitoring and Assessment*, vol. 184, no. 3, pp. 1253–1263, 2012.

[43] J. Norkko, D. C. Reed, K. Timmermann et al., "A welcome can of worms? Hypoxia mitigation by an invasive species," *Global Change Biology*, vol. 18, no. 2, pp. 422–434, 2012.

[44] H. Hauggaard-Nielsen, P. Ambus, and E. S. Jensen, "Temporal and spatial distribution of roots and competition for nitrogen in pea-barley intercrops—a field study employing 32p technique," *Plant and Soil*, vol. 236, no. 1, pp. 63–74, 2001.

[45] K. J. Juneau and C. S. Tarasoff, "The seasonality of survival and subsequent growth of common reed (*Phragmites australis*) rhizome fragments," *Invasive Plant Science and Management*, vol. 6, no. 1, pp. 79–86, 2013.

[46] K. Häggqvist and T. Lindholm, "Phytoplankton dynamics in a shallow lake dominated by common water milfoil," *Inland Waters*, vol. 2, no. 3, pp. 137–146, 2012.

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Research Article

A Carbon Cycle Model for the Social-Ecological Process in Coastal Wetland: A Case Study on Gouqi Island, East China

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Coastal wetlands offer many important ecosystem services both in natural and in social systems. How to simultaneously decrease the destructive effects flowing from human activities and maintaining the sustainability of regional wetland ecosystems are an important issue for coastal wetlands zones. We use carbon credits as the basis for regional sustainable developing policy-making. With the case of Gouqi Island, a typical coastal wetlands zone that locates in the East China Sea, a carbon cycle model was developed to illustrate the complex social-ecological processes. Carbon-related processes in natural ecosystem, primary industry, secondary industry, tertiary industry, and residents on the island were identified in the model. The model showed that 36780 tons of carbon is released to atmosphere with the form of CO_2, and 51240 tons of carbon is captured by the ecosystem in 2014 and the three major resources of carbon emission are transportation and tourism development and seawater desalination. Based on the carbon-related processes and carbon balance, we proposed suggestions on the sustainable development strategy of Gouqi Island as coastal wetlands zone.

1. Introduction

Wetlands are biologically diverse and productive transitional areas between land and water. By occupying zones of transition between terrestrial and marine ecosystems, coastal wetlands, including salt marshes, mangroves, intertidal mudflats, seagrass beds, and shallow subtidal habitats, are the interface of the coastal landscape [1]. Coastal wetlands such as mangroves, salt marshes, intertidal mudflats, and seagrass beds have been suggested to offer many important ecosystem services [2]. Being productive and often spatially diverse habitats, coastal wetlands fulfill important functions such as producing a large variety of food to consumers, providing habitats for flora and fauna including migratory birds, fish, turtles, and cetaceans [3–7], and helping to moderate water quality [8].

Moreover, coastal wetlands zones support a variety of economic activities, including fisheries, aquaculture, tourism, recreation, and transportation. In recent decades, many coastal areas have been heavily modified and intensively developed. Human activities such as waste dumping, land reclamation, aquaculture ponds, and dredging for navigational channels and marinas have resulted in the recent rapid loss of coastal wetland habitats [2, 5]. How to simultaneously decrease the destructive effects flowing from human activities and improve local economic development, thus maintaining the sustainability of regional wetland ecosystems, are an important issue for coastal wetlands zones.

Carbon credit [9] within the area is now considered an effective method during regional sustainability policies making. With carbon credits, we can calculate the carbon emission through human activities and the carbon uptake or removal by natural environmental system such as green plants, algae, and shellfish. Thus the state of "carbon-free" or "carbon neutral" can be considered as the goal for regional sustainable development [10, 11]. To achieve this goal, the carbon emission can be reduced by the introduction of innovative technology, and the carbon removal can be improved

Table 1: The basic information of Gouqi Island.

Gouqi Island	Value
Land area (km^2)	6.62
Sea area (km^2)	1,600
Population	10,470 (2009a)
GDP (million yuan RMB)	912 (2012a)
Primary industry (million yuan RMB)	301 (2012a)
Secondary industry (million yuan RMB)	198
Tertiary industry (million yuan RMB)	413
Forest coverage rate (%)	53
Tidal wetland area (km^2)	0.92
The number of tourists	210,400

by artificial ecological system that can improve or develop new ecosystem service function.

In this article we use carbon credits as the basis for regional sustainable developing policy-making, present the carbon cycle model of Gouqi Island, a typical coastal wetlands zone that locates in the East China Sea, to illustrate the relationship between human activities and natural environment in costal wetland through carbon credits, discuss the regional developing models and paths for Gouqi Island to achieve sustainable development, which can also be generalized to the sustainable development strategy of coastal wetlands zones.

According to the developing strategy of Gouqi Island, tourism has been regarded as one of the major industries in future, together with agriculture (mainly relies on mussel). The island has rich coastline resources; thus tourism activities have witnessed fast development in the past 5 years. Especially after a popular movie named "the continent" was released in 2014 in China, a majority of tourists have come to Gouqi Island to experience leisure in the natural small island. This provides great opportunity for the tourism development in the island. However, as integration of extremely fragile systems, the island is now in heavy demand of sustainable plans that can balance between tourism development, infrastructure building, and tourism activity development through the consideration of economy and environmental conservation through long-term sustainability and natural welfare.

2. Study Area and Methods

2.1. Study Area. Gouqi Island locates in 30°43′1″N and 122°46′3″E where it is in the northeast among Zhoushan Archipelago, it is part of East China Sea coast and Islands wetland ecological system, the map of Gouqi Island is displayed in Figure 1, and the detailed information is listed in Table 1 [12].

The major industries in Gouqi Island are marine fishing industry and marine aquaculture, the marine fishing industries are developed in the area around the island and distant ocean, and the fishing industries are mainly developed along the coast, among which mussel culturing industry contributes around 95% to the industry considering both quantity and economic income. Mussel processing industry is also the traditional and important industry in the island. Besides that, the fresh water in the island relies on the desalination industry; this makes another important industry in the island. The tourism industry in the island has grown fast in the past 5 years and now contributes 45.2% of the GDP for the island. Currently, Gouqi Island receives more than 200,000 tourists per year and some negative impacts have appeared [12]. The energy structure in the island is quite simple, the transportation in the island heavily relies on fossil fuel, and the fossil-fuel plant locating in Gouqi Island provided energy required by desalination and seafood processing industry. The residential electricity is generated by the thermal power plants locating in the Shengsi Island. Besides fishing products, there are no production manufacturers in the island; thus all the food and other materials are imported from outside of the island. Ferry is the only transportation between the island and outside, and transportation in the island relies on transports which use fossil fuels. Because of the narrow road, the main transports on the island are small vehicles, while large trucks or buses cannot be used.

2.2. Modelling Methods. Following the model theory proposed by Jørgensen et al. [16] we develop a carbon cycling model based on the socioecological system with STELLA® software; the data used in the STELLA model mainly come from the following: (1) basic information of Gouqi Island from statistics reports; (2) data collecting from the tourists on the island with a questionnaire; (3), coefficients or parameters of processes in the model are cited from literatures; (4) field survey.

3. Model Description

3.1. Conceptual Model. We develop a carbon cycle model of Gouqi Island based on the socioecological process that determines the emission of CO_2 through human activity and economic development and removals of CO_2 through natural environment process. Jørgensen and Nielsen [17] have developed a carbon cycle model for the Danish island of Samsø to analyze the environmental management policies based on carbon emission and uptake process. We conduct similar procedures to develop a carbon cycle model for Gouqi Island (Figure 2). The main carbon pools, as well as important processes that show the flow of carbon from one pool to another and all the external inputs and outputs of carbon to the island, are reflected in the model. The model includes nature ecosystem (including forest ecosystem and tidal wetland ecosystem), residents (including transportation, electricity, and solid waste), primary industry (aquaculture of mussel), secondary industry (including seafood processing and seawater desalination), and tertiary industry (tourism activity, accommodation, and transportation) [18], and there are some overlaps between residents, secondary industry, and tertiary industry. The main inputs of carbon to the island are the imported food (for residents and tourists), electricity (for residents and tourists use), and fossil fuel. The conceptual model in the STELLA format is shown in Figure 3.

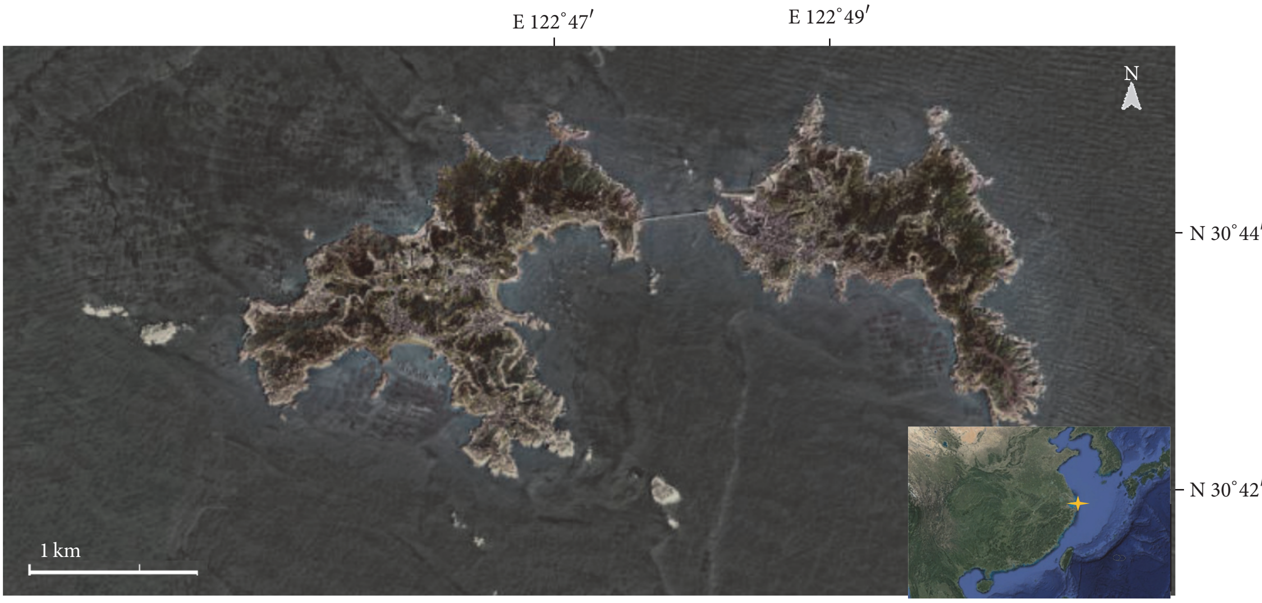

Figure 1: The map of Gouqi Island and its location in East China.

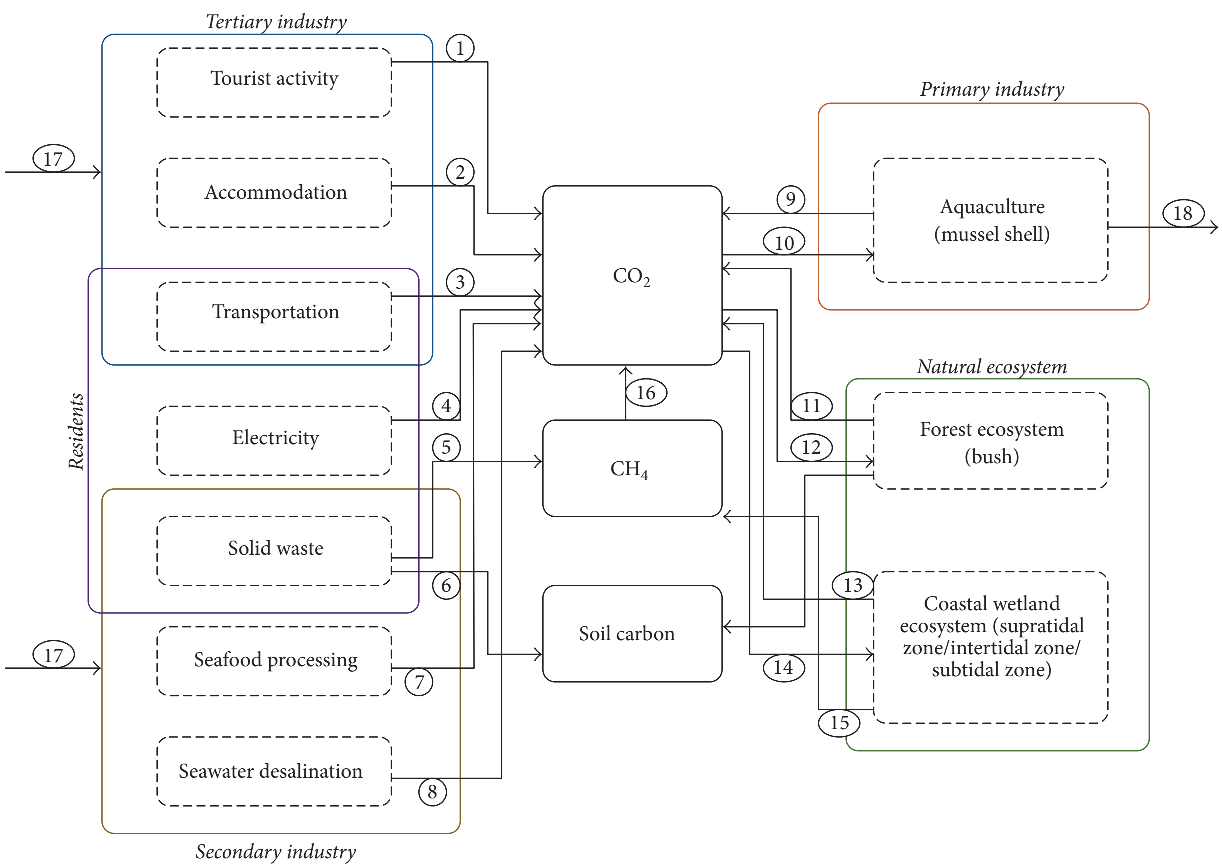

Figure 2: Conceptual model for the social-ecological process carbon cycle model on Gouqi Island. ① Carbon released from tourism activity, ② carbon released from tourism accommodation, ③ carbon released from transportation between island and outside, ④ carbon released of local residents, ⑤ CH_4 released from solid waste landfill, ⑥ carbon stored by soil carbon pool through solid waste landfill, ⑦ carbon released from seafood processing sector, ⑧ carbon released from desalination of seawater, ⑨ carbon released from marine aquaculture, ⑩ carbon captured by marine aqua-culturing, ⑪ respiration in forest ecosystem, ⑫ photosynthesis in forest ecosystem, ⑬ respiration in wetland ecosystem, ⑭ photosynthesis in wetland ecosystem, ⑮ CH_4 released from wetland ecosystem, ⑯ oxidation, ⑰ import, and ⑱ harvest.

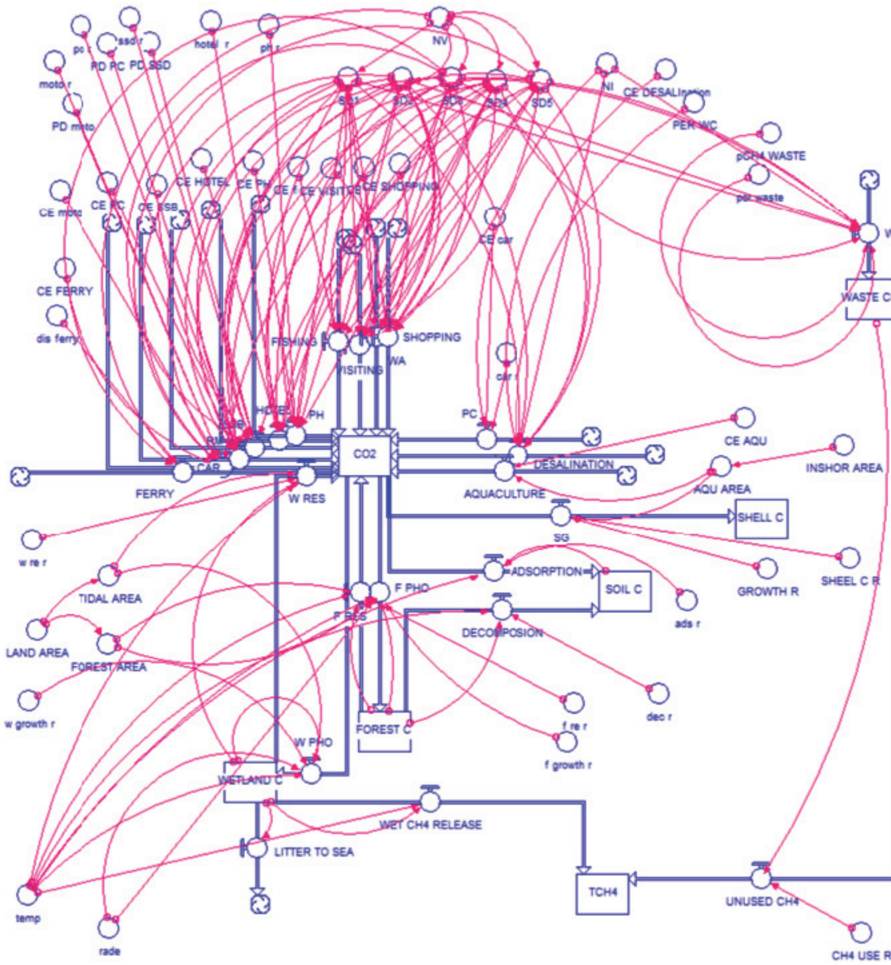

Figure 3: Conceptual STELLA diagram of for the social-ecological process carbon cycle model on Gouqi Island.

Then we apply data of 2014 to the model, calculate the carbon credits and its components, and discuss possible policies for carbon-neutral target based on scenario analysis under different developing strategy. We use year as the basic unit for the analysis, because both the tourism and natural systems are experiencing periodic development each year. We use tons as the unit when calculating the amount of Carbon.

3.2. The Model Components

3.2.1. The State Variables of the Model. The state variables (carbon pools) in the model are expressed the by differential equation following the next format: changes per unit of time equal inputs per unit of time minus outputs per unit of time [19]. The state variables and their symbols in the model are listed in Table 2.

3.2.2. Forcing Functions. The forcing functions or external variables were selected by the development of the conceptual diagram. We list the forcing functions in Table 3 with the symbols used (in STELLA-diagrams a thick arrow with a valve starting or ending with a cloud) in Figure 3.

3.2.3. Processes of the Model. Following the model theory of Jørgensen et al. [16], 24 processes of carbon release and capture ways in social and natural systems of Gouqi Island were described by the development of the conceptual model. Symbols and related units are illustrated in Table 4.

3.2.4. Data Resources of the Model. Published information, questionnaire, and observations by field surveys were used in modelling. Published information is cited from literatures that illustrate similar process or objective, and basic information related to Gouqi Island is cited from the statistic reports. We adopt a Life Cycle Assessment (LCA) questionnaire from Kuo and Chen [13] to collect the tourist data, and the questionnaire includes the following: choice of transportation, choice of accommodation, and activity related to the length of stay (see Appendix); the data are applied directly in the model. Table 5 shows the summary of the parameter symbols used in the model and the resources of the data.

3.2.5. Process Equations. The processes are described either as zero-order, as first-order, or as Michaelis–Menten equations.

Table 2: State variables of the model, all expressed as tons of carbon on Gouqi Island as $f(time)$.

Symbol	Description	Unit
CO_2(t)	Carbon dioxide as $f(time)$	g CO_2/yr
Forest C(t)	Carbon in forest ecosystem as $f(time)$	g C/yr
WETLAND_C(t)	Carbon in wetland ecosystem as $f(time)$	g C/yr
SHELL_C(t)	Carbon in aquaculture mussel shell as $f(time)$	g C/yr
SOIL_C(t)	Carbon in soil sink as $f(time)$	g C/yr
TCH_4(t)	Total methane released as $f(time)$	g CH_4/yr
WASTE_CH_4(t)	Methane released from solid waste as $f(time)$	g CH_4/yr

Table 3: Forcing functions of the model.

Symbol	Meaning
Temp	Temperature
Rade	Solar radiation
Nv	Number of visitors
Ni	Number of inhabitants
Land area	Land area of Gouqi Island
Inshore area	Sea area belonging to Gouqi Island management

Additionally, a logistic growth equation is applied to determine the photosynthetic growth. Basic equations used in the model are illustrated in the following "*Process Equations*". The processes are expressed in the unit $tCyr^{-1}$.

Process Equations. All the equations, parameters, initial values, and forcing functions in the STELLA format are listed in Table 5.

CO2(t) = CO2(t - dt)+(W_RES+F_RES+ AQUACULTURE+PC+DESALINATION+SHOPPING+WA+VISITING+FISHING+HOTEL+PH+FERRY+RM+CAR+SSB - W_PHO - F_PHO - ADSORPTION - SG)*dt

INIT CO2 = 0

INFLOWS:

W_RES = w_re_r*TIDAL_AREA*(20-temp)*WETLAND_C

F_RES = FOREST_C*FOREST_AREA*f_re_r*(20-temp)

AQUACULTURE = AQU_AREA*CE_AQU

PC = car_r*NI*CE_car

DESALINATION = (1*SD1+2*SD2+3*SD3+4*SD4+5*SD5+365*NI)*PER_WC*CE_DESALInation

SHOPPING = (1*SD1+2*SD2+3*SD3+4*SD4+5*SD5)*CE_SHOPPING

WA = (1*SD1+2*SD2+3*SD3+4*SD4+5*SD5)*CE_WA

VISITING = (1*SD1+2*SD2+3*SD3+4*SD4+5*SD5)*CE_VISITING

FISHING = (1*SD1+2*SD2+3*SD3+4*SD4+5*SD5)*CE_fishing

HOTEL = (0*SD1+1*SD2+2*SD3+3*SD4+4*SD5)*hotel__r*CE_HOTEL

PH = (0*SD1+1*SD2+2*SD3+3*SD4+4*SD5)*CE_PH*ph_r

FERRY = NV*dis_ferry*CE_FERRY

RM = (1*SD1+2*SD2+3*SD3+4*SD4+5*SD5)*moto_r*PD_moto*CE_moto

CAR = (1*SD1+2*SD2+3*SD3+4*SD4+5*SD5)*PD_PC*pc_r*CE_PC

SSB = (1*SD1+2*SD2+3*SD3+4*SD4+5*SD5)*PD_SSD*ssd_r*CE_SSB

OUTFLOWS:

W_PHO = WETLAND_C*w_growth_r*TIDAL_AREA*1.05^(20-temp)*(rade/(rade+6)*690)

F_PHO = FOREST_C*f_growth_r*FOREST_AREA*1.05^(20-temp)*(rade/(rade+6)*690)

ADSORPTION = SOIL_C*ads_r*(20-temp)

SG = AQU_AREA*GROWTH_R*SHEEL_C_R/0.27

FOREST_C(t) = FOREST_C(t - dt) + (F_PHO - F_RES - DECOMPOSION)*dt

INIT FOREST_C = 0

INFLOWS:

F_PHO = FOREST_C*f_growth_r*FOREST_AREA*1.05^(20-temp)*(rade/(rade+6)*690)

OUTFLOWS:

F_RES = FOREST_C*FOREST_AREA*f_re_r*(20-temp)

DECOMPOSION = FOREST_C*dec_r*(20-temp)

SHELL_C(t) = SHELL_C(t - dt) + (SG)*dt

INIT SHELL_C = 0

INFLOWS:

SG = AQU_AREA*GROWTH_R*SHEEL_C_R/0.27

SOIL_C(t) = SOIL_C(t - dt) + (ADSORPTION + DECOMPOSION) * dt

INIT SOIL_C = 0

INFLOWS:

ADSORPTION = SOIL_C*ads_r*(20-temp)

DECOMPOSION = FOREST_C*dec_r*(20-temp)

Table 4: Processes of the model.

Process symbol	Meaning	Unit
W RES	CO_2 released from respiration of plants in wetland ecosystem	g CO_2
F RES	CO_2 released from respiration of wetland plants in forest ecosystem	g CO_2
AQUACULTURE	CO_2 released from marine aquaculture industry	g CO_2
PC	CO_2 released from the use of private cars on the island	g CO_2
DESALINATION	CO_2 released from seawater desalination	g CO_2
SHOPPING	CO_2 released from tourist shopping	g CO_2
WA	CO_2 released from tourist water activity that used fossil fuels-driven motors	g CO_2
VISITING	CO_2 released from tourist sight-seeing on the island	g CO_2
FISHING	CO_2 released from tourist offshore angling	g CO_2
HOTEL	CO_2 released from tourist accommodation (hotel)	g CO_2
PH	CO_2 released from tourist accommodation (private home)	g CO_2
FERRY	CO_2 released from ferries that connect the island and outside	g CO_2
RM	CO_2 released from tourist transportation by rental motorcycles on the island	g CO_2
CAR	CO_2 released from tourist transportation by private car on the island	g CO_2
SSB	CO_2 released from tourist transportation by small shuttle bus on the island	g CO_2
W PHO	CO_2 captured by wetland plants through photosynthesis	g CO_2
F PHO	CO_2 captured by forest plants through photosynthesis	g CO_2
ADSORPTION	CO_2 captured by soil respiration	g CO_2
SG	CO_2 captured by mussel for shell growth during aquaculture	g C
DECOMPOSITION	Carbon entering the soil from forest vegetation litter	g C
WETCH_4 RELEASE	CH_4 released from wetland ecosystem	g CH_4
UNUSED CH4	CH_4 released from solid waste landfill that are not collected for further use	g CH_4
WASTE	CH_4 released from solid waste landfill	g CH_4
LITTER TO SEA	Carbon entering marine ecosystem from wetland plant litter	g C

TCH4(t) = TCH4(t - dt) + (WET_CH4_RELEASE + UNUSED_CH4)*dt

INIT TCH4 = 0

INFLOWS:

WET_CH4_RELEASE = 0.1*WETLAND_C*(20-temp)

UNUSED_CH4 = WASTE_CH4*CH4_USE_R

WASTE_CH4(t) = WASTE_CH4(t - dt) + (WASTE - UNUSED_CH4)*dt

INIT WASTE_CH4 = 0

INFLOWS:

WASTE = (1*SD1+2*SD2+3*SD3+4*SD4+5*SD5+365*NI)*per_waste*pCH4_WASTE

OUTFLOWS:

UNUSED_CH4 = WASTE_CH4*CH4_USE_R

WETLAND_C(t) = WETLAND_C(t - dt) + (W_PHO - W_RES - LITTER_TO_SEA - WET_CH4_RELEASE)*dt

INIT WETLAND_C = 0

INFLOWS:

W_PHO = WETLAND_C*w_growth_r*TIDAL_AREA*1.05^(20-temp)*(rade/(rade+6)*690)

OUTFLOWS:

W_RES = w_re_r*TIDAL_AREA*(20-temp)*WETLAND_C

LITTER_TO_SEA = 0.5*WETLAND_C

WET_CH4_RELEASE = 0.1*WETLAND_C*(20-temp)

ads_r = 0.1

AQU_AREA = 0.005*INSHOR_AREA

car_r = 0.1

CE_AQU = 50000

CE_car = 5400000

CE_DESALInation = 2784

CE_FERRY = 106

CE_fishing = 1670

CE_HOTEL = 7900

CE_moto = 0

CE_PC = 63

CE_PH = 1619

CE_SHOPPING = 344

CE_SSB = 40

CE_VISITING = 417

CE_WA = 15300

CH4_USE_R = 1

Table 5: Summary of the parameter symbols used in the model and the resources of the data. (SR = statistic report; FS = field survey; Q = questionnaire; R = reference.)

Abbreviation	Meaning	Unit	Source
Aqu area	Aquaculture area of mussel	km^2	SR
Ads r	Adsorption rate of soil carbon pool	$G\,Cm^{-2}yr^{-1}$	R [12]
Car r	Possessing rate of private car	%	SR
CE AQU	CO_2 released from aquaculture industry	$g\,CO_2km^{-2}$	FS
CE CAR	CO_2 released from small private car per km	$g\,CO_2km^{-1}$	FS
CE DESALINATION	CO_2 released through the desalination per ton of sea water	$g\,CO_2t^{-1}$	FS
CE FERRY	CO2 released from ferry per kilometer	$g\,CO_2km^{-1}$	FS
CE FISHING	CO_2 released from sea fishing per hour	$g\,CO_2hr^{-1}$	FS
CE MOTO	CO_2 released from motor bicycle per km	$g\,CO_2km^{-1}$	R [13]
CE HOTEL	CO_2 released from hotel per day	$g\,CO_2$/night	R [13]
CE Ph	CO_2 released from private house per day	$g\,CO_2$/night	R [13]
CE SHOPPING	CO_2 released from tourist shopping per tourist per time	$g\,CO_2$/visitor	R [13]
CE SSB	CO_2 released from small shuttle bus	$g\,CO_2km^{-2}$	FS
CE VISITING	CO_2 released from cultural tourism activities	$g\,CO_2hr^{-1}$	R [14]
CE WA	CO_2 released from water tourism activities	$g\,CO_2hr^{-1}$	FS
CH4 UNUSE R	Unused rate of CH_4 released from solid waste	%	SR
Dec r	Decomposition rate of forest litterfall	$g\,Cm^{-2}yr^{-1}$	R [14]
Dis ferry	Average driving distance of ferry	km	SR
Forest area	Coverage rate of forest	%	SR
F growth	Growth rate of forest	$g\,Cm^{-2}yr^{-1}$	R [14]
F re r	Respiration rate of forest	$g\,Cm^{-2}yr^{-1}$	R [14]
Growth r	Growth rate of mussel	$g\,Cm^{-2}yr^{-1}$	FS
NI	Number of inhabitants	/	SR
NV	Number of visitors	/	SR
Hotel r	Proportion of tourists choosing hotel	%	Q
Ph r	Proportion of tourists choosing private home	%	Q
Moto r	Proportion of tourists renting motorcycles on the island for transportation	%	Q
Pc r	Portions of tourists driving private cars on the island for transportation	%	Q
Ssd r	Portions of tourists taking small buses on the island for transportation	%	Q
PCH4 WASTE	CH_4 released from solid waste of per kilogram	$g\,CH_4kg^{-1}$	R [13]
Pd moto	Average driving distance of motorcycles per day	km	Q
Pd pc	Average driving distance of private cars	km	Q
Pd ssd	Average driving distance of small buses	km	Q
Per waste	Average solid waste generated by per person	kg	SR
Per WC	Average amount of freshwater consumed by per person per day	t	SR
Rade	Average solar radiation	$MJ\,m^{-2}yr^{-1}$	R [14]
SD1	Proportion of tourists stay for one day	%	Q
SD2	Proportion of tourists stay for two days	%	Q
SD3	Proportion of tourists stay for three days	%	Q
SD4	Proportion of tourists stay for four days	%	Q
SD5	Proportion of tourists stay for more than five days	%	Q
SHELL C R	Proportion of carbon in per kilogram mussel	%	FS
Temp	Temperature	$^{\circ}C$	SR
W growth r	Growth rate of wetland plants	$g\,Cm^{-2}yr^{-1}$	R [15]
W re r	Respiration rate of wetland plants	$g\,Cm^{-2}yr^{-1}$	R [15]

dec_r = 0.15
dis_ferry = 140
FOREST_AREA = 0.53*LAND_AREA
f_growth_r = 0.1
f_re_r = 0.2
GROWTH_R = 1500000
hotel__r = 0.3
INSHOR_AREA = 1500
LAND_AREA = 6.62
moto_r = 0.2
NI = 10000
NV = 1000000
pCH4_WASTE = 80000
pc_r = 0.1
PD_moto = 6
PD_PC = 20
PD_SSD = 14
per_waste = 1.1
PER_WC = 1.2
ph_r = 0.7
rade = 1.2
SD1 = 0.1*NV
SD2 = 0.2*NV
SD3 = 0.5*NV
SD4 = 0.1*NV
SD5 = 0.1*NV
SHEEL_C_R = 0.95
ssd_r = 0.7
temp = 8
TIDAL_AREA = 0.14*LAND_AREA
w_growth_r = 0.1
w_re_r = 0.2

4. Result

Table 6 shows the general result of the carbon cycle process of Gouqi Island in 2014, Figure 4 indicates the island has achieved a positive removal of CO_2 at the amount of 15540 t. Based on the average data from 2011 to 2013, the carbon credit of Gouqi Island is −14460 t, 36780 tons of carbon is released to atmosphere with the form of CO_2, and 51240 tons of carbon is captured by the ecosystem. The three major resources of carbon emission are transportation and tourism development (the emission from tourist transportation are excluded) and seawater desalination, which contribute 35.2%, 20.0%, and 18.7% of the total carbon emission. The process of soil respiration, including soil microbial respiration, root respiration, soil animal respiration, is also a considerable carbon source, which contributes 17.1% of the total carbon emission.

Table 6: General result of the carbon cycle model for Gouqi Island with the data of 2014.

Item	Carbon emission ($t\,Cyr^{-1}$)	Carbon sink ($t\,Cyr^{-1}$)
Ferry	11070	
Small shuttle bus	310	
Private car	1560	
Fishing	1300	
Visiting	320	
Water activity	1200	
Shopping	270	
Hotel	1220	
Private home	600	
Desalination	6870	
Mussel aquaculture activity	2100	
Shell aquaculture		41620
Soil respiration	6430	
Forest ecosystem		3750
Wetland ecosystem		5870
Solid waste	700	

Forest ecosystem and wetland ecosystem contribute to 7.3% and 11.4% of the total carbon capture. The island of Gouqi is named after a kind of major bush Chinese wolfberries *Lycium chinense* (in Chinese it is called Gouqi), which covers 53% percent of the mainland on the island. The ability of carbon sink of the bush forest is weaker than the evergreen broad-leaved forest at the same latitude [20]. Rock estuary is the major constitutes of the wetland ecosystem on the island, and thus its ability of carbon sink is weaker than the tidal marsh at the same latitude [21].

Mussel culturing industry is the most important carbon sink on the carbon cycle model of the island, which contributes to 50.3% of the carbon exchange and 81.2% among the total carbon sink. Shellfish utilize dissolved HCO_3^- from seawater carbon to generate calcium carbonate ($CaCO_3$) shells after $Ca^{2+} + 2HCO_3^- = CaCO_3 + CO_2 + H_2O$ [22]. As the output of the primary industry and major input for the secondary industry on the island, mussel itself plays an important role for the carbon neutrality on the island.

5. Discussion

The application of the model as the environmental management tool could give the advices on the following questions.

(1) What Will Happen with the Carbon Balance If the Island Transportation Becomes Fossil Fuel-Free? The coastal line around the island is 7.1 km; it is possible that we use electricity as the power of transportation in place of fossil fuels. As illustrated in Figure 4, if we use electricity that generates no carbon emission, 1890 $tCyr^{-1}$ can be saved. Further, if

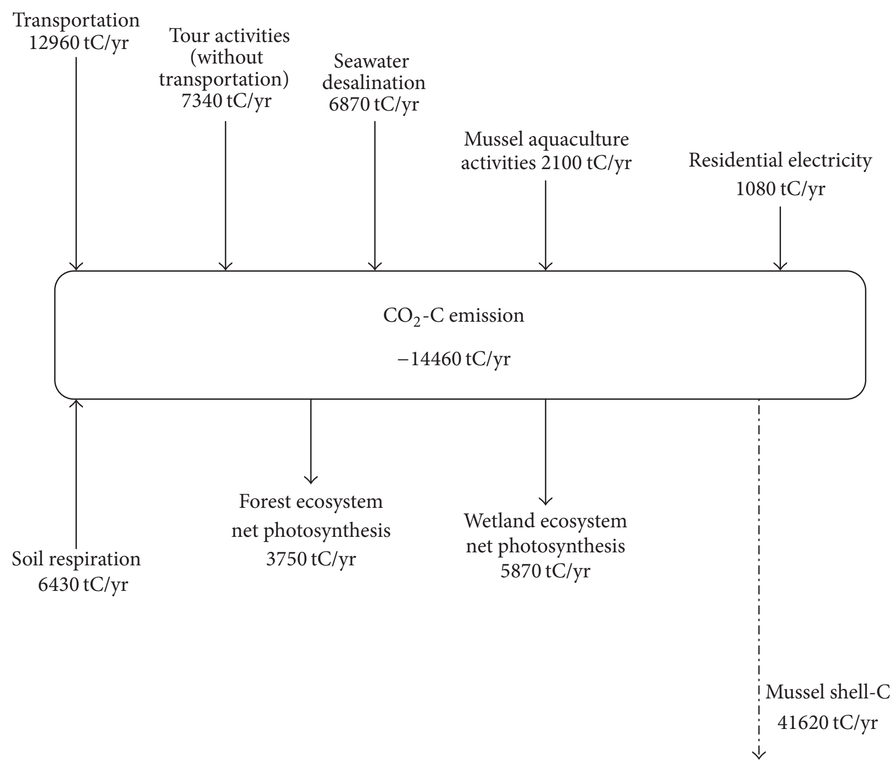

Figure 4: The carbon balance based on the results of the social-ecological process carbon cycle model on Gouqi Island.

the ferry can be driven by electricity, 12960 $tCyr^{-1}$ can be reduced, which can decrease 40.4% of the total carbon emission of the island.

(2) What Will Happen with the Carbon Cycle If the Island Develops More Carbon Sink Industry? As the above analysis, the carbon credits of the island were negative because mussel plays a critical role of carbon capturing during its growth, which contributes more than 50% of the total carbon removal. As the mussel culturing has a high requirement of sea currents, the sea area around the island is limited and can hardly extend new area for mussel culturing; what is more, the frequent hurricane is a threat to the industry. Macroalga is less restricted by sea currents and has significant wave-breaking effect and carbon-capturing effect at the rate 3350 tC/yr·km^2. If ecological project of marine carbon sink is introduced to the island, the growing condition of mussel can be safer, and the carbon credit of the island can be further improved. Assuming that we culture large alga beyond the mussel culturing area at a scale of three times of current mussel area, the total carbon credit will improve 70%.

(3) What Will Happen with the Carbon Cycle If the Island Conducts Sustainable Tourism Strategy? Tourism, characterized by the rapidly increasing tourists number, is now a strategic direction for Gouqi Island with great potential on both economic and social perspective [23]. Meanwhile, tourism has been identified as an important contributor to carbon emission, accounting for a share of about 5% of global emissions CO_2 [24], especially on the island-featured destination which relies heavily on fossil fuels to provide transportation, accommodation, food, and tourism activities [25]. In this article, we focus on the environmental influence of tourism considering the carbon neutrality on the island; thus we just consider the transportation between the island and the major ferry ports and on the island. If we want to discuss the carbon credit of the sustainable tourism on the island, it is necessary to consider the carbon emission inbound and outbound transportation [26]. It is necessary for the island to consider the components of carbon cycle on the island at different scenarios according to different tourism development strategy, for instance, tourists scale, tourist-guest source marketing, tourism attractions, total area and contents for tourist activity, tourism infrastructure construction, tourist accommodation upgrading, and so forth. This will help the island to achieve carbon neutrality balancing the economic and ecological goal.

6. Conclusion

Sustainable tourism destination has been an important research question, particularly for the island destination which has complex social-ecological interaction. We develop a carbon cycle model of Gouqi Island illustrating the carbon credit and its components. With the average data from 2011 to 2013, we can calculate that the carbon credit of the island in 2014 was −14460 tC/yr. The main sources of carbon emission are transportation, tourist activity, and seawater desalination, while the major carbon sink is mussel culturing which contributes to more than 50% of the carbon capture.

We then discuss the carbon credit under different scenarios, for instance, the use of fossil-free energy in transportation and the development of ecological project such as large alga.

The carbon cycle model as a tool for environmental management model can extend to the development of industrial strategic design, by the way of modifying the coefficient or process in the model according to different strategy.

Appendix

Questionnaire Sample

Tourist No. —

Duration of the trip: From — to —

Day 1

Accommodation type:

□ Hotel
□ Private home
□ Others.

Stay — hrs

Transportation type:

Site A to Site B (travel distance: — km), vehicle: —
Site B to Site C (travel distance: — km), vehicle: —
Site C to Site D (travel distance: — km), vehicle: —

Activity:

□ Sight seeing: — hrs;
□ Fishing: —

Day 2

⋮

Day *N*

Competing Interests

The authors declare that they have no competing interests.

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References

[1] L. M. Varnell, D. A. Evans, and K. J. Havens, "A geomorphological model of intertidal cove marshes with application to wetlands management," *Ecological Engineering*, vol. 19, no. 5, pp. 339–347, 2003.

[2] R. T. Woodward and Y.-S. Wui, "The economic value of wetland services: a meta-analysis," *Ecological Economics*, vol. 37, no. 2, pp. 257–270, 2001.

[3] G. C. Ray, "Coastal-zone biodiversity patterns," *BioScience*, vol. 41, no. 7, pp. 490–498, 1991.

[4] K. B. Gedan, M. L. Kirwan, E. Wolanski, E. B. Barbier, and B. R. Silliman, "The present and future role of coastal wetland vegetation in protecting shorelines: answering recent challenges to the paradigm," *Climatic Change*, vol. 106, no. 1, pp. 7–29, 2011.

[5] S. Y. Lee, R. J. K. Dunn, R. A. Young et al., "Impact of urbanization on coastal wetland structure and function," *Austral Ecology*, vol. 31, no. 2, pp. 149–163, 2006.

[6] M. Sheaves, R. Baker, I. Nagelkerken, and R. M. Connolly, "True value of estuarine and coastal nurseries for fish: incorporating complexity and dynamics," *Estuaries and Coasts*, vol. 38, no. 2, pp. 401–414, 2015.

[7] R. L. King, *Spatial Ecology and Wetland Use of Aquatic Turtles in the Coastal Plain of Georgia*, The University of Georgia, 2014.

[8] W. J. Mitsch and N. Wang, "Large-scale coastal wetland restoration on the laurentian great lakes: determining the potential for water quality improvement," *Ecological Engineering*, vol. 15, no. 3-4, pp. 267–282, 2000.

[9] Q. Wang, J. Lu, S. T. Hsieh, P. Bowler, F. Wang, and Y. Ma, "Wetland research and eco-development through Chinese-United States EcoPartnership relationships," *Journal of Renewable and Sustainable Energy*, vol. 7, no. 4, Article ID 041503, 2015.

[10] S. Gössling, "Carbon neutral destinations: a conceptual analysis," *Journal of Sustainable Tourism*, vol. 17, no. 1, pp. 17–37, 2009.

[11] D. Reiche, "Renewable energy policies in the gulf countries: a case study of the carbon-neutral 'Masdar City' in Abu Dhabi," *Energy Policy*, vol. 38, no. 1, pp. 378–382, 2010.

[12] *2014 Statistical Yearbook of Shengsi County*, Statistical Bureau of Shengsi County, Zhoushan, China, 2015.

[13] N.-W. Kuo and P.-H. Chen, "Quantifying energy use, carbon dioxide emission, and other environmental loads from island tourism based on a life cycle assessment approach," *Journal of Cleaner Production*, vol. 17, no. 15, pp. 1324–1330, 2009.

[14] X. Fei, D. Zekkos, and L. Raskin, "Quantification of parameters influencing methane generation due to biodegradation of municipal solid waste in landfills and laboratory experiments," *Waste Management*, 2015.

[15] L. Zou, L. Wang, A. Lin, H. Zhu, Y. Peng, and Z. Zhao, "Estimation of global solar radiation using an artificial neural network based on an interpolation technique in southeast China," *Journal of Atmospheric and Solar-Terrestrial Physics*, vol. 146, pp. 110–122, 2016.

[16] S. E. Jørgensen, B. D. Fath, S. Bastianoni et al., *A New Ecology: Systems Perspective*, Elsevier, 2011.

[17] S. E. Jørgensen and S. N. Nielsen, "A carbon cycling model developed for the renewable Energy Danish Island, Samsø," *Ecological Modelling*, vol. 306, pp. 106–120, 2015.

[18] S. Becken and M. Patterson, "Measuring national carbon dioxide emissions from tourism as a key step towards achieving sustainable tourism," *Journal of Sustainable Tourism*, vol. 14, no. 4, pp. 323–338, 2006.

[19] S. E. Jørgensen and G. Bendoricchio, *Fundamentals of Ecological Modeling*, Elsevier, 2001.

[20] C.-Z. Guo, Y.-Y. Wu, and J. Ni, "Estimation for vegetation carbon storage in Tiantong National Forest Park," *The Journal of Applied Ecology*, vol. 25, no. 11, pp. 3099–3109, 2014.

[21] A. Ma, J. Lu, and T. Wang, "Effects of elevation and vegetation on methane emissions from a freshwater estuarine wetland," *Journal of Coastal Research*, vol. 28, no. 6, pp. 1319–1329, 2012.

[22] Q. Tang, J. Zhang, and J. Fang, "Shellfish and seaweed mariculture increase atmospheric CO2 absorption by coastal ecosystems," *Marine Ecology Progress Series*, vol. 424, pp. 97–104, 2011.

[23] *2016 Report of the Government of Zhejiang Province, China*, Government of Zhejiang Province, 2016.

[24] U. N. E. Programme, "Climate change and tourism: responding to global challenges," *Climate Change & Tourism Responding to Global Challenges*, vol. 12, no. 4, pp. 168–181, 2008.

[25] S. Gössling and K. P. Schumacher, "Implementing carbon neutral destination policies: issues from the Seychelles," *Journal of Sustainable Tourism*, vol. 18, no. 3, pp. 377–391, 2010.

[26] S. Gossling, "Carbon neutral destinations: a conceptual analysis," *Journal of Sustainable Tourism*, vol. 17, no. 1, pp. 17–37, 2009.

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Research Article

Composition and Biomass of Aquatic Vegetation in the Poyang Lake, China

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The distribution of aquatic vegetation and associated community diversity and biomass in the Poyang Lake were investigated. The results showed that (1) 43 species of aquatic vascular plants were found in the Poyang Lake watershed which belonged to 22 families; (2) the vegetation of the Poyang Lake scattered in different areas which could be divided into 31 major plant communities and 5 plant zones including amphibian, emergent, floating-leaved, submerged, and floating input; (3) there were 67 aquatic plants in the lake area, and the standing stock (fresh weight) was 1519.41 t. The number of amphibians was the dominant plant species in the Poyang Lake, and the quantity and percentage of amphibians were predominant, which was far more than the other three life forms.

1. Introduction

Aquatic plants are an important component of lake ecosystems and are often regarded as indicators of lake environmental changes; they play an important role in maintaining the structure and function of lake ecosystems [1, 2]. Aquatic plants are the primary producers of aquatic ecosystems, being a kind of food source of many kinds of fish and other aquatic animals [3, 4]. They can regulate the lake water body, degrade various pollutants, and improve the transparency [5, 6]. In addition, aquatic plants can provide habitat to many organisms and increase the spatial niche of aquatic ecosystems [7, 8]. Therefore, it is important to know the distribution of aquatic plants and their communities to analyze the status of aquatic plants in the lake area.

Vegetation is an important component of wetland ecosystems, and biomass can quantify the contribution of wetland vegetation to carbon sinks and carbon sources [9]. Biomass estimation of wetlands plays an important role in understanding dynamic changes of the wetland ecosystem [10]. In previous studies, biomass estimation using indirect and direct methods has been conducted. For example, remote sensing technology and radar data have been widely used to estimate the biomass in recent years [2, 11]. Han et al. [11] investigated four decades of winter wetland changes in the Poyang Lake based on Landsat observations between 1973 and 2013. Luo et al. [2] revealed the distribution of aquatic vegetation types in Taihu Lake, China, by using a series of remotely sensed images with a resolution of 30 m (HJ-CCD and Landsat TM). Shen et al. [9] retrieved vegetation biomass in the Poyang Lake wetland by using polarimetric RADARSAT-2 data. These indirect methods offer the ability to continuously monitor growth and phenology on the same individuals over large areas but depend on the existence of strong relationships between the predictor variables and plant biomass [12–14]. Therefore, ground investigation of biomass is also very essential.

Poyang Lake, the largest freshwater lake in China, is well known for its ecological importance as a wetland system [9, 11]. The Poyang Lake region provides significant environmental benefits, such as supplying water resources and maintaining carbon storage and biodiversity [15, 16]. Guan et al. [17] pointed out that aquatic vegetation of Poyang Lake is very rich in number of species. The vegetation distributes in about 2262 km^2 and accounts for 80.8% area of the lake. Jian et al. [18] studied the distribution area and biomass of Poyang Lake beach vegetation in the years of 1999 and 2000, and they found there were 28 families, 56 genera, 95 species, and 3 varieties. In April, 2000, they found that the total biomass of beach vegetation was 3.81×10^6 tons (fresh weight), with an average biomass of 3736 g·m^{-2} (fresh weight). Peng et al. [19] studied the association type diversity, community species diversity, community coverage, and biomass of aquatic plants in the fresh water lakes of the Poyang Plain District of China in 2001, and they found 42 associations of aquatic plants in the lakes, which include 11 amphibian, 6 emergent, 11 leaf-floating, and 14 submerged associations. Among all the associations, *Carex cinerascens* Ass. and *Zizania latifolia* Ass. possess the highest coverage and highest biomass, respectively. Although some survey of aquatic plants in the Poyang Lake has been conducted, ground investigation of aquatic plants in Poyang Lake was very rare in recent years.

In this paper, the species and biomass of aquatic plants in the Poyang Lake were investigated based on ground survey data in September, 2013. And the objective of this study is to investigate the composition of aquatic plants and community types and habitats of major aquatic plants in the Poyang Lake. This study will provide information for the monitoring and conservation of aquatic plant in the Poyang Lake.

2. Data and Methodology

2.1. Study Area and Data. Poyang Lake is located on the southern bank of the lower Yangtze reach (28° 22′∼29° 45′N and 11° 47′∼116° 45′E). It has five main tributary rivers: Ganjiang River, Fuhe River, Xinjiang River, Raohe River, and Xiushui River, and several smaller rivers. The basin area of the five rivers is 162,200 km^2, occupying 9% of Yangtze River basin. The climate is characterized as a subtropical, humid, monsoon climate with a 1620 mm mean annual precipitation and an annual average temperature of approximately 17°C [20].

The environmental conditions of the Poyang Lake are very suitable for the growth and reproduction of aquatic plants. Aquatic plants can provide habitat for many organisms, improve habitat diversity, and increase the spatial niche of aquatic ecosystem [21]. But the annual water level of the Poyang Lake changes greatly, especially in the flood season (April to November) and dry season (December to March). The characteristics of periodical variation of water level in the Poyang Lake might lead to the conversion of the lake beach and grassland.

2.2. Methodology. The species, community structure, coverage, and biomass of the Poyang Lake were investigated from September 7, 2013, to September 14, 2013. GPS and lake electronic map were used to set sampling points, and the samples of aquatic plants were collected at each sampling site. The category of aquatic plants in the Poyang Lake was defined according to the reference of Cook [22]. Division of "cluster" is based on the principle of dominant species, which is named as the name of the group. If a cluster is with two or more dominant species, different dominant species of the same layer were connected with "+," and the dominant species of different layers were connected with "−." At the sampling sites, amphibians, emergent plants, floating-leaved plants, and floating input plants were directly observed and recorded. Submerged plants were preliminarily identified with water sickle after collecting water and the characters of flowers, leaves, and fruits of each plant as well as field growth photos and accurate identification of aquatic plant species and genus [23]. For amphibian and emergent plant clusters, 6 samples of 2 m × 2 m were set randomly within 500 m^2 of each sampling point, and the total biomass (fresh quality) of the aerial parts of all the plants was measured and the biomass per unit area was calculated. For plants with large plant biomass, such as *Phragmites communis* Trin., we could estimate the number of plants in the quadrats. After selecting the representative fresh weight of the plants, we could calculate the standing amount of the plants. As the species with smaller biomass, such as *Alternanthera philoxeroides* (Mart.) Griseb., all the plants in the quadrats were collected and their fresh weight was calculated, and the mean values were calculated after several measurements of plant height. The floating-leaved plant cluster, the floating input plant cluster, and the submerged plant cluster were harvested 6 times randomly in the range of 500 m^2 for each sampling point with an underwater sickle with a cross-sectional area of 0.785 m^2, and the plant was uprooted and then washed, removing residual sticks and other impurities. The fresh weight of each plant was weighed and the frequency, coverage area, and coverage were recorded (visual method). All the plants, together with the roots, were dug and washed, and the plant depth was measured [24].

3. Results and Discussion

3.1. Characteristics of Aquatic Plants in the Poyang Lake. There were 43 species of aquatic plants in the lake area which belonged to 37 genera and 22 families according the survey in 2013 (see Figure 1 and Table 1). And it could be found that there were 16, 13, and 9 species, accounting for 37.21%, 30.23%, and 20.93% of the total species for the amphibians, emergent plants, and submerged plants, respectively. However, there were only 2 and 3 species for floating input plants and floating-leaved plants in the Poyang Lake, accounting for 4.65% and 6.98% of the total species.

It could be seen that the distribution of all kinds of plants was different; the most extensive degrees of various classes were *Carex* spp., *Eleocharis tuberosa* (Roxb.) Roem. et Schult., *Polygonum* L., and *Nymphoides peltatum* (Gmel.) O. Kuntze. Most of the species were distributed in a large area or in a continuous distribution in the wetland or near shore of the lake, which were widely distributed and had strong adaptability to adversity. Most of the plants in the whole lake were in vegetative period.

The survey recorded 22 families, 37 genera, and 43 aquatic plant species; the number of species was significantly lower than the survey of aquatic plant species number in Poyang

Table 1: The composition of macrophytes in the Poyang Lake (September 7–14, 2013).

Serial number	Plant community	Phenological period	Lifestyle
1	*Sagittaria pygmaea* Miq.	Flowering fruit bearing stage	Emerged plant
2	*Trapella sinensis* Oliv.	Fruit period (florescence)	Floating-leaved plant
3	*Carex rhynchophysa* C. A. Mey.	Vegetative period	Amphibian
4	*Eichhornia crassipes* (Mart.) Solms	Florescence	Floating input plant
5	*Aeschynomene indica* Linn.	Flowering fruit bearing stage	Emerged plant
6	*Sambucus chinensis* Lindl.	Vegetative period	Amphibian
7	*Isachne globosa* (Thunb.) Kuntze	Florescence	Amphibian
8	*Limnophila sessillflora* (Vahl.) Bl.	Vegetative period	Amphibian
9	*Paspalum distichum* L.	Vegetative period	Emerged plant
10	*Hydrocharis dubia* (Bl.) Back.	Fruit period	Floating input plant
11	*Ottelia alismoides* (L.) Pers.	Flowering fruit bearing stage	Submerged plant
12	*Ludwigia adscendens* (L.) Hara	Flowering fruit bearing stage	Amphibian
13	*Juncellus serotinus*	Flowering fruit bearing stage	Amphibian
14	*Blyxa japonica* (Miq.) Maxim.	—	Submerged plant
15	*Marsilea quadrifolia* L.	Vegetative period	Amphibian
16	*Nephrolepis cordifolia* (L.) Presl	Fruit period	Amphibian
17	*Carex* spp.	Florescence	Amphibian
18	*Microcarpaea minima* (Koen.) Merr.	Vegetative period	Amphibian
19	*Oryza rufipogon* Griff.	Flowering fruit bearing stage	Amphibian
20	*Myriophyllum spicatum* L.	Vegetative period	Submerged plant
21	*Typha angustifolia* L.	Fruit period	Emerged plant
22	*Monochoria vaginalis* (Burm.f.) Presl	—	Emerged plant
23	*Rotala rotundifolia* (Buch.-Ham) Koehne	—	Emerged plant
24	*Imperata cylindrica* (Linn.) Beauv	Vegetative period	Amphibian
25	*Eleocharis tuberosa* (Roxb.) Roem. et Schult.	Vegetative period	Emerged plant
26	*Najas marina*	Fruit period	Submerged plant
27	*Triarrherca sacchariflora*	Vegetative period	Emerged plant
28	*Cynodon dactylon* (Linn.) Pers.	Vegetative period	Amphibian
29	*Zizania caduciflora* (Turcz. ex Trin.) H.-M.	Vegetative period	Emerged plant
30	*Hydrilla verticillata* (L.f.) Royle	Vegetative period	Submerged plant
31	*Utricularia aurea* Lour.	Florescence	Submerged plant
32	*Ceratophyllum demersum* L.	Vegetative period	Submerged plant
33	*Vallisneria natans* (Lour.) Hara	Flowering fruit bearing stage	Submerged plant
34	*Utricularia vulgaris* L.	Flowering fruit bearing stage	Submerged plant
35	*Polygonum* L.	Vegetative period	Amphibian
36	*Trapa bispinosa* Roxb.	Flowering fruit bearing stage	Floating-leaved plant
37	*Phragmites communis* Trin.	Vegetative period	Emerged plant
38	*Triarrhena lutarioriparia*	Vegetative period	Amphibian
39	*Alternanthera Philoxeroides* (Mart.) Griseb.	Florescence	Emerged plant
40	*Nymphoides peltatum* (Gmel.) O. Kuntze	Florescence	Floating-leaved plant
41	*Limnophila heterophylla* (Roxb.) Benth.	Flowering fruit bearing stage	Emerged plant
42	*Myriophyllum tuberculatum* Roxburgh	Fruit period	Emerged plant
43	*Elymus dahuricus* Turcz.	Vegetative period	Amphibian

Lake [3], which may be because the latter sampling sites in the Poyang Lake were a bit too much. The results of this study were compared with the results of the survey of 28 families, 56 genera, 95 species, and 3 varieties of aquatic vegetation in the Poyang Lake [18]. The number of species was different and the reason may be the strength of this investigation was not enough, and the accuracy of traditional field survey research method was much lower than that of TM Landsat remote sensing technology. In view of this survey was only a single survey in autumn and was limited to water habitat, lakeside vegetation was not a detailed investigation, and, therefore, the actual distribution of Poyang Lake aquatic plant species should be lower than the findings of Guan et al. [3] and Jian et al. [18].

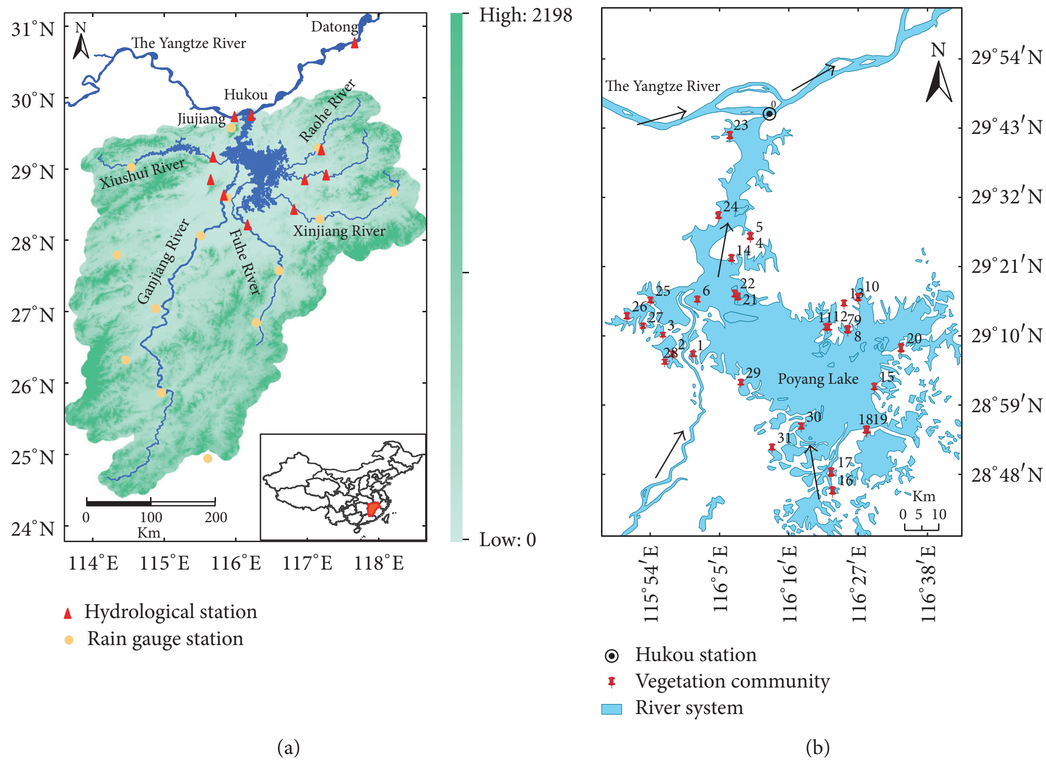

Figure 1: Location of the Poyang Lake river basin (a) and concerned distribution of the vegetation chart (b).

3.2. Community Types and Habitats of Major Aquatic Plants in the Poyang Lake. The species of aquatic plants, the spatial structure, and ecological environment of the community were listed in Table 2 in the Poyang Lake. There were 31 main plant clusters in the Poyang Lake.

The amphibians mainly distributed in the water depth of 0.5–2.5 m meters of the beach in the flood season. The main species were amphibious plants that grew in both shallow and wetlands. There were 16 kinds of amphibians in the Poyang Lake, of which the most important and wide distribution was *Carex* spp., the least distribution was *Sambucus chinensis* Lindl., *Isachne globosa* (Thunb.) Kuntze, *Microcarpaea minima* (Koen.) Merr., and *Oryza rufipogon* Griff.

Emergent plants were located in the 12–15 m elevation of the shoal, and the water depth was generally 0.5–3.5 m in the flood season. The main species were plants that had only the base or lower part of the plant to be submerged in water, but the upper part of the plant was quite out of water. There were 13 kinds of emergent plants in the Poyang Lake, of which the most important and wide distribution was *Eleocharis tuberosa* (Roxb.) Roem. et Schult., and the least distribution was *Aeschynomene indica* Linn., *Typha angustifolia* L., *Monochoria vaginalis* (Burm.f.) Presl, *Rotala rotundifolia* (Buch.-Ham) Koehne, and *Myriophyllum tuberculatum* Roxburgh.

Floating-leaved plants were located 11–13 meters above the lake bottom, where the lake water level was generally 2.5–4.5 meters in the flood season. The main species were some plants rooted in the lake, but the leaves were floating on the surface of the water. Moreover, a large number of submerged plants could also be found in this plant belt, such as *Vallisneria natans* (Lour.) Hara and *Hydrilla verticillata* (L.f.) Royle. There were 3 kinds of floating-leaved plants in the Poyang Lake, of which the most important and wide distribution was *Nymphoides peltatum* (Gmel.) O. Kuntze, and the least distribution was *Trapella sinensis* Oliv.

Submerged plants distributed in the 9–12 m elevation of the lake, and the lake water depth was generally 3.5–6.5 meters in the flood season. The main species were some of the plants immersed in water, such as *Hydrilla verticillata* (L.f.) Royle, *Ceratophyllum demersum* L. There were 9 kinds of submerged plants in the Poyang Lake, of which the most important and wide distribution was *Ceratophyllum demersum* L. and *Vallisneria natans* (Lour.) Hara, and the least distribution was *Blyxa japonica* (Miq.) Maxim., *Myriophyllum spicatum* L., and *Utricularia vulgaris* L.

The distribution area for floating input plant was very small, and the floating input plant was mainly located in small patches or sporadic distribution in the bay. There were 2 kinds of floating input plant in the Poyang Lake, of which the most important and wide distribution was *Eichhornia crassipes* (Mart.) Solms, and the least distribution was *Hydrocharis dubia* (Bl.) Back.

The main habitat types of aquatic macrophytes in the Poyang Lake were summarized in Table 3. Among the amphibians, the coverage of the typical *Carex* spp. cluster was generally 40–80%, and the plant height was about 25–100 cm. But in the lower edge of the distribution area, the growth of the *Carex* spp. cluster was delayed, and the plant coverage and

Table 2: The aquatic plant's association types in the Poyang Lake.

Serial number	Plant community
1	*Vallisneria natans* (Lour.) Hara + *Carex* spp. + *Limnophila sessiliflora* (Vahl.) Bl. + *Ceratophyllum demersum* L.
2	*Carex* spp. + *Elymus dahuricus* Turcz. +*Vallisneria natans* (Lour.) Hara + *Myriophyllum tuberculatum* Roxburgh + *Nymphoides peltatum* (Gmel.) O. Kuntze + *Ceratophyllum demersum* L. + *Limnophila heterophylla* (Roxb.) Benth. + *Eleocharis tuberosa* (Roxb.) Roem. et Schult.
3	*Carex* spp.
4	*Alternanthera philoxeroides* (Mart.) Griseb. + *Paspalum distichum* L. + *Paspalum distichum* L.-*Alternanthera philoxeroides* (Mart.) Griseb.+*Typha angustifolia* L.-*Alternanthera philoxeroides* (Mart.) Griseb.
5	*Aeschynomene indica* Linn. + *Juncellus serotinus* + *Oryza rufipogon* Griff.
6	*Carex* spp. + *Polygonum* L.
7	*Marsilea quadrifolia* L.
8	*Paspalum distichum* L.
9	*Eichhornia crassipes* (Mart.) Solms + *Nymphoides peltatum* (Gmel.) O. Kuntze
10	*Trapa bispinosa* Roxb. + *Microcarpaea minima* (Koen.) Merr. + *Isachne globosa* (Thunb.) Kuntze + *Carex* spp.
11	*Eleocharis tuberosa* (Roxb.) Roem. et Schult. + *Ludwigia adscendens* (L.) Hara + *Elymus dahuricus* Turcz.-*Carex* spp. + *Myriophyllum spicatum* L.-*Hydrilla verticillata* (L.f.) Royle + *Hydrilla verticillata* (L.f.) Royle-*Vallisneria natans* (Lour.) Hara.
12	*Carex rhynchophysa* C. A. Mey.
13	*Blyxa japonica* (Miq.) Maxim. + *Juncellus serotinus* + *Sagittaria pygmaea* Miq.-*Hydrilla verticillata* (L.f.) Royle + *Monochoria vaginalis* (Burm.f.) Presl-*Rotala rotundifolia* (Buch.-Ham) Koehne.
14	*Eichhornia crassipes* (Mart.) Solms + *Nymphoides peltatum* (Gmel.) O. Kuntze + *Marsilea quadrifolia* L. + *Alternanthera philoxeroides* (Mart.) Griseb.-*Eichhornia crassipes* (Mart.) Solms.
15	*Hydrocharis dubia* (Bl.) Back.+ *Triarrherca sacchariflora* + *Carex* spp. + *Hydrilla verticillata* (L.f.) Royle + *Vallisneria natans* (Lour.) Hara + *Ceratophyllum demersum* L. + *Najas marina*-*Eleocharis tuberosa* (Roxb.) Roem. et Schult. + *Carex* spp.-*Triarrherca sacchariflora.*
16	*Vallisneria natans* (Lour.) Hara + *Nymphoides peltatum* (Gmel.) O. Kuntze + *Carex* spp. + *Limnophila heterophylla* (Roxb.) Benth. + *Ottelia alismoides* (L.) Pers. + *Trapa bispinosa* Roxb.-*Ottelia alismoides* (L.) Pers. + *Utricularia aurea* Lour. + *Hydrocharis dubia* (Bl.) Back.-*Trapa bispinosa* Roxb. + *Carex rhynchophysa* C. A. Mey. + *Trapella sinensis* Oliv. + *Sagittaria pygmaea* Miq. + *Eleocharis tuberosa* (Roxb.) Roem. et Schult.
17	*Carex* spp. + *Nymphoides peltatum* (Gmel.) O. Kuntze + *Eleocharis tuberosa* (Roxb.) Roem. et Schult. + *Zizania caduciflora* (Turcz. ex Trin.) H.-M. + *Elymus dahuricus* Turcz. + *Phragmites communis* Trin. + *Ceratophyllum demersum* L.
18	*Carex* spp.
19	*Elymus dahuricus* Turcz.
20	*Carex* spp. + *Eleocharis tuberosa* (Roxb.) Roem. et Schult. + *Nymphoides peltatum* (Gmel.) O. Kuntze + *Eleocharis tuberosa* (Roxb.) Roem. et Schult.-*Polygonum* L.
21	*Carex* spp. + *Polygonum* L. + *Eleocharis tuberosa* (Roxb.) Roem. et Schult.-*Polygonum* L. + *Carex* spp.-*Eleocharis tuberosa* (Roxb.) Roem. et Schult.
22	*Carex* spp. + *Polygonum* L.
23	*Carex* spp. + *Polygonum* L. + *Cynodon dactylon* (Linn.) Pers.
24	*Carex* spp. + *Elymus dahuricus* Turcz. + *Phragmites communis* Trin.-*Cynodon dactylon* (Linn.) Pers.
25	*Ceratophyllum demersum* L. + *Nymphoides peltatum* (Gmel.) O. Kuntze + *Imperata cylindrica* (Linn.) Beauv + *Carex* spp. + *Hydrilla verticillata* (L.f.) Royle + *Vallisneria natans* (Lour.) Hara + *Eleocharis tuberosa* (Roxb.) Roem. et Schult.
26	*Carex* spp.
27	*Carex* spp. + *Eleocharis tuberosa* (Roxb.) Roem. et Schult. + *Imperata cylindrica* (Linn.) Beauv.
28	*Carex* spp. + *Trapa bispinosa* Roxb. + *Ceratophyllum demersum* L. + *Utricularia aurea* Lour. + *Triarrhena lutarioriparia* + *Triarrherca sacchariflora.*
29	*Hydrilla verticillata* (L.f.) Royle + *Najas marina* + *Eleocharis tuberosa* (Roxb.) Roem. et Schult. + *Ceratophyllum demersum* L. + *Carex* spp. + *Nymphoides peltatum* (Gmel.) O. Kuntze + *Vallisneria natans* (Lour.) Hara.
30	*Najas marina* + *Trapa bispinosa* Roxb. + *Carex* spp. + *Zizania caduciflora* (Turcz. ex Trin.) H.-M. + *Triarrhena lutarioriparia.*
31	*Polygonum* L. + *Limnophila sessiliflora* (Vahl.) Bl. + *Utricularia vulgaris* L. + *Carex* spp. + *Eleocharis tuberosa* (Roxb.) Roem. et Schult. + *Elymus dahuricus* Turcz. + *Polygonum* L.

Table 3: The habitat profiles of main association types in macrophyte communities in the Poyang Lake.

Serial number	Cluster type	Sampling date	Multiplicity-cluster degree	Height (cm)	Number of plants	Coverage (%)
1	*Limnophila sessiliflora* (Vahl.) Bl.	2013/9/7	4, 3	15–25	82	60
		2013/9/14	3, 3	3-3	78	60
2	*Limnophila heterophylla* (Roxb.) Benth.	2013/9/7	5, 4	37–45	144	90
		2013/9/13	4, 4	3–10	135	50
3	*Carex* spp.	2013/9/7	4, 5	45–70	294	80
		2013/9/7	3, 3	50–80	170	40
		2013/9/8	5, 5	50–100	3600	100
		2013/9/13	4, 5	25–65	300	70
4	*Vallisneria natans* (Lour.) Hara	2013/9/7	5, 3	45–70	26	90
		2013/9/12	5, 3	15–30	20	90
5	*Eleocharis tuberosa* (Roxb.) Roem. et Schult.	2013/9/7	3, 3	20–50	850	30
		2013/9/11	5, 5	30–85	1000	90
		2013/9/14	1, 4	18–35	350	10
6	*Phragmites communis* Trin.	2013/9/7	2, 2	60–250	80	15
		2013/9/14	1, 1	56–70	36	10
7	*Zizania caduciflora* (Turcz. ex Trin.) H.-M.	2013/9/7	4, 5	60–120	260	60
		2013/9/13	5, 4	70–168	144	85
8	*Eleocharis tuberosa* (Roxb.) Roem.et Schult.-Polygonum L.	2013/9/8	3, 3	3–60	240	40
		2013/9/9	5, 5	25–49	1000	95
9	*Alternanthera philoxeroides* (Mart.) Griseb.	2013/9/9	5, 4	10–50	900	100
		2013/9/9	5, 5	42–70	672	95
10	*Paspalum distichum* L.-*Alternanthera philoxeroides* (Mart.) Griseb.	2013/9/9	4, 3	10–25	1600	70
11	*Typha angustifolia* L.-*Alternanthera philoxeroides* (Mart.) Griseb.	2013/9/9	4, 3	80–200	25	60
12	*Paspalum distichum* L.	2013/9/9	4, 5	32–55	600	80
		2013/9/10	4, 5	20–35	1000	100
13	*Aeschynomene indica* Linn.	2013/9/9	5, 3	180–200	23	85
14	*Oryza rufipogon* Griff.	2013/9/9	5, 5	70–120	160	100
15	*Juncellus serotinus*	2013/9/9	2, 1	90–120	23	30
		2013/9/10	5, 3	20–60	850	90
16	*Carex* spp.-*Eleocharis tuberosa* (Roxb.) Roem. et Schult.	2013/9/9	4, 3	43–79	640	60
17	*Polygonum* L.	2013/9/9	3, 4	4–10	2000	40
		2013/9/10	5, 5	20–45	1000	90
		2013/9/14	3, 3	20–72	57	60
18	*Cynodon dactylon* (Linn.) Pers.	2013/9/10	5, 5	6–10	420	90
19	*Marsilea quadrifolia* L.	2013/9/10	3, 2	15–25	368	40
20	*Eichhornia crassipes* (Mart.) Solms	2013/9/10	3, 2	45–55	21	80
		2013/9/11	5, 4	12–25	170	95
21	*Isachne globosa* (Thunb.) Kuntze	2013/9/10	5, 4	20–35	700	80
22	*Elymus dahuricus* Turcz.-*Carex* spp.	2013/9/10	2, 5	70–120	150	30
23	*Ludwigia adscendens* (L.) Hara	2013/9/10	5, 4	20–40	108	95
24	*Carex rhynchophysa* C. A. Mey.	2013/9/10	3, 3	100–150	750	50
25	*Sagittaria pygmaea* Miq.-*Hydrilla verticillata* (L.f.) Royle	2013/9/10	3, 3	10–17	23	40
26	*Phragmites communis* Trin.-*Cynodon dactylon* (Linn.) Pers.	2013/9/11	3, 3	30–90	95	40
27	*Nymphoides peltatum* (Gmel.) O. Kuntze	2013/9/11	4, 3	3–8	560	60
		2013/9/13	4, 3	35–55	30	60

Table 3: Continued.

Serial number	Cluster type	Sampling date	Multiplicity-cluster degree	Height (cm)	Number of plants	Coverage (%)
28	*Imperata cylindrica* (Linn.) Beauv	2013/9/11	5, 5	35–65	2560	90
		2013/9/12	5, 4	60–110	960	80
		2013/9/12	5, 4	60–110	960	80
29	*Triarrhena lutarioriparia*	2013/9/12	5, 4	70–165	650	85
		2013/9/13	5, 4	70–98	570	80
30	*Triarrherca sacchariflora*	2013/9/12	5, 5	80–103	200	95
31	*Hydrocharis dubia* (Bl.) Back.	2013/9/12	3, 3	10–15	30	50
32	*Ottelia alismoides* (L.) Pers.	2013/9/13	3, 4	15	20	80
		2013/9/13	4, 4	15–30	20	80
33	*Hydrocharis dubia* (Bl.) Back.-*Trapa bispinosa* Roxb.	2013/9/13	4, 4	10–15	96	70
34	*Carex rhynchophysa* C. A. Mey.	2013/9/13	4, 4	60–100	350	60
35	*Sagittaria pygmaea* Miq.	2013/9/13	5, 3	10–20	52	75
36	*Nephrolepis cordifolia* (L.) Presl	2013/9/14	5, 5	13–25	500	95
		2013/9/14	5, 5	13–20	400	90
37	*Sambucus chinensis* Lindl.	2013/9/14	2, 2	10–27	100	40
38	*Elymus dahuricus* Turcz.	2013/9/7	1, 2	10–30	340	10
		2013/9/11	3, 3	20–70	40	50
		2013/9/14	5, 4	62–131	280	80

height were relatively small. At the upper edge of the distribution area, with the increase of the elevation, the plant growth rate began to decline, and the coverage reduced accordingly. The plant height of the typical *Phragmites communis* Trin. cluster in the emerged plant was 56–250 cm, and the coverage was lower and was only 10–15%. The height of submerged plant cluster was 15–70 cm, and its coverage was large and up to 90%. The coverage of floating input plant cluster was about 50–95% and the plant height was 10–55 cm. The cluster coverage of floating-leaved plants was about 60% and plant height was 3–55 cm.

3.3. Quantitative Characteristics of Aquatic Plant Communities in the Poyang Lake. The number of aquatic plants in the lake area was 67, and the total of the aquatic plants' fresh weight was 1519.41 t. Among them, the *Carex* spp., *Eleocharis tuberosa* (Roxb.) Roem. et Schult., and emergent plants were 919.66 t, while the submerged plants and floating-leaved plants were up to 599.75 t. The distribution of amphibians accounted for 58.21% of the total number of samples, and there were 14.93%, 16.42%, and 10.45% of the total plants number for the emergent plants, submerged plants, and floating-leaved plants in the Poyang Lake, respectively. It could be found that the number of amphibians was the dominant plant species in the Poyang Lake, and their quantity and percentage of the total biomass were predominant, which were far more than the other three life forms (Table 4).

Among the 16 typical cluster types, the total biomass was the highest in the *Carex* spp. cluster, followed by the *Polygonum* L. cluster, the *Zizania caduciflora* (Turcz. ex Trin.) H.-M. cluster, and the *Imperata cylindrica* (Linn.) Beauv cluster

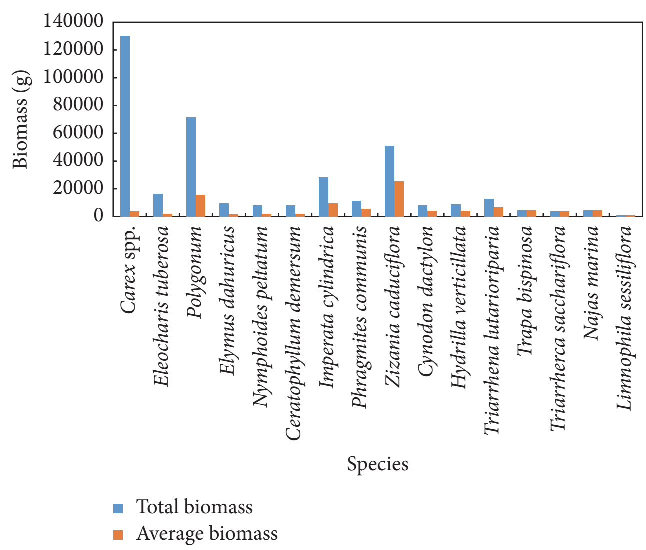

Figure 2: The total biomass and average biomass of the typical aquatic macrophyte's association types in the Poyang Lake.

(Figure 2). This was in contrast to 1998 aquatic vegetation survey [25], the difference was large, and the proportion of *Vallisneria natans* (Lour.) Hara and *Hydrilla verticillata* (L.f.) Royle was the largest, while the proportion of *Carex* spp. and *Polygonum* L. was very low. In the amphibians, *Carex* spp. cluster was the dominant species, and the distribution number was 34. In the emergent plants, *Zizania caduciflora* (Turcz. ex Trin.) H.-M. cluster, and *Phragmites communis*

Table 4: Number of the association area and standing crop (fresh weight) in the total vegetation area and standing crop in the Poyang Lake.

Serial number	Plant community	Quadrat number	Existing quantity (t)			
			Fresh weight	Fresh weight range	Dry weight	Dry weight range
1	*Imperata cylindrica* (Linn.) Beauv	3	38.33	35–45	10.26	9.18–10.8
2	*Eleocharis tuberosa* (Roxb.) Roem. et Schult.	6	94.08	11.5–380	6.285	4.62–11.34
3	*Eleocharis tuberosa* (Roxb.) Roem. et Schult.-*Polygonum* L.	2	137.5	75–200	21.345	20.65–22.04
4	*Najas marina*	1	35	—	25.53	—
5	*Triarrherca sacchariflora*	1	40	—	9.32	—
6	*Cynodon dactylon* (Linn.) Pers.	1	40	—	5.8	—
7	*Zizania caduciflora* (Turcz. ex Trin.) H.-M.	2	99.76	9.52–190	104.08	6.46–201.7
8	*Hydrilla verticillata* (L.f.) Royle	2	42.5	35–50	6.195	3.61–8.78
9	*Utricularia aurea* Lour.	1	50	—	15.28	—
10	*Elymus dahuricus* Turcz.	3	148.05	14.15–340	23.86333	5.99–55
11	*Ceratophyllum demersum* L.	4	80	35–150	6.7775	3.07–10.08
12	*Vallisneria natans* (Lour.) Hara	2	29.75	4.5–55	3.86	3.24–4.48
13	*Utricularia vulgaris* L.	1	25	—	3.6	—
14	*Polygonum* L.	6	34.11333	10.34–50	6.218333	4.6–9.71
15	*Trapa bispinosa* Roxb.	2	57.5	55–60	6.35	2.93–9.77
16	*Phragmites communis* Trin.	1	120	—	15.43	—
17	*Phragmites communis* Trin.-*Cynodon dactylon* (Linn.) Pers.	1	20	—	9.4	—
18	*Triarrhena lutarioriparia*	1	27.03	—	11.73	—
19	*Limnophila sessiliflora* (Vahl) Blume	1	12.8	—	4.29	—
20	*Carex* spp.	20	72.992	24.84–360	8.7425	4.22–16.96
21	*Carex* spp.-*Eleocharis tuberosa* (Roxb.) Roem. et Schult.	1	35	—	8.53	—
22	*Nymphoides peltatum* (Gmel.) O. Kuntze	5	280	20–280	43.4	3.3–43.4

Trin. cluster distribution were in large quantities, each for 2. In the submerged plants, the distribution of *Ceratophyllum demersum* L. cluster was the largest, with 4 species. In the floating-leaved plants, the distribution of *Nymphoides peltatum* (Gmel.) O. Kuntze cluster was the largest, with 4 species. This pattern of distribution was associated with its reproductive strategies. The vast majority of grassland vegetation in the Poyang Lake interrupted their breeding during the summer season, which was suitable for plant growth during the flood season. Therefore, the plant with vigorous regenerative ability, for example, *Carex* spp., had become the most dominant species in the Poyang Lake grassland vegetation and *Carex* spp. were submerged by flood and became the carp and crucian carp spawning attachment. cluster in Poyang Lake was not only beneficial to grazing firewood and green manure, but also conducive to the reproduction of grass-laying fish sand the growth of herbivorous fish. The production of carp and crucian carp in the fishery of the Poyang Lake was about half of the total output, which was closely related to the seasonal succession of *Carex* spp. [26].

4. Conclusions

In this study, the distribution of aquatic vegetation and associated community diversity and biomass in the Poyang Lake were investigated with the aim of getting information for the monitoring and protection of aquatic plants in the Poyang Lake. Some interesting conclusions were obtained as follows:

(1) Nine species of aquatic plants were found in the Poyang Lake and *Carex* spp. dominated absolutely with multi grads 58 in the Poyang Lake. Most of the aquatic plant species were amphibians and a small number of the species were found on the beach floating plants or submerged plants or emergent plants.

(2) The vegetation of the Poyang Lake scattered in different areas which could be divided into 31 major plant communities and 5 plant zones including amphibian, emergent, floating-leaved, submerged, and floating input. The majority of aquatic vegetation distribution area in the Poyang Lake was *Carex* spp. cluster.

(3) There were 67 aquatic plants in the Poyang Lake, and the total of the aquatic plants' fresh weight was 1519.41 t. Moreover, it could be found that the amphibians were the dominant plant species in the Poyang Lake.

Competing Interests

The authors declare that they have no competing interests.

Authors' Contributions

Wei Du, Ziqi Li, and Zengxin Zhang analyzed the data, drew the figures, and finished the draft of the manuscript. Qiu Jin, Xi Chen, and Shanshan Jiang participated in the writing of this manuscript. All authors read and approved the final manuscript.

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References

[1] H. Jun, G. Xiao-hong, and L. Guo-feng, "Aquatic macrophytes in East Lake Taihu and its interaction with water environment," *Journal of Lake Sciences*, vol. 20, no. 6, pp. 790–795, 2008.

[2] J. Luo, X. Li, R. Ma et al., "Applying remote sensing techniques to monitoring seasonal and interannual changes of aquatic vegetation in Taihu Lake, China," *Ecological Indicators*, vol. 60, pp. 503–513, 2016.

[3] S. Guan, Q. Lang, and B. Zhang, "Aquatic vegetation of Poyang Lake," *Acta Hydrobiologica Sinica*, no. 1, pp. 9–21, 1987 (Chinese).

[4] C. Lan, Y. Shen, B. Wang et al., "Investigation of aquatic plants and benthic macroinvertebrates of lakes in Inner Mongolia-Xinjiang Plateau," *Journal of Lake Sciences*, no. 6, pp. 888–893, 2010 (Chinese).

[5] C. Mânzatu, B. Nagy, A. Ceccarini, R. Iannelli, S. Giannarelli, and C. Majdik, "Laboratory tests for the phytoextraction of heavy metals from polluted harbor sediments using aquatic plants," *Marine Pollution Bulletin*, vol. 101, no. 2, pp. 605–611, 2015.

[6] S. Rezania, S. M. Taib, M. F. Md Din, F. A. Dahalan, and H. Kamyab, "Comprehensive review on phytotechnology: heavy metals removal by diverse aquatic plants species from wastewater," *Journal of Hazardous Materials*, vol. 318, pp. 587–599, 2016.

[7] P. Massicotte, A. Bertolo, P. Brodeur, C. Hudon, M. Mingelbier, and P. Magnan, "Influence of the aquatic vegetation landscape on larval fish abundance," *Journal of Great Lakes Research*, vol. 41, no. 3, pp. 873–880, 2015.

[8] R. T. J. Churchill, M. L. Schummer, S. A. Petrie, and H. A. L. Henry, "Long-term changes in distribution and abundance of submerged aquatic vegetation and dreissenid mussels in Long Point Bay, Lake Erie," *Journal of Great Lakes Research*, vol. 42, no. 5, pp. 1060–1069, 2016.

[9] G. Shen, J. Liao, H. Guo, and J. Liu, "Poyang Lake wetland vegetation biomass inversion using polarimetric RADARSAT-2 synthetic aperture radar data," *Journal of Applied Remote Sensing*, vol. 9, no. 1, 2015.

[10] J. Liao, G. Shen, and L. Dong, "Biomass estimation of wetland vegetation in Poyang Lake area using ENVISAT advanced synthetic aperture radar data," *Journal of Applied Remote Sensing*, vol. 7, no. 1, Article ID 12215, 2013.

[11] X. Han, X. Chen, and L. Feng, "Four decades of winter wetland changes in Poyang Lake based on Landsat observations between 1973 and 2013," *Remote Sensing of Environment*, vol. 156, pp. 426–437, 2014.

[12] R. J. Daoust and D. L. Childers, "Quantifying aboveground biomass and estimating net aboveground primary production for wetland macrophytes using a non-destructive phenometric technique," *Aquatic Botany*, vol. 62, no. 2, pp. 115–133, 1998.

[13] T. S. F. Silva, M. P. F. Costa, and J. M. Melack, "Assessment of two biomass estimation methods for aquatic vegetation growing on the Amazon Floodplain," *Aquatic Botany*, vol. 92, no. 3, pp. 161–167, 2010.

[14] D. Zhao, H. Jiang, T. Yang, Y. Cai, D. Xu, and S. An, "Remote sensing of aquatic vegetation distribution in Taihu Lake using an improved classification tree with modified thresholds," *Journal of Environmental Management*, vol. 95, no. 1, pp. 98–107, 2012.

[15] L. Zhang, J. Yin, Y. Jiang, and H. Wang, "Relationship between the hydrological conditions and the distribution of vegetation communities within the Poyang Lake National Nature Reserve, China," *Ecological Informatics*, vol. 11, pp. 65–75, 2012.

[16] X. Wang, L. Xu, R. Wan, and Y. Chen, "Seasonal variations of soil microbial biomass within two typical wetland areas along the vegetation gradient of Poyang Lake, China," *Catena*, vol. 137, pp. 483–493, 2016.

[17] S. Guan, Q. Lang, and B. Zhang, "Biomass of macrophytes of the Poyang Lake with suggestions of its rational exploitation," *Acta Hydrobiologica Sinica*, no. 3, pp. 219–227, 1987 (Chinese).

[18] Y. Jian, R. Li, J. Wang, and J. Chen, "Aquatic plant diversity and remote sensing of the beach vegetation in lake Poyang," *Acta Phytoecologica Sinica*, vol. 25, no. 5, pp. 581–642, 2001 (Chinese).

[19] Y. Peng, Y. Jian, and R. Li, "Community diversity of aquatic plants in the lakes of Poyang Plain District of China," *Journal of Central South Forestry University*, no. 4, pp. 22–27, 2003 (Chinese).

[20] Z. Zhang, X. Chen, C.-Y. Xu, Y. Hong, J. Hardy, and Z. Sun, "Examining the influence of river-lake interaction on the drought and water resources in the Poyang Lake basin," *Journal of Hydrology*, vol. 522, pp. 510–521, 2015.

[21] H. Vereecken, J. Baetens, P. Viaene, F. Mostaert, and P. Meire, "Ecological management of aquatic plants: effects in lowland streams," *Hydrobiologia*, vol. 570, no. 1, pp. 205–210, 2006.

[22] C. D. K. Cook, *Water Plants of the World*, Springer Science & Business Media, Bath, UK, 1974.

[23] H. Chen, "The structure and dynamicsofaquatic vascular plant communities in DongLake, Wuhan," *Oceanologia et Limnologia Sinica*, no. 3, pp. 275–284, 1980 (Chinese).

[24] W. Hua, C. Shuiping, C. Peihong et al., "Investigation of aquatic macrophytes in lakes of Beihu Watershed of East Lake area in Wuhan, in the autumn of 2009," *Journal of Lake Sciences*, vol. 23, no. 3, pp. 401–408, 2011.

[25] W. Li, G. Liu, B. Xiong et al., "The restoration of aquatic vegetation in lakes of Poyang Lake nature reserve after catastrophic flooding in 1998," *Journal of Wuhan Botanical Research*, no. 4, pp. 301–306, 2004 (Chinese).

[26] B. Zhang and J. Wang, "Preliminary opinions on the protection and utilization of natural resources in Poyang Lake," *Freshwater Fisheries*, no. 3, pp. 1–5, 1982 (Chinese).

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Research Article

Long-Term Study of the Relationship between Precipitation and Aquatic Vegetation Succession in East Taihu Lake, China

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Based on long-term rainfall measurements (1956–2012), water level records (1956–2006), and aquatic plants field survey data (1960–2014), the relationship between precipitation and aquatic vegetation succession in east Taihu Lake, China, is studied. Neither abrupt changes nor any trends were found in the annual rainfall series in Taihu Lake during the studied period (1956–2012). However, for seasonal variations, statistically significant decreases are found in spring and autumn, while the rainfall in winter exhibits statistically significant increase. No significant trend was obtained in summer. A "dry" period was detected in our studied period (1963/1964~1978/1979). Total annual rainfall was significantly positively correlated to the number of rain-days ($r = 0.59$) and the water level ($r = 0.84$). Our results indicate that the variations of rainfall and water level may have an impact on the aquatic plants in Taihu Lake. The dry period may be not suitable for the growth of the aquatic plants. All aquatic plants in Taihu Lake were dramatically reduced in the dry period, especially for submerged macrophytes and floating-leaf macrophytes. Our results may be helpful for the aquatic restoration in the future.

1. Introduction

The formation, development, and maintenance of wetland ecosystems are closely related to the water resources [1, 2]. The excessive utilization of water resources directly leads to the dryness of wetlands and eventually the degradation of natural wetlands [3, 4]. Therefore, overall planning of water resources is important for the stable and healthy operation of natural/restored wetlands [5]. In order to fully understand and effectively manage the wetland ecosystems, data accumulations (including both physical and biological variables) are required in various scales and under contrasting climatic and biophysical condition [6–8]. However, researches in relationships between long-term observations of water dynamics (e.g., precipitation and/or water level) and plant vegetation responses are relatively rare, especially in humid lands such as wetlands.

As a functional group, aquatic plants are one of the most important components in aquatic ecosystems [9, 10]. Aquatic plants are the main producers of oxygen, the basis of the food chain and a major part of the energy flow in the aquatic ecosystem [11]. The plants form a complex spatial habitat that provides food and sanctuary for organisms living in and around, stabilizes sediments and participates in matter cycling, and regulates wetland hydrological conditions [12]. Many recent researches have pointed out that the response of aquatic plants, especially submerged macrophytes, is associated with the changes in water levels [13–15]. Several models have been developed to predict vegetation responses as a function of annual, seasonal, or monthly rainfall or of rainfall-related indices [16–18]. Unfortunately, most of the current studies related to precipitation are from terrestrial ecosystem. The effect of long-term precipitation on aquatic vegetation has not been fully understood in most cases due to the lack of sufficient data to conduct such long-term assessments.

Since the beginning of the industrial era, anthropogenic greenhouse gas emissions (largely driven by economic and population growth) and the increasing greenhouse gas concentrations in the atmosphere have led to various changes in the Earth's climate at an unprecedented rate [19]. According to the IPCC fifth report [20], global warming is an indisputable fact. The global average surface temperature (including land and ocean) has shown a warming temperature of 0.85°C (0.65

to 1.06°C) over the period 1880 to 2012, and the increase in temperature is likely to continue and even intensify [20]. Climate change may lead to a large variation in interannual and intra-annual rainfall [21], plus extreme weather/climate events (e.g., droughts, floods, and extreme precipitation), which would affect the structure and function of global ecosystems and the sustainable development of many human systems.

Global climate change will directly lead to changes in precipitation and the rainfall imbalance. In general, precipitation will be decreased in the low latitude areas, while it will be increased significantly in the high latitude regions [22]. However, there is a certain degree of uncertainty in global climate model. For instance, Piao et al. [23] analyzed data from 355 precipitation monitoring sites in China since 1960. It is found that there is no significant change in the total rainfall over the long-term period in China. But significant regional precipitation trends are found. The rainfall in drier regions of northeastern China is decreasing in summer and autumn. In contrast, the wetter area of southern China is receiving more rainfall in both summer and winter. Such trends may result in droughts in the northern part of China and flooding in the southern part of China, which will significantly change the coverage and species composition of wetland vegetation and soil physical and chemical characteristics [24].

Taihu Lake is the third-largest freshwater lake in China, occupying a surface area of 2,425 km^2 and the average depth is about 1.9 m. The lake is located in the core of the Yangtze Delta within the lower reaches of the Yangtze River Basin, which is the most developed areas in China. Due to the rapid socioeconomic development, the aquatic ecosystem of Taihu Lake has degraded. The distribution and community structure of aquatic vegetation have clearly been changed. The main aim of this paper is to investigate the relationship between rainfall patterns and aquatic vegetation-related processes based on the point of view of plant ecology, by using the long-term (1956–2012) rainfall records from the weather station in Taihu Lake, Dongshan (Suzhou, Jiangsu province, China).

2. Data and Methods

The long-term observations used in this paper are from the Dongshan weather station, which is freely available from the National Meteorological Information Center (http://data.cma.cn/). The Dongshan weather station (31°4′N, 120°26′E) is located near east Taihu bay (Figure 1), with the altitude of 17.5 m above sea level. The data ranges from 1956 to 2012. For each measurement, meteorological variables such as pressure, temperature, relative humidity, rainfall, and winds are recorded. Regarding the rainfall record, it measured one-day precipitation depth from 20 LTC (local time) to 20 LTC. Besides the rainfall data, the data of annual water level are also used in this paper, which is from 1956 to 2006. The average annual temperature at Dongshan station is 16.5°C, the average total annual rainfall is 1127.7 mm (1956–2012), and the average annual water level of Taihu Lake is about 3.09 m (1956–2006).

The monthly rainfall values, which are calculated from daily rainfall data, are used to obtain the seasonal and annual rainfall values. Regarding the rain-day, when rainfall volume was ≥1 mm, it was considered to be a rain-day. Annual and monthly rain-day averages were then calculated. Following the method described in Lázaro et al. [25], rainfall was considered "normal" between the first (25th) and the third (75th) quartile, under 25th as "dryer" than normal and over 75th "wetter" than normal (Table 1). The 10th and 90th percentiles were considered thresholds for extreme values. This criterion is commonly followed to characterize "very dry," "dry," "normal," "rainy," and "very rainy" climatic conditions.

Table 1: Annual and seasonal statistical summary from monthly rainfall series (1956–2012) at Dongshan station.

Parameter	Annual	Winter	Spring	Summer	Autumn
Average (mm)	1127.7	145.9	303.8	461.4	216.7
Variance (mm^2)	43171.9	4507.9	6210.3	31264.4	8768.7
SD (mm)	207.8	67.1	78.8	176.8	93.6
Minimum (mm)	680.1	27.1	124.3	103.4	78.4
10th percentile (mm)	860.1	65.3	214.0	297.4	122.7
25th percentile (mm)	999.6	90.0	246.7	341.4	154.6
50th percentile (mm)	1101.3	136.3	300.2	432.3	201.0
75th percentile (mm)	1252.7	196.6	364.7	549.0	249.8
90th percentile (mm)	1402.5	255.0	393.5	694.1	347.6
Maximum (mm)	1699.7	297.0	488.3	1163.8	536.6

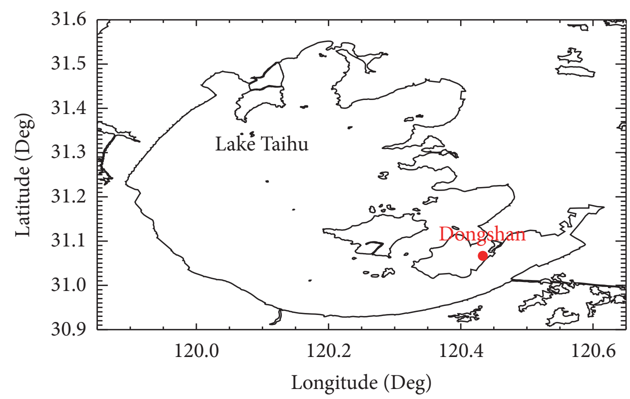


Figure 1: Location of the Dongshan weather station.

The cumulative sums of deviations are calculated to investigate the possible nonabrupt changes in the long-term rainfall records and differentiate wet and dry periods. It can be calculated as follows:

$$S_k = \sum_{i=1}^{k} d_i, \quad k = 1, 2, \ldots, n. \quad (1)$$

A change in the slope of S_k denoted a climatic change in the long-term series. The occurrence of the maximum $|S_k|$ represented a change point. The advantage of this method is that it can detect the invisible changes with strong variability in rainfall or runoff series [26, 27].

To investigate the possible existence of abrupt changes and trends, two methods are adopted in this paper. The first one is the sequential version of the Mann–Kendall test proposed by Sneyers [28], which is applied for the annual

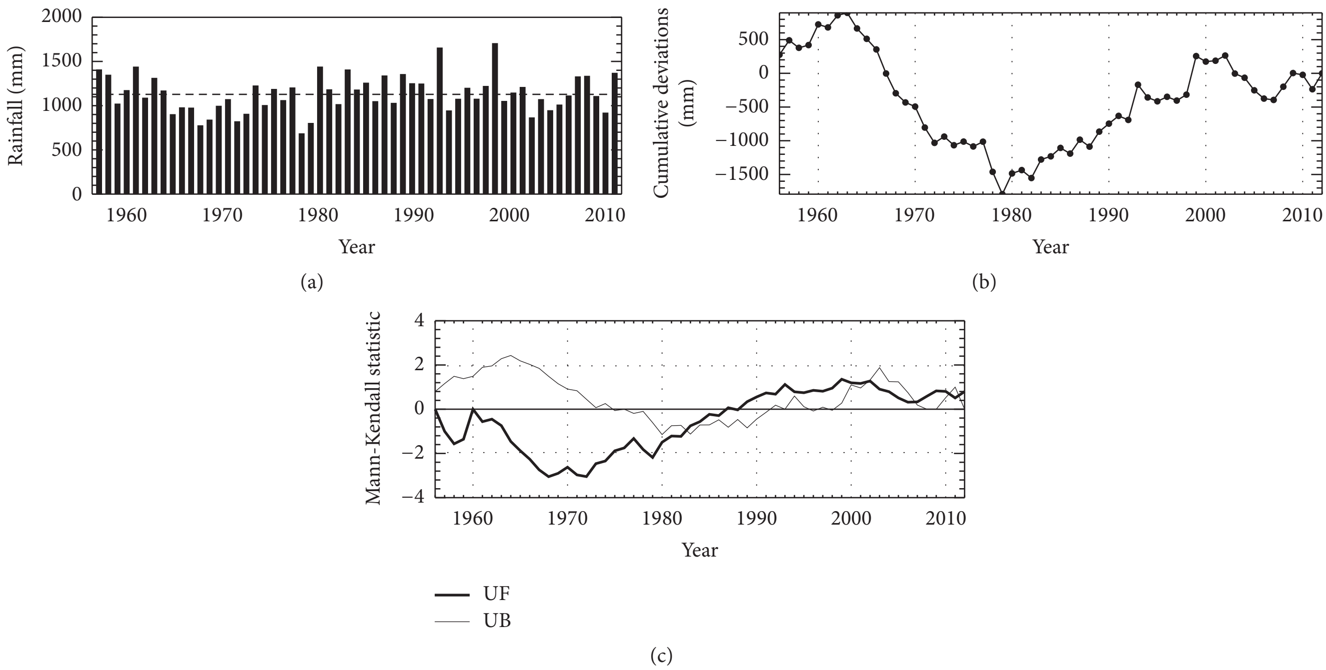


Figure 2: (a) Time series of total annual rainfall at Dongshan weather station (1956–2012). The horizontal line shows the average rainfall throughout the studied period. (b) Cumulative sum of deviations from the average rainfall. (c) Results of the Mann–Kendall test. The solid line is the UF statistic and the thin line is the retrograde series UB.

series. The Mann–Kendall test is a rank-based nonparametric method. It is thought to be suitable for data not with a normal distribution, which is usually used in rainfall series. It is based on the following procedure. Considering a time series of length n (x_i, where $i = 1, 2, \ldots, n$), for each x_i, the number r_i of $x_i > x_j$ with $j < i$ is computed. Then, the sum is calculated:

$$s_k = \sum_{i=1}^{k} r_i. \quad (2)$$

This sum presents a normal distribution with an average (E) and variance (Var):

$$E(s_k) = \frac{k(k-1)}{4},$$
$$\operatorname{var}(s_k) = \frac{k(k-1)(2k+5)}{72}, \quad (3)$$
$$k = 2, 3, \ldots, n.$$

Then, the UF statistic is calculated as follows:

$$\mathrm{UF}_k = \frac{[s_k - E(s_k)]}{\sqrt{\operatorname{var}(s_k)}}, \quad k = 1, 2, \ldots, n \quad (4)$$

and UF_k is compared with a standard normal distribution at the required level of significance, which is usually $\alpha = 0.05$ [28, 29]. The null hypothesis is rejected when $|\mathrm{UF}_k| > 1.96$. $\mathrm{UF}_k > 0$ indicates an increase and $\mathrm{UF}_k < 0$ indicates a decrease.

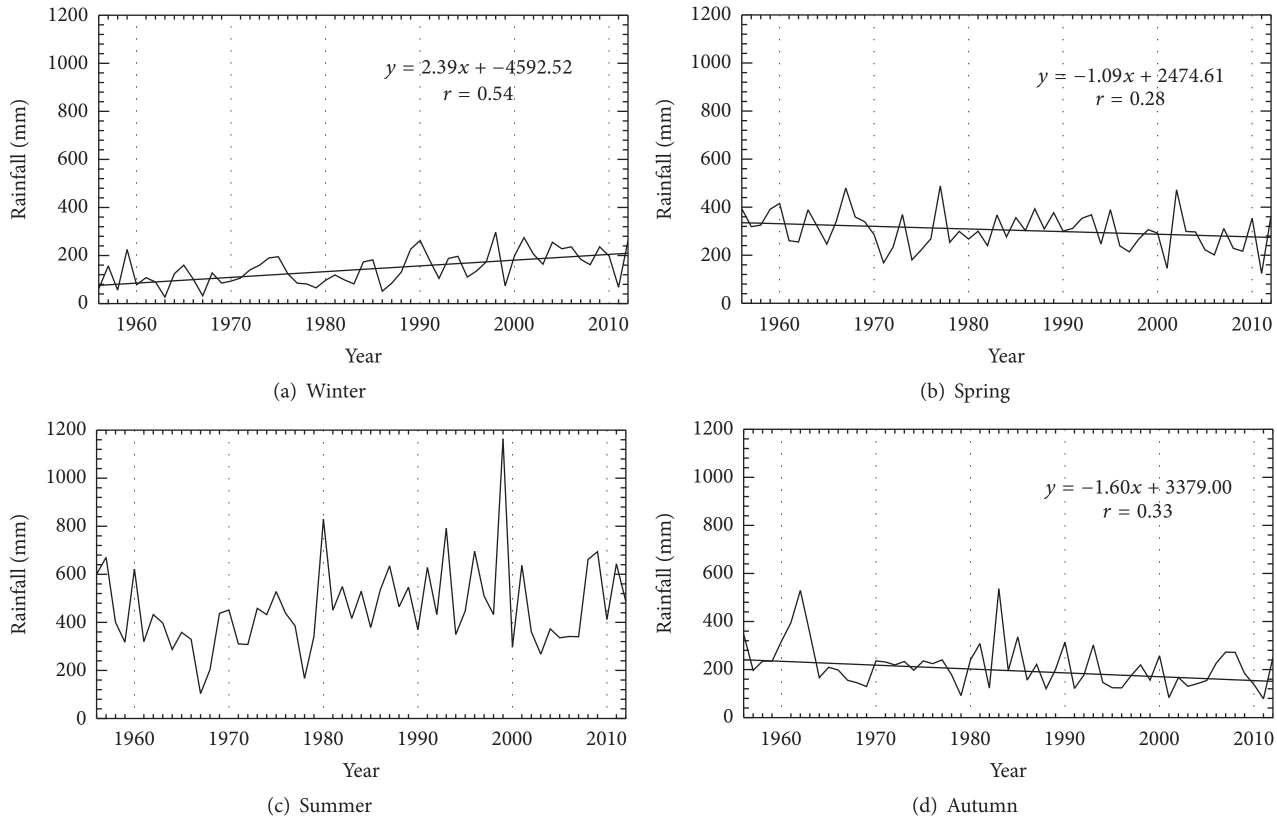
By using the retrograde series of x_i and also going through the whole procedure above, the UB_k statistic is obtained. If curve UF_k or UB_k passes through the 5% significance level from inside outwards, it is considered that the trend is significant. In addition, if curves UF_k and UB_k clearly cross each other (not overlap) between the critical values at the 5% level, there is an abrupt change, with the intersection point representing the beginning of that change.

The second trend estimate approach was computed for each of the four three-month seasons (December-January-February, DJF, etc.) using data for the period 1956–2012. Trends were computed using the nonparametric median of pairwise slopes method [30], with statistical significance levels based on Spearman rank-order tests.

There has been extensive amounts of aquatic vegetation field survey data collected from east Taihu Lake. In the current study, we collected a series of published field survey papers and reports covering aquatic vegetation surveys from 1960 to 2014 [31–36] (all data was published in Chinese). Aquatic vegetation surveys undertaken for this current study (from 1960 to 2014) covered (1) species number at the survey sites (east Taihu Lake) and (2) vegetative production (g/m^2) of each type of vegetation (i.e., submerged macrophytes, floating-leaf macrophytes, and emerged macrophytes).

3. Results and Discussion

According to Figure 2(a), the maximum annual rainfall at studied station (1649.9 mm) occurred in 1999, while the



(a) Winter (b) Spring (c) Summer (d) Autumn

FIGURE 3: Time series of seasonal rainfall during the period 1956–2012. Linear trend lines are plotted for series with trends significant at 95% confidence level or greater.

minimum rainfall (680.1 mm) happened in 1978. The cumulative sums of deviations detected 3 periods (Figure 2(b)): 1956~1962/1963, 1963/1964~1978/1979, and 1979/1980~2012, which had average rainfalls of 1239.8, 959.7, and 1182.0 mm, respectively. Using percentiles (Table 1), the second period is classified as "dry" and the other two periods as "normal." There is a slightly decreasing trend until approximately 1979, and a slight increase after that year (Figure 2(c)). Since 1980, the two statistics are around 0, which indicates that annual rainfall after 1980 fluctuated around the average value. In addition, the curve UF overlaps curve UB, which means that neither abrupt changes nor any trends were found in the annual rainfall series during the studied period (1956–2012).

The rainfall at the studied station shows an apparent seasonal variation (Figure 3). The peak rainy season is mainly summer, with the average depth 461.4 mm. The winter is relatively dry, with only about 145.9 mm in all three months. Table 1 summarizes the statistical results of annual and seasonal rainfall values. The main contribution to total annual rainfall comes from spring and summer values. Statistically significant decreases are found in spring and autumn in past several decades. On the other hand, the rainfall in winter exhibits statistically significant increase during 1956–2012. For summer, no significant trend was obtained. The maximum summer rainfall occurred in 1999, with a value 1163.8 mm, which is comparable to the average annual rainfall.

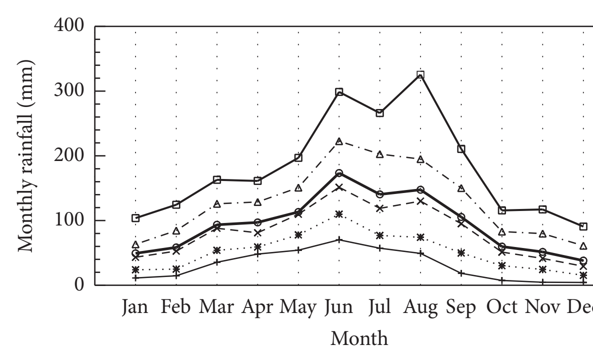


FIGURE 4: The percentiles of monthly rainfall are plotted as 10th (plus solid line), 25th (asterisk dot line), 50th (× dash line), 75th (triangle dash dot line), and 90th (square solid line). Average monthly rainfall is shown as circle thick solid line.

According to the monthly percentiles (Figure 4), monthly distribution of rainfall showed a maximum in June-August and minimum in December and January. Table 2 provides the complete monthly statistical data for each month. In summer time, the 75th and 90th percentiles had the highest rainfall volumes, which implies that these three months have a greater probability of having the highest rainfall values. Interestingly, the maximum rainfall occurs mostly in June, except for 90th percentile. This indicates that the extreme rainfalls occurred

Table 2: Statistical summary of monthly rainfall data at Dongshan (1956–2012).

Parameter	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Average (mm)	49.3	58.5	93.5	97.1	113.2	173.4	140.4	147.5	105.5	59.7	51.5	38.1
Variance (mm^2)	1361.6	1484.7	2366.1	2051.2	2392.2	11454.6	6472.8	8799.5	5926.2	2259.3	1460.7	875.5
SD (mm)	36.9	38.5	48.6	45.3	48.9	107.0	80.5	93.8	77.0	47.5	38.2	29.6
Minimum (mm)	0.0	0.0	14.9	30.4	24.5	23.6	19.1	0.1	6.5	0.0	1.3	0.0
10th percentile (mm)	11.2	14.6	35.5	48.2	54.2	70.2	57.2	49.2	18.4	7.4	4.6	4.3
25th percentile (mm)	23.9	24.9	53.9	59.1	77.8	109.8	76.9	74.1	49.8	30.2	24.4	15.3
50th percentile (mm)	43.3	52.8	88.0	81.1	109.7	151.4	118.8	130.0	95.1	51.2	41.4	29.3
75th percentile (mm)	63.3	84.5	125.9	128.7	150.9	222.7	202.7	195.0	150.2	82.9	79.8	61.1
90th percentile (mm)	103.7	124.4	162.9	161.2	196.9	298.7	266.1	325.4	210.6	115.6	117.1	90.7
Maximum (mm)	189.7	146.5	246.0	242.3	213.3	696.6	361.8	417.2	377.2	279.7	147.5	118.5

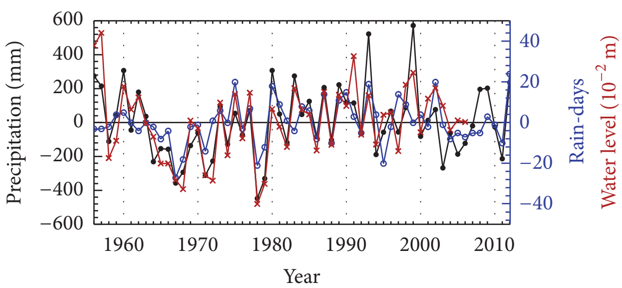


Figure 5: Relationship among the anomalies of annual rainfall (black solid circle), annual rain-days (blue open circle), and annual water level of Taihu Lake (red ×).

more often in August. The "normal" values (between 25th and 75th percentiles) for June, July, and August were 109.8–222.7 mm, 76.9–202.7 mm, and 74.1–195.0 mm, respectively. In winter, the "normal" values were between 15.3 and 61.1 mm for December and between 23.9 and 63.3 mm for January. The median rainfalls are to be lower than the averages in all cases. The biases are larger in summer than those in other seasons.

The annual average of rain-days at Dongshan station was 94. According to the relationship between the anomalies of annual rainfall and annual rain-days (Figure 5, black and blue curves), the number of rain-days was significantly positively correlated to total annual rainfall (Spearman $r = 0.59, n = 57, p < 0.001$). In dry period (1963/1964~1978/1979), the rain-days matched with the rainfall volume quite well. A decrement in rainfall depth usually corresponded with a decrease in number of rain-days. However, for those "normal" periods, annual rainfall and annual rain-days did not show this strong relationship. The rain-days remained low in spite of the high precipitation (such as, 1999, 2008, and 2009).

The variation of annual water level of Taihu Lake is significantly positively correlated with the annual rainfall (Figure 5, black and red curves). The correlation coefficient between them is 0.84 ($n = 51, p < 0.001$). It is reported that the changes of water level may limit the time for plastic responses of morphological and/or physiological traits in aquatic plants. In addition, if the water level change results in high level of disturbance, it may cause physical damage to the plants (e.g., stolon breakage, uprooting, or other damage, [37]), which eventually bring adverse impacts on plant growth [38, 39].

Since 1960, a total of 35 species of aquatic plants disappeared from Taihu Lake. The maximum total species number was observed in 1997 (75 species); however, only 40 species was observed in the most recent (2014) field survey (Figure 6(a)). The extinction species, such as *Potamogeton macckianus*, used to be widely distributed in the middle and lower Yangtze River. In the Taihu Lake, the most produced vegetation type is the emergent species, including *Phragmites australis* and *Zizania caduciflora*. Both species' biomass (g/m^2) increased from 1960 to 1997; however, it decreased from 1997 to 2014 (Figure 6(c)). Similarly, the biomass of submerged macrophytes was also increased from 1960 to 1997 and decreased from 1997 to 2014 (Figure 6(b)). The species compositions were more complicated than emergent species. In 1980, the dominant submerged macrophytes were *Potamogeton malaianus*, *Vallisneria spiralis*, and *Hydrilla verticillata*; in 1996, the dominant submerged macrophytes were *Potamogeton macckianus*, *Elodea nuttallii* (exotic species), *Vallisneria spiralis*, and *Hydrilla verticillata*; in 2002, the dominant submerged macrophytes were *Elodea nuttallii*, *Ceratophyllum demersum*, and *Potamogeton macckianus*. For the floating-leaf species, there was a significant increase from 1960 to 2002 and a decrease from 2002 to 2014 (Figure 6(d)). The dominant species were *Nymphoides peltatum*, *Nymphoides indica*, and *Trapa bicornis*.

Previous studies [40, 41] revealed that eutrophication and its consequences of turbid water had led to the disappearance of aquatic vegetation in Taihu Lake. Similarly, our results show that since 2002, with the process of eutrophication, aquatic vegetation in east Taihu Lake significantly decreased in its biomass and species biodiversity. From a historical point of view, 1980 is the turning point of aquatic vegetation change in east Taihu Lake. From 1960 to 1980, the biomass of submerged macrophytes and floating-leaf macrophytes were relatively small compared with those in other periods. According to the investigation in long-term rainfall and water level measurements, sharp jumps were found in both 1960 and 1980, especially for the year 1980. In addition, based on Figure 2(b), the period 1960–1980 was classified as a "dry" period, in which all aquatic plants in Taihu Lake were dramatically reduced, especially for submerged macrophytes

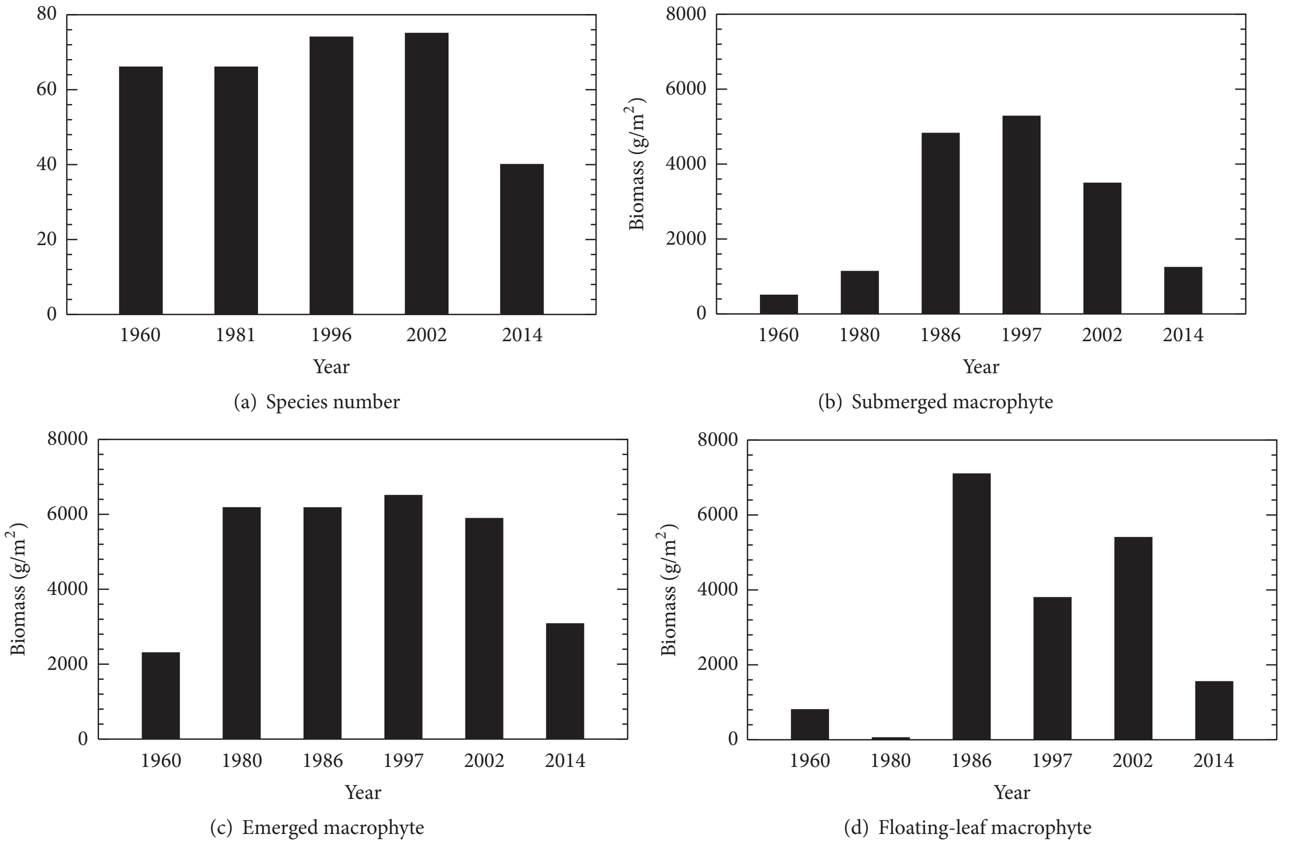


Figure 6: Species number (a), submerged macrophytes biomass (b), emerged macrophytes biomass (c), and floating-leaf macrophytes biomass (d) in east Taihu Lake from 1960 to 2014.

and floating-leaf macrophytes. These indicate that the variations of rainfall and water level may have an impact on the growth of the aquatic plant in Taihu Lake. The dry period may be not suitable for the growth of the aquatic plant. It is noteworthy that the comprehensive lake survey is relatively rare and more importantly most of these survey data were obtained in summer. Therefore, seasonal results about the aquatic vegetation in east Taihu Lake cannot be obtained from the current available data. In addition, it should be aware that there are necessary delays in the response of the aquatic vegetation growth to the water level changes. More detailed investigation in relationship between aquatic vegetation and the precipitation/water level is needed.

4. Conclusion

In this paper, the relationship between precipitation and aquatic vegetation succession in east Taihu Lake, China, is investigated by using long-term rainfall measurements (1956–2012), water level records (1956–2006), and also aquatic plant survey data from 1960 to 2014. The main findings are as follows:

(1) Neither abrupt changes nor any trends were found in the annual rainfall series in Taihu Lake during the studied period (1956–2012). However, for seasonal variations, statistically significant decreases are found in spring and autumn, while the rainfall in winter exhibits statistically significant increase. No significant trend was obtained in summer. A "dry" period was detected in our studied period (1963/1964~1978/1979). The other years of the studied period are classified as "normal" period.

(2) Total annual rainfall was significantly positively correlated to the number of rain-days ($r = 0.59$) and the water level ($r = 0.84$). However, the strong correlation between annual rainfall and rain-days only occurred in dry period. In those normal period, it is usually found that the rain-days remained low in spite of the high precipitation. Regarding the water level, the strong correlation exists throughout the studied period.

(3) Our results indicate that the variations of rainfall and water level may have an impact on the aquatic plant in Taihu Lake. The dry period may be not suitable for the growth of the aquatic plant. All aquatic plants in Taihu Lake were dramatically reduced in the dry period, especially for submerged macrophytes and floating-leaf macrophytes.

Competing Interests

No competing interests are declared.

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References

[1] J. L. Nel, D. J. Roux, R. Abell et al., "Progress and challenges in freshwater conservation planning," *Aquatic Conservation: Marine and Freshwater Ecosystems*, vol. 19, no. 4, pp. 474–485, 2009.

[2] S. Linke, E. Turak, and J. Nel, "Freshwater conservation planning: the case for systematic approaches," *Freshwater Biology*, vol. 56, no. 1, pp. 6–20, 2011.

[3] N. H. Euliss Jr. and D. M. Mushet, "Water-level fluctuation in wetlands as a function of landscape condition in the prairie pothole region," *Wetlands*, vol. 16, no. 4, pp. 587–593, 1996.

[4] W. Ripl and K. D. Wolter, "Ecosystem function and degradation," in *Phytoplankton Productivity: Carbon Assimilation in Marine and Freshwater Ecosystems*, pp. 291–317, John Wiley & Sons, 2007.

[5] J. B. Zedler and S. Kercher, "Wetland resources: status, trends, ecosystem services, and restorability," *Annual Review of Environment and Resources*, vol. 30, pp. 39–74, 2005.

[6] A. R. G. Large, W. M. Mayes, M. D. Newson, and G. Parkin, "Using long-term monitoring of fen hydrology and vegetation to underpin wetland restoration strategies," *Applied Vegetation Science*, vol. 10, no. 3, pp. 417–428, 2007.

[7] R. J. Orth, M. R. Williams, S. R. Marion et al., "Long-term trends in submersed aquatic vegetation (SAV) in Chesapeake Bay, USA, related to water quality," *Estuaries and Coasts*, vol. 33, no. 5, pp. 1144–1163, 2010.

[8] G. Wiegleb, U. Bröring, M. Filetti, H. Brux, and W. Herr, "Long-term dynamics of macrophyte dominance and growth-form types in two north-west German lowland streams," *Freshwater Biology*, vol. 59, no. 5, pp. 1012–1025, 2014.

[9] R. Chmara, J. Szmeja, and W. Ulrich, "Patterns of abundance and co-occurrence in aquatic plant communities," *Ecological Research*, vol. 28, no. 3, pp. 387–395, 2013.

[10] D. Xie, H. Zhou, H. Ji, and S. An, "Ecological restoration of degraded wetlands in China," *Journal of Resources and Ecology, Institute of Geographic Sciences and Natural Resources Research, Chinese Academy of Sciences*, vol. 4, no. 1, pp. 63–69, 2013.

[11] C. D. K. Cook, *Aquatic Plant Book*, SPB Academic Publishing, New York, NY, USA, 1990.

[12] S. Peng, T. Zhou, and D. Wang, "Restoration of degraded ecosystem," *Contemporary Ecology Research in China*, pp. 235–263, 2015.

[13] P. Wang, Q. Zhang, Y.-S. Xu, and F.-H. Yu, "Effects of water level fluctuation on the growth of submerged macrophyte communities," *Flora: Morphology, Distribution, Functional Ecology of Plants*, vol. 223, pp. 83–89, 2016.

[14] L.-F. Yu and D. Yu, "Responses of the threatened aquatic plant Ottelia alismoides to water level fluctuations," *Fundamental and Applied Limnology*, vol. 174, no. 4, pp. 295–300, 2009.

[15] X. Zhang, X. Liu, and Q. Ding, "Morphological responses to water-level fluctuations of two submerged macrophytes, *Myriophyllum spicatum* and *Hydrilla verticillata*," *Journal of Plant Ecology*, vol. 6, no. 1, pp. 64–70, 2013.

[16] A. I. J. M. Van Dijk and L. A. Bruijnzeel, "Modelling rainfall interception by vegetation of variable density using an adapted analytical model. Part 2. Model validation for a tropical upland mixed cropping system," *Journal of Hydrology*, vol. 247, no. 3-4, pp. 239–262, 2001.

[17] A. M. Holm, W. A. Loneragan, and M. A. Adams, "Do variations on a model of landscape function assist in interpreting the growth response of vegetation to rainfall in arid environments?" *Journal of Arid Environments*, vol. 50, no. 1, pp. 23–52, 2002.

[18] C. De Michele, R. Vezzoli, H. Pavlopoulos, and R. J. Scholes, "A minimal model of soil water-vegetation interactions forced by stochastic rainfall in water-limited ecosystems," *Ecological Modelling*, vol. 212, no. 3-4, pp. 397–407, 2008.

[19] D. A. Lashof and D. R. Ahuja, "Relative contributions of greenhouse gas emissions to global warming," *Nature*, vol. 344, no. 6266, pp. 529–531, 1990.

[20] R. K. Pachauri, M. R. Allen, V. R. Barros et al., *IPCC Climate Change 2014: Synthesis Report. Contribution of Working Groups I, II and III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change*, IPCC, 2014.

[21] M. C. Acreman, J. R. Blake, D. J. Booker et al., "A simple framework for evaluating regional wetland ecohydrological response to climate change with case studies from Great Britain," *Ecohydrology*, vol. 2, no. 1, pp. 1–17, 2009.

[22] J. W. Day Jr., J. Barras, E. Clairain et al., "Implications of global climatic change and energy cost and availability for the restoration of the Mississippi delta," *Ecological Engineering*, vol. 24, no. 4, pp. 253–265, 2005.

[23] S. Piao, P. Ciais, Y. Huang et al., "The impacts of climate change on water resources and agriculture in China," *Nature*, vol. 467, no. 7311, pp. 43–51, 2010.

[24] Y. Qian, Z. Wu, L. Zhang, H. Zhou, S. Wu, and Q. Yang, "Eco-environmental evolution, control, and adjustment for Aibi Lake catchment," *Environmental Management*, vol. 36, no. 4, pp. 506–517, 2005.

[25] R. Lázaro, F. S. Rodrigo, L. Gutiérrez, F. Domingo, and J. Puigdefábregas, "Analysis of a 30-year rainfall record (1967-1997) in semi-arid SE spain for implications on vegetation," *Journal of Arid Environments*, vol. 48, no. 3, pp. 373–395, 2001.

[26] J. M. Mitchell, *Climatic Change*, WMO Technical Publications, Geneva, Switzerland, 1966.

[27] R. Lázaro, R. W. Alexander, and J. Puigdefábregas, "Cover distribution patterns of lichens, annuals and shrubs in the Tabernas Desert, Almería, Spain," in *Vegetation Mapping: From Patch to Planet*, R. W. Alexander and A. C. Millington, Eds., pp. 19–40, John Wiley & Sons, 2000.

[28] R. Sneyers, "On the use of statistical analysis for the objective determination of climatic change," *Meterorol. Zeitschrift*, vol. 1, pp. 247–256, 1992.

[29] C. H. Goossens and A. Berger, "Annual and seasonal climatic variation over the Northern Hemisphere and Europe during the last century," *Annales Geophysicae*, vol. 4, pp. 385–400, 1986.

[30] J. R. Lanzante, "Resistant, robust and non-parametric techniques for the analysis of climate data: theory and examples, including applications to historical radiosonde station data," *International Journal of Climatology*, vol. 16, no. 11, pp. 1197–1226, 1996.

[31] Nanjing Institute of Geography; Chinese Academy of Sciences, *A Preliminary Report on the Comprehensive Investigation of Taihu Lake*, Science Press, Beijing, China, 1965.

[32] S. Z. Zhang, "Vegetation and its environmental condition in East Taihu Lake," in *Miscellany of Oceanology and Limnology*, pp. 84–87, Jiangsu Science Press, Nanjing, China, 1986.

[33] Taihu Basin Authority, ministry of Water Resources, Nanjing Institute of Geography and Limnology, and Chinese Academy of Sciences, *Atlas of Ecology and Environment of the Taihu Lake*, Science Press, Beijing, China, 2000.

[34] B. Q. Qin, W. P. Hu, and W. M. Chen, *Evolution Process and Mechanism of Water Environment in Taihu*, Science Press, Beijing, China, 2004.

[35] X.-H. Gu, S.-Z. Zhang, X.-L. Bai, W.-P. Hu, Y.-H. Hu, and X.-R. Wang, "Evolution of community structure of aquatic macrophytes in East Taihu Lake and its wetlands," *Acta Ecologica Sinica*, vol. 25, no. 7, pp. 1541–1548, 2005.

[36] K. Zhao, Y. Zhou, Z. Jiang et al., "Changes of aquatic vegetation in Taihu Lake from 1960," *Journal of Lake Science*, 2017.

[37] F.-L. Luo, X.-X. Jiang, H.-L. Li, and F.-H. Yu, "Does hydrological fluctuation alter impacts of species richness on biomass in wetland plant communities?" *Journal of Plant Ecology*, vol. 9, no. 4, pp. 434–441, 2016.

[38] C. Hudon, S. Lalonde, and P. Gagnon, "Ranking the effects of site exposure, plant growth form, water depth, and transparency on aquatic plant biomass," *Canadian Journal of Fisheries and Aquatic Sciences*, vol. 57, no. S1, pp. 31–42, 2000.

[39] G. Zhu, W. Li, M. Zhang, L. Ni, and S. Wang, "Adaptation of submerged macrophytes to both water depth and flood intensity as revealed by their mechanical resistance," *Hydrobiologia*, vol. 696, no. 1, pp. 77–93, 2012.

[40] B. Qin, P. Xu, Q. Wu, L. Luo, and Y. Zhang, "Environmental issues of Lake Taihu, China," *Hydrobiologia*, vol. 581, no. 1, pp. 3–14, 2007.

[41] C. Le, Y. Zha, Y. Li, D. Sun, H. Lu, and B. Yin, "Eutrophication of lake waters in China: cost, causes, and control," *Environmental Management*, vol. 45, no. 4, pp. 662–668, 2010.

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Research Article

Physicochemical Process, Crustacean, and *Microcystis* Biomass Changes In Situ Enclosure after Introduction of Silver Carp at Meiliang Bay, Lake Taihu

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In order to control cyanobacteria blooms with silver carp in Lake Taihu, an in situ experiment was carried out by stocking silver carp at a biomass of 35, 70, and 150 $g\,m^{-3}$ and no carp control in waterproof enclosures. Physicochemical water parameters and biomass of plankton were measured in enclosures to evaluate the suitable stocking density of silver carp for relieving internal nutrients and constraining cyanobacteria growth in Lake Taihu. It is found that the 35 $g\,m^{-3}$ silver carp group and 70 $g\,m^{-3}$ silver carp group presented lower total phosphorus, lower chlorophyll-*a*, and higher water transparency. Increased nitrogen to phosphorus ratio, which indicated the result of algae decline in fish presence enclosures, was attributed to decline of phosphorus. Phosphorus decline also exerted limitation on reestablish of cyanobacteria bloom. Crustacean zooplankton biomass and *Microcystis* biomass decreased significantly in fish presence enclosures. Silver carp could be more effective to regulate algae bloom in enclosures with dense cyanobacteria. Therefore, nonclassic manipulation is supposed to be appropriate method to get rid of cyanobacteria blooms in Lake Taihu by stocking 35 to 70 $g\,m^{-3}$ silver carp in application.

1. Introduction

In freshwater ecosystems, fish affect the structure and dynamics of pelagic plankton communities by trophic cascading effects [1, 2], which usually relates consumers to their environments by food web chains [3]. Removal of planktivorous fish could relieve the predation pressure on zooplankton community by top-down control; thus the enhancement of crustaceans zooplankton leads to decline of algal density [4, 5]. This method, used to be defined as classic biomanipulation, however, usually malfunctions [6] because the presence of zooplankton grazing-resistant species such as frequent carpet of fetid cyanobacteria disables or weakens top-down force in nutrient enrichment lakes. Moreover, as the absence of large size zooplankton such as *Daphnia* in these lakes, crustacean communities are not able to control algal bloom by zooplankton-target manipulation: indeed grazing pressure by zooplankton is useless [7].

Introduction of filter-feeding planktivorous fish to hypertrophic shallow freshwater lakes successfully is another effective method to regulate the algal community which is called nonclassic biomanipulation [8–10]. As for nonclassic biomanipulation, planktivorous fish directly collect food by filtering water via their gill rakers and hence, unselectively ingest plankton and detritus. However, the ecological effects of filter-feeding fish introduced to specific lakes have remained

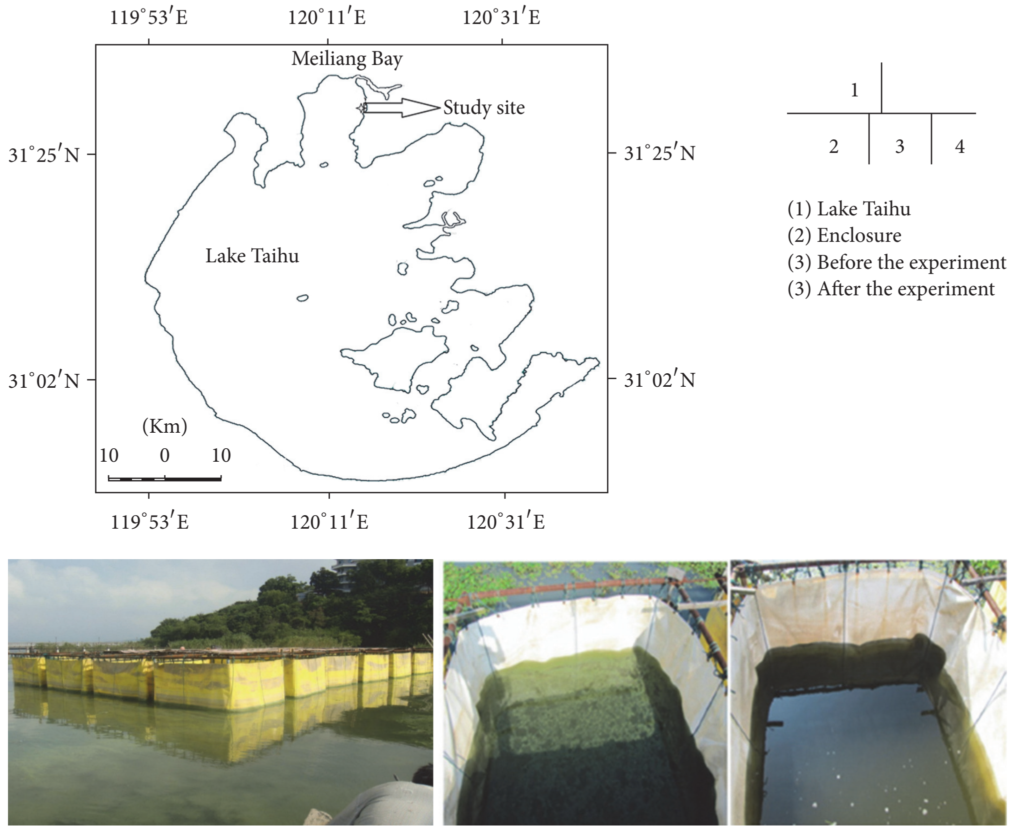

Figure 1: The study site and experimental enclosures.

controversial [5]. Three conditions for successful control of algae by planktivorous fish's grazing should be taken into account: (1) the stocking density and body size of filter-feeding silver carp [5]; (2) the initial plankton species pool [11] and (3) environmental conditions [12].

Lake Taihu is the third largest shallow freshwater lake located at the lower reach of the Yangtze River. Due to accumulative nutrient-rich sewage and agricultural run-off inflow, it became eutrophic with heavy cyanobacteria blooms from late spring to autumn every year in last decades [13]. In this lake, dominant crustacean zooplankton has changed from large body sized individuals, such as *Daphnia*, to small ones, such as *Limnoithona sinensis* during the last 60 years [14]. Hence, biomanipulation method to control algal blooms to improve water quality became impossible in Lake Taihu. As for nonclassic biomanipulation successfully in the hypereutrophic Lake Donghu, we consider using filter-feeding planktivorous fish to control cyanobacteria blooms in situ enclosure experiments in Lake Taihu.

Some previous studies [9, 10] investigated the ecological and biological effects of planktivorous silver carp and bighead carp with a certain density in large pen/enclosure at Meiliang Bay and Gonghu Bay, which are all located at the most serious bloom area in Lake Taihu. But we could not obtain optional stocking density of planktivorous fishes to control the dense algal blooms in this lake. Therefore, we carried out in situ enclosure experiments to explore the suitable stocking density of filter-feeding silver carp on controlling cyanobacteria blooms and hope to enhance fisheries resources for sustainable development based on decreasing the algal blooms and reasonable fish biodiversity in future.

2. Materials and Methods

2.1. Experiment Site and Device Settlement. Our experiment was located at Meiliang Bay (31°31–325′N, 120°09–340′E, Figure 1), which is administrated by Wuxi City, Jiangsu Province (Figure 1). Meiliang Bay receives heavy nutrient loads and has suffered eutrophication with serious algal blooms. The mean total phosphorus (TP) and total nitrogen (TN) are 0.1 mg L^{-1} (max 0.2 mg L^{-1}) and 2.3 mg L^{-1} (max 5.6 mg L^{-1}) in the northern east of Meiliang Bay [9, 15].

Four facets cuboid waterproof PVC enclosures (2.5 × 2.5 × 3 m installed in water depth at approximate 1.5 m littoral zone) were fixed to a cage of steel pipes. Bottom margins of each enclosure with heavy stone cages were sank into the lake sediment as possible to avoid water exchange between inside and outside of the enclosure during the study period. Its cost is about 3000 YUAN (equal to about 450 USD) per enclosure including all the materials and labour costs.

2.2. Study Protocol and Sampling Collection. Planktivorous fish silver carp (average weight 137 g ± 3 g) were obtained from a local aquatic farm and then acclimated in a nearby pond until they were transferred into the enclosures. The whole experiment aimed to explore the proper density of silver carp to control algae bloom.

In this experiment, 12 enclosures were chosen and randomly divided into four groups with triplications representing for control group (CG), low fish density group (LDG), medium fish density group (MDG), and high density group (HDG). CG, LDG, MDG, and HDG had fish biomass at 0, 35, 70, and 150 g m^{-3}. The experiment lasted from 30th May to 23rd June 2011 with a sampling interval of 3-4 days depending on the local weather. The average cyanobacteria biomass before the experiment in each enclosure was approximate about 5.8 g L^{-1} before the fish were introduced (confirmed with 2010 *Microcystis* biomass).

2.3. Experimental Parameters. Integrated water sample was collected by a 5 liter modified Patalas's bottle sampler. We determined water temperature, dissolved oxygen (DO), total dissolved solids (TDS), and pH using YSI Professional Plus (YSI Inc., Yellow Springs, Ohio, USA) water quality monitor. Water transparency was determined by using a 20-cm in diameter Secchi's disk and represented as Secchi's depth (Z_{Sd}). Turbidity was measured by a turbidimeter (Model TN-100, Eutech Instrument, Pte Ltd., Singapore).

50 mL samples of quantitative crustaceans were collected by filtering 10 L integrated water samples through $25^{\#}$ (69 μm) plankton net and then fixed with 1 mL saturated formalin. All individuals were counted after precipitation for 1 day by using an Olympus compound microscope (model BH2-RFC; Olympus America, Inc., Melville, NY, USA) at total 4×10 magnification in the samples to calculate density and biomass. Copepods and cladocerans were identified based on these papers [16, 17], and their wet weight was calculated according to the formula of these papers [18, 19]. Integrated water samples fixed with 1 mL saturated formaldehyde solution that was set volume to 50 mL was prepared for *Microcystis* spp. quantitative measurement. Colonial *Microcystis* was broken up to individual cells by an ultrasonic wave cell knapper (Model JY88-II, SCIENIZ, Ningbo, Zhejiang Prov., China) so that single cells could be counted. Fixed samples (0.1 mL) were tested using Olympus compound microscope under magnification of 40×10. Wet weight of *Microcystis* was calculated based on the formula of this paper [19]. All samples were collected at 7:00–8:30 a.m. to minimize variations between each sampling point.

Water chemistry including ammonia nitrogen (NH_4^+), nitrate nitrogen (NO_3^-), total nitrogen (TN), dissolved inorganic phosphorus, total dissolved nitrogen (TDN), total dissolved phosphorus (TDP), total phosphorus (TP), and chlorophyll-a were determined based on [20]. *Microcystis* was measured at a sampling interval between two sampling points, and water chemistry was tested on every sampling point.

In this experiment, physicochemical water parameters, zooplankton, and *Microcystis* spp. data were collected. Regressions analysis among physiochemical parameters over the fish groups was undertaken to investigate interactions of each parameter.

2.4. Statistical Analysis. Data were normalized and variance was adjusted for homogeneity before analysis. Means of dissolved oxygen, pH, total dissolved solids, and transparency were compared between control group and each treatment during whole experiment using independent T-test. Chemical parameters and chlorophyll-a indicators with treatments and time as two factors were subjected to two-way ANOVA (analyze of variance) using post hoc multiple by comparisons test (LSD) [21] and expressed as means ± standard deviation (STDEV). Data for regression analyses were subjected to $\ln(x + 1)$ transformation. Differences were measured against control values and considered to be statistically significant at $P < 0.05$. Statistical analyses mentioned above were undertaken using SPSS (Statistical Product and Service Solutions, IBM Inc.) 13.0 for Windows. Statistical figures were output by R [22] and OriginPro 8.0 (OriginLab Corporation).

3. Results

3.1. Physicochemical Water Parameters. In this experiment, water temperature varied from 22.1°C to 26.2°C (lowest and highest values were recorded on 30th May and 3rd July). pH and total dissolved solids did not show significant different between control group and each treatment. MDG showed the lowest dissolved oxygen in the whole experiment. Higher fish density group presented lower dissolved oxygen. In the LDG, transparency was measured to be significantly higher than in the CG whereas the lowest occurred in the HDG and even lower than in the CG (Table 1).

3.2. Nutrients Change. Nitrate and ammonia were found to be significantly higher ($P < 0.05$, $df_{7,3}$, other statistics were shown in Figure 2) in MDG and HDG during this experiment. Dissolved inorganic phosphorus was lowest in MDG (Figure 3). Total dissolved nitrogen was lowest in LDG, increased with increasing fish biomass and highest in no fish group (Figure 3). None of differences were found between treatments in total dissolved phosphorus and total nitrogen (Figure 4). Chlorophyll-a and total phosphorus were lower in LDG and MDG that compared with control and HDG (Figure 5).

During this experiment, TN : TP ratio values were lowest in CG (Table 2). Significant difference of TN : TP value was found between CG and MDG ($P = 0.028$, $t = -2.27$). Regression analyzes in fish groups between chlorophyll-a to TN and TP showed that TP was positively related to chlorophyll-a ($P < 0.001$, $r = 0.413$) while TN was less relative ($P = 0.095$, $r = 0.2$) to chlorophyll-a fluctuate (Table 3). Regression analyzes showed that TN was positively related to TN : TP ($P < 0.001$, $r = 0.856$) while TP was not related to TN : TP ($P = 0.514$, $r = -0.079$).

Transparency (Z_{Sd}) was negative related to *Microcystis* ($P < 0.01$, $r = -0.47$), TP ($P < 0.01$, $r = -0.33$), and chlorophyll-a ($P = 0.01$, $r = -0.54$). But TN was not observed related to all these parameters (Table 3).

Table 1: pH, dissolved oxygen (DO), total dissolved solids (TDS), and Secchi depth (Z_{Sd}) were measured during this experiment among each treatment. Significant differences compared between treatments and control groups were presented as means ± STDEV with different minuscules.

Parameters	Treatments	Means ± STDEV	Levene's test of variance of homogenous		T-test of means compared with control group		
			F	Sig.	t	df	P (two tails)
pH	CG	8.19 ± 0.42 (a)	—	—	—	—	—
	LDG	8.00 ± 0.40 (a)	0.506	0.481	1.602	46	0.116
	MDG	7.93 ± 0.36 (b)	1.508	0.226	2.284	46	0.027
	HDG	8.03 ± 0.39	0.400	0.530	1.319	46	0.194
DO (mg/L)	CG	7.72 ± 3.42 (a)	—	—	—	—	—
	LDG	6.21 ± 1.71 (a)	5.258	0.026	1.930	33.872$	0.062
	MDG	5.73 ± 2.23 (b)	2.099	0.154	2.384	46	0.021
	HDG	6.03 ± 2.17 (b)	2.129	0.151	2.036	46	0.048
TDS (g/L)	CG	0.42 ± 0.06	—	—	—	—	—
	LDG	0.45 ± 0.11	1.520	0.224	−1.004	46	0.321
	MDG	0.44 ± 0.10	1.010	0.320	−0.606	46	0.547
	HDG	0.40 ± 0.10	0.402	0.529	1.208	46	0.233
Z_{Sd} (cm)	CG	68.7 ± 9.9 (a)	—	—	—	—	—
	LDG	83.3 ± 12.7 (b)	0.088	0.768	−4.390	46	0.000
	MDG	75.2 ± 12.9 (b)	1.704	0.198	−1.950	46	0.057
	HDG	65.6 ± 12.7 (a)	0.856	0.360	0.956	46	0.344

$, variance was not equal.

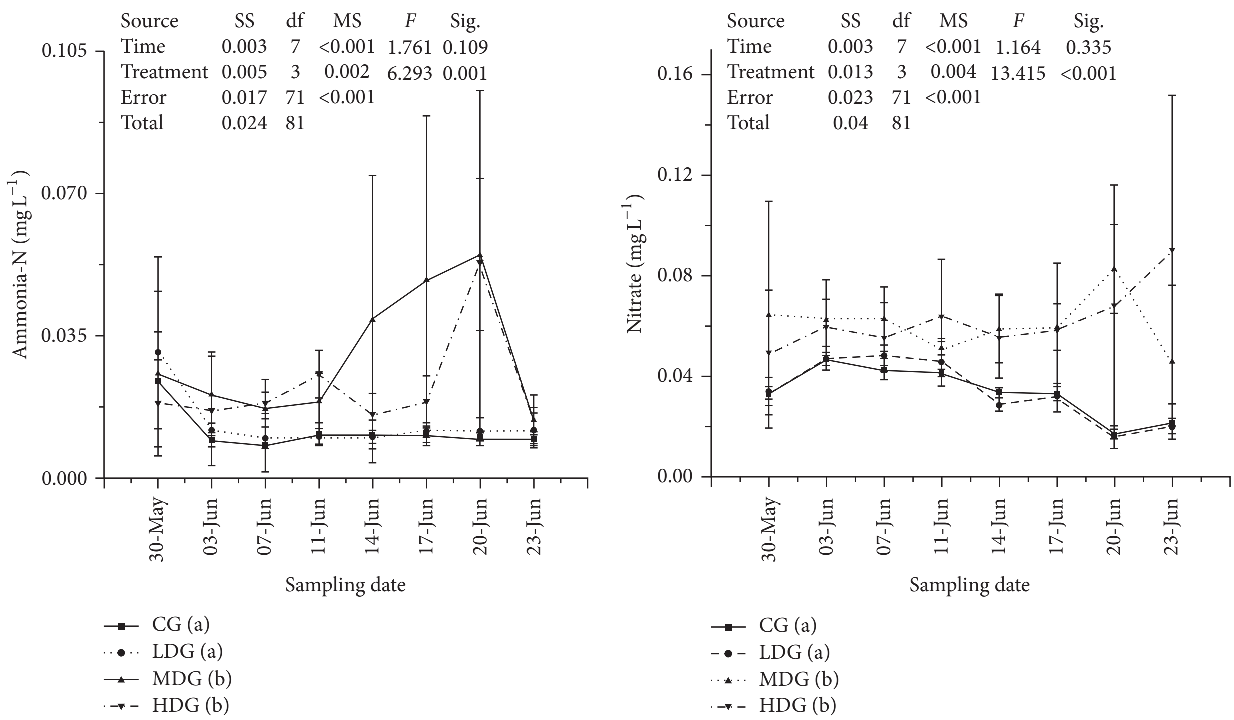

Figure 2: Nutrients values including ammonia and nitrate in each enclosure treatment during sampling points. Values were presented by mean ± STDEV. Treatments with different minuscules mean significant differences between each other and parameters in each treatment from low to high were arranged alphabetically (i.e., treatment with minuscule (a) means the lowest, at $P < 0.05$, $\alpha = 0.05$). SS: type III sum of squares, df: freedom, and MS: mean square.

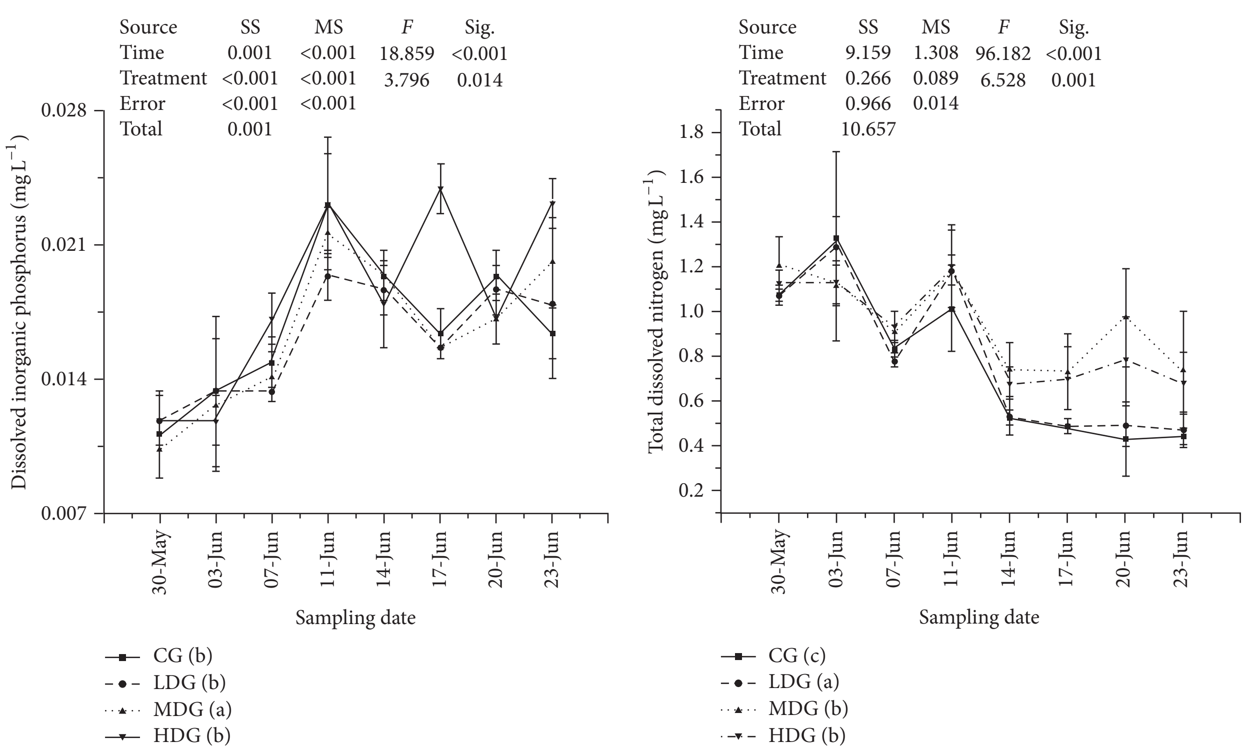

Figure 3: Nutrients values including dissolved inorganic phosphorus and total dissolved nitrogen in each enclosure treatment during sampling points. Values were presented by mean ± STDEV. Treatments with different minuscules mean significant differences between each other and parameters in each treatment from low to high were arranged alphabetically (i.e., treatment with minuscule (a) means the lowest, at $P < 0.05$, $\alpha = 0.05$). SS: type III sum of squares, df: freedom, and MS: mean square.

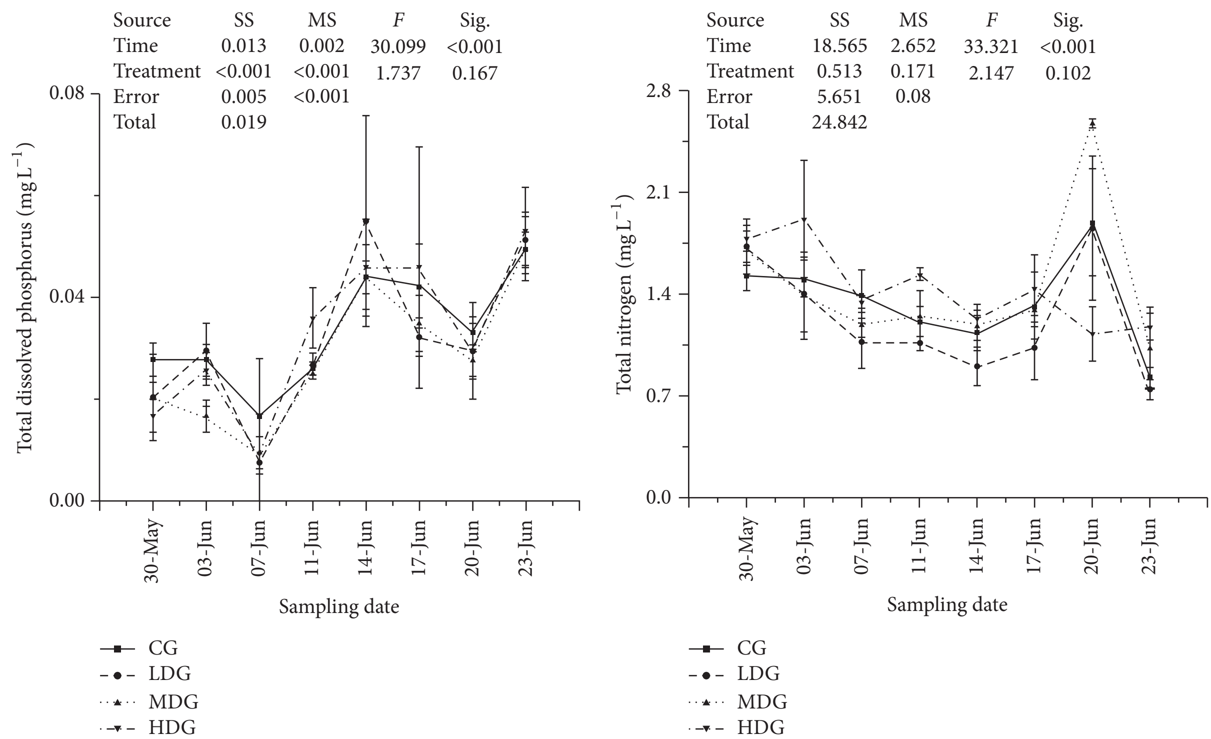

Figure 4: Nutrients values including total dissolved phosphorus and total nitrogen in each enclosure treatment during sampling points. Values were presented by mean ± STDEV. No significant differences between each other and parameters in each treatment. SS: type III sum of squares, df: freedom, and MS: mean square.

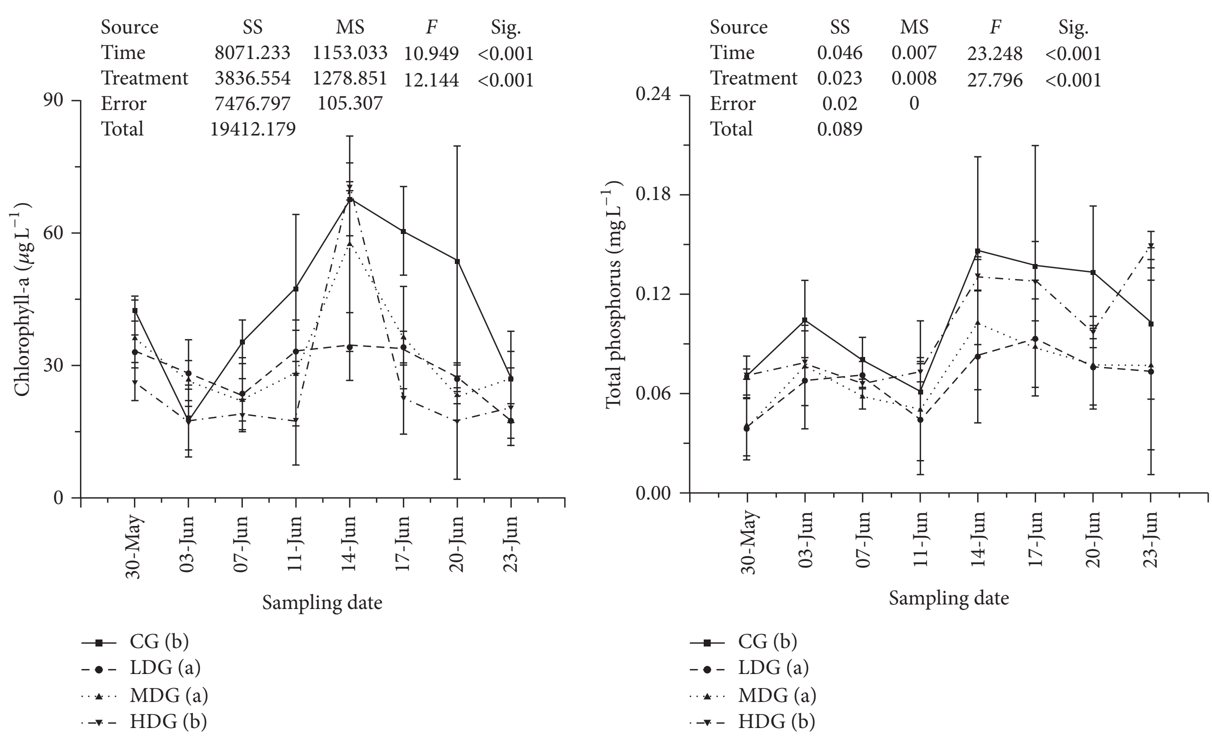

Figure 5: Nutrients values including total phosphorus and chlorophyll-a in each enclosure treatment during sampling points. Values were presented by mean ± STDEV. Treatments with different minuscules mean significant differences between each other and parameters in each treatment from low to high were arranged alphabetically (i.e., treatment with minuscule (a) means the lowest, at $P < 0.05$, $\alpha = 0.05$). SS: type III sum of squares, df:freedom, and MS: mean square.

Table 2: N : P ratio values and comparison between CG and treatments for each treatment. Results of TN : TP ratio are performed as mean ± STDEV; means with asterisk "*" stand for significant difference (T-test).

Group	TN : TP ratio values	Test for equality of variances		T-test for equality of means		
		F	Sig.	t	df	Sig.
CG	12.43 ± 6.57	Compared with CG				
LDG	14.70 ± 7.58	0.670	0.418	−1.095	45	0.279
MDG	17.72 ± 9.31*	2.230	0.134	−2.271	46	0.028
HDG	13.87 ± 7.71	1.908	1.908	−0.693	46	0.492

Table 3: Pearson (upper-right corner) and Spearman's rank (lower-left corner) correlation coefficients (r) between variables. TP: total phosphorus, TN: total nitrogen, NO_3: nitrate, NH_4: ammonia, MC: *Microcystis*, Z_{Sd}: Secchi depth, and Chla: chlorophyll-a.

	TP	TN	NO_3	NH_4	MC	pH	Z_{Sd}	Chla
TP		0.57	0.13	0.10	0.21	0.03	0.00	0.01
TN	−0.07		0.86	0.84	0.18	0.14	0.16	0.10
NO_3	−0.49	0.21		0.01	0.17	0.45	0.63	0.05
NH_4	0.03	−0.03	0.37		0.89	0.34	0.88	0.07
MC	0.54	−0.16	−0.10	0.40		0.01	0.00	0.32
pH	0.04	−0.29	0.31	0.30	0.36		0.76	0.44
Z_{Sd}	−0.33	0.21	−0.22	−0.11	−0.47	−0.43		0.01
Chla	0.43	0.20	0.35	0.27	0.44	0.72	−0.54	

3.3. Microcystis spp. and Crustaceans. Mean *Microcystis* during this experiment was lowest in the LDG while the highest was in the CG. The mean *Microcystis* biomass was significantly different in the fish groups to the control (Table 4). Dominant crustacean zooplanktons in our study were identified as *Limnoithona sinensis*, *Mesocyclops leuckarti*, *Thermocyclops taihokuensis*, *Bosmina* spp., and *Diaphanosoma* spp. Other zooplankton species were also found: *Ceriodaphnia cornuta*, *Sinocalanus dorrii*, *Thermocyclops* spp., *Canthocamptus* spp., and *Moina micrura*. During this experiment, crustaceans decreased significantly in the fish enclosures (Table 4).

4. Discussion

In the present study, fish at the lower density ($35\,g\,m^{-3}$ to $70\,g\,m^{-3}$) inhibited cyanobacteria blooms more efficiently. These enclosures performed as refined water quality, lower nutrient, and cyanobacteria density; relative higher zooplankton biomass than higher fish group.

The vital debates on successful biomanipulations in a long period usually depend on whether they can efficiently release internal load [23]. Closed system, just like in the present study, soluble nutrients, for example, NH_4^+ and NO_3^-, increased with increasing fish density, indicating density dependent effects that fish interfere water chemical process by their metabolism. Studies have illustrated TN, TP, and chlorophyll-a in the fish presence enclosures significantly lower than in the fish absence [24]. At relatively low densities, the silver carp were able to graze for particles of food directly, resulting in a decline in phosphorus and chlorophyll-a levels within the Lake Taihu enclosures [25]. Hence, the threshold of fish density usually should be taken into account for evaluating the risk and the advantage that fish could bring out. If the systems were enlarged to full-lake scale manipulation, the results are mixed at best.

The authors reported that reduction of stocking fish density promoted water quality in four Netherlands and Denmark shallow lakes [26]. Pond study also supported the result [27]. In the 1990s, an in situ enclosure experiment of silver carp manipulation was carried out in Lake Donghu, the experimental results showed that stocking density of 46–$50\,g\,m^{-3}$ silver carp can more effectively control the algae bloom, and the algae bloom was removed accompanied with decreased nutrients, which is similar to our experimental results [8]. Previous study proposed that water quality in pen stocked with about $40\,g\,m^{-3}$ filter-feeding silver carp near our experiment area did not differ from outside water because of the large water exchange both inside and outside pen area [9]. The authors countered those 3 cases out of 11 filter-feeding fish biomanipulations in enclosure experiments which showed decreased total phosphorus, while 5 cases showed no effects and remaining 3 cases showed increased total phosphorus [28]. These evidences prove that filter feedings fish could interfere with water physicochemical process, so we want to study how the filter feedings fish could interfere with water physicochemical process.

In the present study, the increasing value of total nitrogen to total phosphorus ratio in fish presence enclosures indicated that cyanobacteria bloom was alleviated by fish, by referring

Table 4: Crustacean and *Microcystis* biomass for each treatment. Statistics are performed as mean ± STDEV; means with different minuscules stand for significant difference.

Variable	Treatments			
	CG	LDG	MDG	HDG
Crustacean biomass (μg/L)	87.67 ± 23.50 (a)	12.34 ± 6.77 (b) $P < 0.01$	6.89 ± 4.34 (b) $P < 0.01$	7.03 ± 5.51 (b) $P < 0.01$
Microcystis spp. biomass (mg/L)	6.53 ± 2.44 (a)	2.55 ± 0.43 (c) $P < 0.01$	4.18 ± 0.37 (b) $P = 0.042$	3.94 ± 0.22 (b) $P = 0.031$

cyanobacteria bloom explosion usually results in decline of TN : TP ratio [29]. In the present study, N did not decrease by fish's grazing, and consequently, increased TN : TP ratio in fish groups should be caused by TP decline. This means the top-down effects by fish to algal community was triggered by P fluctuation. Usually, P is considered to be the first regulatory factor that can limit growth of algae communities, while N is the secondary factor [30]. According to the positive relation between TP and chlorophyll-a in the present study, P decline promoted the possibility of limitation of algal growth from bottom-up, even though the absolute phosphorus (average TP > 150 μg L^{-1}) and nitrogen (average TN > 1 mg L^{-1}) in the study area were enough for the cyanobacteria growth.

5. Conclusions

The present study provide evidences that, in enclosure conditions, fish at a density of 35 g m^{-3} to 70 g m^{-3} could be effective in controlling *Microcystis* blooms, promotion of fish production, and ameliorating the aquatic environment. Non-classic biomanipulation is a proper means to reduce nutrients and phytoplankton under conditions of (1) eutrophic or hypereutrophic water, (2) lack of large sized zooplankton, and (3) dominance of filamentous or colonial algae.

Competing Interests

The authors declare that there is no conflict of interests regarding the publication of this paper.

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References

[1] K. Christoffersen, B. Riemann, A. Klysner, and M. Sondergaard, "Potential role of fish predation and natural populations of zooplankton in structuring a plankton community in eutrophic lake water," *Limnology and Oceanography*, vol. 38, no. 3, pp. 561–573, 1993.

[2] M. Coll and K. Hargadon, "Trophic and functional cascades in tropical versus temperate aquatic microcosms," *Aquatic Ecology*, vol. 46, no. 1, pp. 55–71, 2012.

[3] M. Gophen, "Biomanipulation: retrospective and future development," *Hydrobiologia*, vol. 200-201, no. 1, pp. 1–11, 1990.

[4] D. J. McQueen, J. R. Post, and E. L. Mills, "Trophic relationships in freshwater pelagic ecosystems," *Canadian Journal of Fisheries and Aquatic Sciences*, vol. 43, no. 8, pp. 1571–1581, 1986.

[5] I. Domaizon and J. Devaux, "Experimental study of the impacts of silver carp on plankton communities of eutrophic Villerest reservoir (France)," *Aquatic Ecology*, vol. 33, no. 2, pp. 193–204, 1999.

[6] P. Xie and J. K. Liu, "Practical success of biomanipulation using filter-feedingfish to control cyanobacteria blooms: a synthesis of decades of research and application in a subtropical hypereutrophic lake," *TheScientificWorldJournal*, vol. 1, pp. 337–356, 2001.

[7] H. Ma, F. Cui, Z. Fan, Z. Liu, and Z. Zhao, "Efficient control of *Microcystis* blooms by promoting biological filter-feeding in raw water," *Ecological Engineering*, vol. 47, pp. 71–75, 2012.

[8] P. Xie, *Silver Carp and Bighead, and Their Use in the Control of Algal Blooms*, Science Press, Beijing, China, 2003.

[9] Z. Ke, P. Xie, L. Guo, Y. Liu, and H. Yang, "In situ study on the control of toxic Microcystis blooms using phytoplanktivorous fish in the subtropical Lake Taihu of China: a large fish pen experiment," *Aquaculture*, vol. 265, no. 1-4, pp. 127–138, 2007.

[10] L. G. Guo, Q. Wang, P. Xie et al., "A non-classical biomanipulation experiment in Gonghu Bay of Lake Taihu: control of *Microcystis* blooms using silver and bighead carp," *Aquaculture Research*, vol. 46, no. 9, pp. 2211–2224, 2015.

[11] S. Datta and B. B. Jana, "Control of bloom in a tropical lake: grazing efficiency of some herbivorous fishes," *Journal of Fish Biology*, vol. 53, no. 1, pp. 12–24, 1998.

[12] M. Fukushima, N. Takamura, L. Sun, M. Nakagawa, K. Matsushige, and P. Xie, "Changes in the plankton community following introduction of filter-feeding planktivorous fish," *Freshwater Biology*, vol. 42, no. 4, pp. 719–735, 1999.

[13] L. Guo, "Doing battle with the green monster of Taihu Lake," *Science*, vol. 317, no. 5842, article 1166, 2007.

[14] X. Sun, M. Tao, B. Qin et al., "Large-scale field evidence on the enhancement of small-sized cladocerans by *Microcystis* blooms in Lake Taihu, China," *Journal of Plankton Research*, vol. 34, no. 10, pp. 853–863, 2012.

[15] M. Tao, P. Xie, J. Chen et al., "Use of a generalized additive model to investigate key abiotic factors affecting microcystin cellular quotas in heavy bloom areas of lake Taihu," *PLoS ONE*, vol. 7, no. 2, Article ID e32020, 2012.

[16] S. C. Chiang and N. S. Du, *Fauan Sinica, Crustacea, Freshwater Cladocera*, Science Press, Beijing, China, 1979.

[17] J. R. Sheng, *Fauna Sinica, Crustacea, Freshwater Copepoda*, Science Press, Beijing, China, 1979.

[18] X. F. Huang and C. Y. Hu, *Regression Equations of the Body Weight to Body Length for Common Freshwater Species of Cladocera*, Transaction of the Chinese Crustacean Society, Science Press, Beijing, China, 1986.

[19] Z. S. Zhang and X. F. Huang, *Research Methods of Freshwater Plankton*, Science Press, Beijing, China, 1991.

[20] A. E. Greenberg, L. S. Clesceri, and A. D. Eaton, *Standard Methods for the Examination of Water and Wastewater*, American Public Health Association, Washington, DC, USA, 1992.

[21] R. G. D. Steel and J. H. Torrie, *Principles and Procedures of Statistics*, A Biometrical Approach, McGraw-Hill, New York, NY, USA, 2nd edition, 1980.

[22] R Core Team, *R: A Language and Environment for Statistical Computing*, R Foundation for Statistical Computing, Vienna, Austria, 2014, http://www.R-project.org/.

[23] R. Drenner and K. D. Hambright, "Piscivores, trophic cascades, and lake management," *The Scientific World Journal*, vol. 2, pp. 284–307, 2002.

[24] C. L. Yi, L. G. Guo, L. Ni, and C. Luo, "Silver carp exhibited an enhanced ability of biomanipulation to control cyanobacteria bloom compared to bighead carp in hypereutrophic Lake Taihu mesocosms," *Ecological Engineering*, vol. 89, pp. 7–13, 2016.

[25] C. Yi, L. Guo, L. Ni, C. Yin, J. Wan, and C. Yuan, "Biomanipulation in mesocosms using silver carp in two Chinese lakes with distinct trophic states," *Aquaculture*, vol. 452, pp. 233–238, 2016.

[26] M. L. Meijer, E. Jeppesen, E. van Donk et al., "Long-term responses to fish-stock reduction in small shallow lakes: interpretation of five-year results of four biomanipulation cases in The Netherlands and Denmark," *Hydrobiologia*, vol. 275, no. 1, pp. 457–466, 1994.

[27] I. J. Holopainen, W. M. Tonn, and C. A. Paszkowski, "Effects of fish density on planktonic communities and water quality in a manipulated forest pond," *Hydrobiologia*, vol. 243-244, no. 1, pp. 311–321, 1992.

[28] X. Zhang, P. Xie, and X. Huang, "A review of nontraditional biomanipulation," *The Scientific World Journal*, vol. 8, pp. 1184–1196, 2008.

[29] P. Xie, *A Review on the Cause of Cyanobacterial Blooms from an Evolutionary, Biogeochemical and Ecological View of Point*, Science Press, Beijing, China, 2007.

[30] H.-J. Wang, X.-M. Liang, P.-H. Jiang, J. Wang, S.-K. Wu, and H.-Z. Wang, "TN: TP ratio and planktivorous fish do not affect nutrient-chlorophyll relationships in shallow lakes," *Freshwater Biology*, vol. 53, no. 5, pp. 935–944, 2008.

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Research Article

Effect of Hydrologic Alteration on the Community Succession of Macrophytes at Xiangyang Site, Hanjiang River, China

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With the intensification of human activities over the past three decades in China, adverse effects on river ecosystem become more serious especially in the Hanjiang River. Xiangyang site is an important spawn ground for four domestic fishes in the downstream region of Hanjiang River. Based on the field survey results of macrophytes during 1997–2000 and 2013-2014, community succession of aquatic macrophytes at Xiangyang site was evaluated and discussed. Two-key ecologic-related hydrologic characteristics, flow regime and water level, were identified as the main influence factors. The EFC (environmental flow components) parameters were adopted to evaluate the alteration of flow regimes at Xiangyang site during 1941–2013. Evaluation results demonstrate a highly altered flow process after being regulated by reservoir. The flow patterns tend to be an attenuation process with no large floods occurring but a higher monthly low flow. Furthermore, the water level decreased and fluctuation reduced after the dam was built, which caused the decrease of biomass but favored the submerged macrophytes during 1995–2009. However, with the water level increasing after 2010 and gently fluctuating, due to uplift by the hydraulic projects downstream as well as the flow attenuation, the dominant position of submerged macrophytes will be weakened.

1. Introduction

The benign reproductive of aquatic macrophytes is increasingly recognized as a crucial factor to the aquatic ecosystem health and the food chain integrity [1]. Aquatic macrophytes influence many aspects of river structure and function, and their developmental changes strongly affect the whole river ecosystem process [2, 3]. Meanwhile, macrophytes abundance and community composition can be useful indicators of aquatic ecological health, as they reflect the quality of both physical and chemical habitat conditions [4, 5]. With the intense influences of human activities and climate change, rivers throughout world are at risk for loss of macrophytes communities and community succession [6]. Many surveys and experiments have been carried out to evaluate the status of aquatic macrophytes and try to find the main influence reasons, including illumination, nutrition, sediment, flow, water level, and temperature [7–10]. But few concerns focused on the change of natural characteristics of river based long-term historical data. Studying the evolution of natural hydrologic regime will probably reveal the external reason of degradation of macrophytes and communities succession [11, 12].

Hydrology is considered to be the most important determining factor in wetland functions [13]. Hydrological characteristics, such as water level and water flow, are significant factors influencing macrophytes [14]. Based on the studies on aquatic habitats, including macrophytes and fishes, Wantzen et al. [15] concluded that aquatic macrophytes growing in the littoral zone were sensitive to changes in the water level fluctuation regime. Ogdahl and Steinman [5] surveyed macrophytes beds in Muskegon Lake over a 4-year period to assess the ecological benefits of the restoration project and found that macrophyte biomass was affected strongly by the physical features of the individual sites (i.e., hydrologic exposure) and a possible environmental cause (i.e., water level and air temperature). Mjelde et al. [14] developed a water level drawdown index for Nordic lakes to identify the suitable fluctuation amplitude range for the sensitive and tolerant aquatic plants. Zhu et al. [16] conducted an experiment to investigate the plant growth, root anchorage strength, and stem tensile properties of five submerged macrophytes under three initial water levels (1.0, 2.5, and 4.0 m) with four water level fluctuation speeds (0, 5, 15, and 25 cm d^{-1}). The results showed that deep water can inhibit plant growth and decrease their mechanical resistance. All of those studies demonstrated an important role of hydrologic characteristics in the growth of aquatic macrophytes. However, it is worth noting that those studies were based on a short-term series, normally a few months or 4-5 years, which was too short to evaluate the change of hydrologic characteristics. The hydrological characteristics around the world present complex changes due to human disturbance, such as reservoir operation, water diversion, and land appropriation [17–19] that require long history data to evaluate. In this paper, long-term hydrologic data between 1943 and 2014 were used to evaluate the river ecosystem-related hydrologic alteration and reveal the reasons of community succession of Xiangyang site, which is the important spawn ground of the four domestic fishes in Hanjiang River. The paper is organized as follows: Section 2 introduces the data and methodology used for evaluation; Section 3 presents the evaluation results of community succession and hydrological alteration; Section 4 conducts the discussion of the influences of hydrological alteration; the conclusions are summarized in Section 5.

2. Study Area, Data, and Methods

2.1. Study Area. Xiangyang site is located at the mid reach of Hanjiang River with a humid, monsoon climate. The average annual precipitation, temperature, and runoff during the period of 1960–2013 were 865 mm, 16°C, and 1197 m^3/s, respectively. About 85% of the annual runoff comes from the Han River, with the rest from the South River, North River, Tanghe River, and Xiaoqing River. The current aquatic macrophytes communities at Xiangyang site include *P. perfoliatus, P. malaianus, Vallisneria natans, Salvinia natans, A. philoxeroides,* and *Artemisia selengensis* [20]. Since 1960s, the river flow of Xiangyang site is greatly regulated by the Danjiangkou reservoir in the upstream region, which is a large-comprehensive reservoir and required to supply water for the mid-route of South-to-North water diversion project in 2014. With more water stored and transferred from the upstream Hanjiang River, the hydrological characteristics of Xiangyang site have been greatly changed, which boost the pressure of local water management and ecological restoration. It is necessary to study the hydrological alteration and evaluate the aquatic ecosystem health to provide scientific basis for the protection of river ecosystem and sustain the function of spawning ground of four domestic fishes.

2.2. Data. In this study, the daily flow data of Xiangyang hydrological station were obtained from 1941 to 1960 and 1973 to 2013. The observed data during 1961–1972 was not available because Danjiangkou Dam was built and blocked the river. We also collected the 6-hour water level measurement data during 1956–1960 and 1973–2014; the missing data during a day are filled by using linear interpolation. Besides, the cross-section measurement data of Xiangyang site were obtained as well. All of those hydrologic data were excerpted from the "Hydrological Year Book" published by the Hydrological Bureau of the Ministry of Water Resources.

We also collected some measurement data about the aquatic macrophytes of Mid-Lower Hanjiang River from two field surveys conducted during 1997–2000 [20] and 2013-2014 (unpublished). During 1997–2000, three parallel areas were selected as sample site at the Xiangyang site with the sampling interval of 5 km. The covered distance from the center of each sample site was 1 km, during which the coastal area, subcoastal zone, central deep water area, and shoal were selected as the sampling sites. GPS was used for positioning. The aquatic plant communities in the shallow water area and the wetlands were collected by harvesting, and 2 m × 2 m grass samplers were chosen for recording. During 2013-2014, the plant communities were sampled with the methods of sampling in parallel and repeated small random sampling plots. The heights of the plant community above 2 m, 2 m-1 m, and 1 m were sampled at 2 m × 2 m, 1 m × 1 m, and 0.5 m × 0.5 m, respectively, and the minimum sampling area of each community was not less than 3 samples. The aquatic plant communities in the shallow water area and the river surface were collected by harvesting, and 0.5 m × 0.5 m grass samplers were used for the river surface sampling. All the plants in the quadrats were uprooted, washed, and weighed (wet weight). The two-field surveys revealed the changes of macrophytes types and distribution at Xiangyang site.

2.3. Environmental Flow Component. Not only is it essential to maintain adequate flows during low flow periods, but higher flows and floods and extreme low flow conditions also perform important ecological functions [21]. Ecological researchers have demonstrated that river hydrographs can be divided into a repeating set of hydrographic patterns that are ecologically relevant. Based on the widely used evaluation indexes of flow regime [22], five types of flow events were proposed to represent the full spectrum of flow conditions that are crucial to sustain riverine ecological integrity; they are low flows, extreme low flows, high flow pulses, small floods, and large floods [23]. The five EFC types are described in more detail in Table 1.

The five EFC types were classified based on the predam daily flow data series, which were regarded as the natural flow without human disturbance. We firstly defined the high flows and low flows, which were simply separated by using a sing fixed threshold. The flows that exceeded 75% of daily flow for the period will be classified as high flows, while those below this level will be classified as low flows. A small flood event was defined as an initial high flow with a peak flow greater than 2-year return interval event. A large flood event was defined as an initial high flow with a peak flow greater than 10-year return interval event. An extreme low flow was defined as an initial low flow below 10% of daily flows for the period.

3. Results

3.1. Community Succession of Macrophytes at Xiangyang Site. Based on the survey results, the aquatic macrophytes

Table 1: Summary of environmental flow component (EFC) parameters and their ecosystem influences.

EFC type	Hydrologic parameters	Ecosystem influences
(1) Monthly low flows	Mean or median values of low flows during each calendar month *(Subtotal 12 parameters)*	(1) Provide adequate habitat for aquatic organisms; (2) maintain suitable water temperatures, dissolved oxygen, and water chemistry; (3) maintain water table levels in floodplain, soil moisture for plants; (4) provide drinking water for terrestrial animals; (5) keep fish and amphibian eggs suspended; (6) enable fish to move to feeding and spawning areas; (7) support hyporheic organisms (living in saturated sediments)
(2) Extreme low flows	Frequency of extreme low flows during each water year or season Mean or median values of extreme low flow event Duration (days) Peak flow (minimum flow during event) Timing (Julian date of peak flow) *(Subtotal 4 parameters)*	(1) Enable recruitment of certain floodplain plant species; (2) purge invasive and introduced species from aquatic and riparian communities; (3) concentrate prey into limited areas to benefit predators
(3) High flow pulses	Frequency of high flow pulses during each water year or season Mean or median values of high flow pulse event Duration (days) Peak flow (maximum flow during event) Timing (Julian date of peak flow) Rise and fall rates *(Subtotal 6 parameters)*	(1) shape physical character of river channel, including pools, riffles; (2) determine size of streambed substrates (sand, gravel, cobble); (3) prevent riparian vegetation from encroaching into channel; (4) restore normal water quality conditions after prolonged low flows, flushing away waste products and pollutants; (5) aerated eggs in spawning gravels and prevent siltation; (6) maintain suitable salinity conditions in estuaries
(4) Small floods	Frequency of small flood during each water year or season Mean or median values of high flow pulse event Duration (days) Peak flow (maximum flow during event) Timing (Julian date of peak flow) Rise and fall rates *(Subtotal 6 parameters)*	Applies to small and large floods: (1) provide migration and spawning cues for fish; (2) trigger new phase in life cycle (i.e., insects); (3) enable fish to spawn in floodplain and provide nursery area for juvenile fish; (4) provide new feeding opportunities for fish, water flow; (5) recharge floodplain water table; (6) maintain diversity in floodplain forest types through prolonged inundation (i.e., different plant species have different tolerances); (7) control distribution and abundance of plants on floodplain; (8) deposit nutrients on floodplain
(5) Large flood	Frequency of large flood during each water year or season Mean or median values of high flow pulse event Duration (days) Peak flow (maximum flow during event) Timing (Julian date of peak flow) Rise and fall rates *(Subtotal 6 parameters)*	Applies to small and large floods: (1) maintain balance of species in aquatic and riparian communities; (2) create sites for recruitment of colonizing plants; (3) shape physical habitats of floodplain; (4) deposit gravel and cobbles in spawning areas; (5) flush organic materials (food) and woody debris (habitat structures) into channel; (6) purge invasive, introduced species from aquatic and riparian communities; (7) disburse seeds and fruits of riparian plants; (8) drive lateral movement of river channel, forming new habitats (secondary channels, oxbow lakes); (9) provide plant seedlings with prolonged access to soil moisture

communities during 1997–2000 and 2013-2014 are evaluated in this section. It is clear that the major community types in the two periods are totally different (Table 2). During 1997–2000, there were two dominant plant communities, Ass. *P. pectinatus* + *P. perfoliatus* and Ass. *P. malaianus* + *Hydrocharis dubia*; both belonged to the submerged aquatic macrophytes. When coming to 2013-2014, the macrophyte communities of Xiangyang site changed to submerged plants, floating plants, emergent plants, and hygrophytes, respectively, represented by *P. malaianus* and *P. perfoliatus*, *Salvinia natans* and *V. natans*, *A. philoxeroides,* and *Artemisia selengensis*. It demonstrates the succession of aquatic macrophytes community occurring during 2013-2014 at Xiangyang site. According to their surveys in the whole Mid-Lower Hanjiang River, the ratio of submerged macrophytes association with the total population decreased from 70% in 1997–2000 to 31.58% in 2013-2014, and the ratio of hygrophytes association increased from zero to 31.58%. It seems that the community succession of macrophytes occurred in the whole Mid-Lower Hanjiang River basin, which also influences the macrophytes types

Table 2: The Biomass and distribution of aquatic plant communities at Xiangyang site in different periods.

1997–2000			2013-2014		
Major community type	Biomass (g/m^2)	Coverage (%)	Major community type	Biomass (g/m^2)	Coverage (%)
(1) Ass. *P. pectinatus* + *P. perfoliatus* (2) Ass. *P. malaianus* + *Hydrocharis dubia*	12025	70	(1) Ass. *P. malaianus* + *P. perfoliatus* (2) Ass. *Salvinia natans* and *V. natans* (3) Ass. *A. philoxeroides* (4) Ass. *Artemisia selengensis*	7789	75

of Xiangyang site. Moreover, the aquatic macrophytes of Xiangyang site were greatly degraded with the biomass declining from 12025 g/m^2 to 7789 g/m^2. The community succession and degradation of macrophytes strongly indicate the evolution of the river ecosystem, which is probably caused by human disturbance and the change of natural environment.

To assess the influence of hydrologic alteration, we calculate the EFC of Xiangyang site during 1997–2014 based on the daily flow data. The thresholds of low flow, high pulse flow, extreme low flow, small foods, and large floods were all determined by the predam observed daily data, which was regarded as the natural regime. Form Figure 1, we clearly see that no flood events were observed during the whole period of 1997–2013, which indicates a smaller and smoother flow regime compared with the natural flow. Figure 1 also demonstrates a smaller low flow and a less occurrence of high flow pulse during 2013-2014 compared with 1997–2000. For the period 2013-2014 is shorter than 1997–2000, we calculated the occurrence proportions of the three flow events during each period. The results show that between 1997 and 2000, the high flow pulse accounted for 11% of the whole flow events, while in 2013-2014 the occurrence proportion of high flow pulse was just 5%. However, the occurrence proportion of low flows was increased from 88% during 1997–2000 to 95% in 2013-2014, and no extreme low flows were observed in 2013-2014. It seems that the river flow during 2013-2014 was more flat with a relative smaller volume and fluctuations, which is more favorable for those low flow-enduring aquatic plants and small fluctuation-appetite plants.

The water level fluctuation is also very important with regard to affecting the function of river ecosystem, especially in the growth and distribution of aquatic plants [24, 25]. Based on the water level measurement data between June and October during 1997–2014, we surprisingly find that the water level during 2013-2014 presented a quite flat and high level (Figure 2). The water level of 2013-2014 maintained a high level of above 64.5 m, much higher than the average water level during 1997–2000. Zhu et al. [16] found that the submerged macrophytes are inadaptable to deep water level that their biomass, relative growth rate, and roots anchorage strength decreased with increasing initial water level. Moreover, with the water level increasing, the illumination decreases. Changes in light intensity and nutrient concentrations were considered to be key determinants of

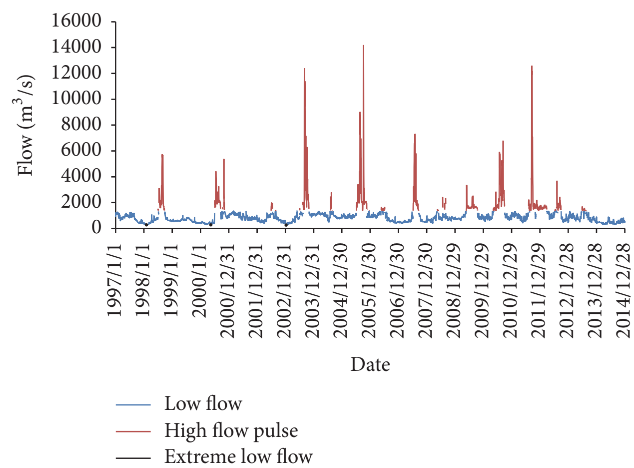

Figure 1: The environmental flow component of Xiangyang site during 1997–2014.

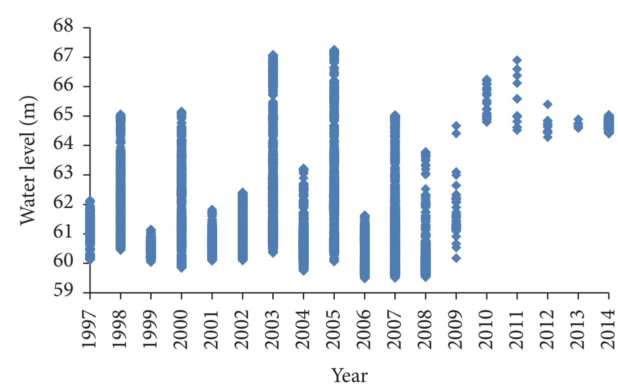

Figure 2: The distribution of water level within a year during 1997–2014.

growth of submerged macrophytes [26, 27]. This partly explains the decrease of submerged macrophytes during the high water level period of 2013-2014. Besides, the fluctuation range of water level was significantly reducing. Before 2010, the fluctuation range of water level was large with the maximum fluctuation amplitude of 6.74 m occurring in 2005 and average water level fluctuation amplitude of 2.98 m. However, the fluctuation amplitude reduced suddenly after 2009 with a value of 0.3 m in 2013 and 0.62 m in 2014. The reduced fluctuation of water level probably promotes the growth of some aquatic macrophytes.

As an important factor of human disturbance, water quality determines the sound growth of aquatic plants in many aspects. According to the agriculture information data from the local government in 2000 and 2014, the NH_4-N of Xiangyang site decreased from 0.256 mg/L to 0.248 mg/L, the TN increased from 1.39 mg/L to 1.74 mg/L, and the TP increased from 0.034 mg/L to 0.059 mg/L. The change of chemical concentrate was not significant in the 14 years compared with the change of hydrologic characteristics. Consequently, a conclusion can be preliminarily derived where the macrophytes degradation and community succession at Xiangyang site occurred. It is strongly related to hydrologic alteration, such as a smaller flow and a less fluctuation of water level, which may change the types and distribution of aquatic macrophytes and influence the effectiveness of river ecological restoration. Accordingly, it is necessary to deeply evaluate the hydrologic alteration at Xiangyang site so as to provide more scientific suggestions for river ecosystem restoration.

3.2. The Hydrologic Alterations. To evaluate the change of hydrologic characteristics, we firstly divide the study period into predam periods (1941–1960) and postdam periods (1974–2013) based on the daily flow data of Xiangyang hydrological station. And then we separate the postdam period into two stages: the less human disturbance stage (1973–1990) and the intense human disturbance stage (1991–2013).

According to the EFC assessment frame, we calculated the 32 EFC parameters in the predam periods and postdam periods, respectively, listed in Table 3. It is notable that the length of observed flow data in the predam period is less than 20 years that some information will be lost when calculating the frequency of flood events. For example, the frequency of large food in predam period is nearly 0 but actually happened 4 times. Therefore, we substitute counts for the frequency to reveal the happening of the flood events. From Table 3, we clearly find that, compared with the predam period, all of the 32 parameters changed in different degrees with the large flood events changing mostly. Since the Danjiangkou Dam was built, no large flood events happened in terms of the zero values of all the six parameters in the large floods group, although the length of postdam period is longer than predam period. On the other side, the low flow of the 12 months all became bigger than in predam period in terms of the positive relative alteration rate. It is worth noting that the largest relative alteration rate all happened in low flow seasons, such as January, February, and December. It indicated a bigger flow process in dry seasons than predam period. In addition, significant decrease is seen clearly in the frequency and duration of extreme low flows (RB = −0.77 and −1, resp.). The flow peak, duration, frequency, rising rate, and falling rate of high flow pulse all presented decreasing trend with the rising rate and falling rate decreasing most drastically (RA = −0.83 and −0.73, resp.). The small flood counts greatly decreased compared with predam period (9 versus 42) even though the postdam period was much longer than the predam period. The flow regime alteration after dam was built demonstrates that the river flow of Xiangyang hydrological station behaved in a homogenization and attenuation tendency.

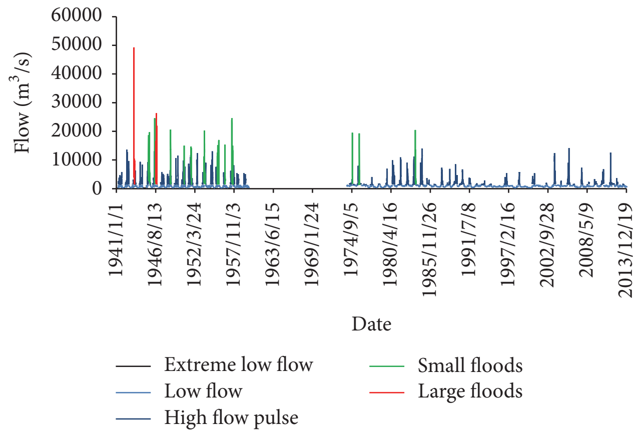

Figure 3: The results of environmental flow component (EFC) from 1941 to 2013.

It is notable that most of the parameters with significant change primarily happened during 1991–2013, the intense human disturbance period, such as the January low flow, February low flow, December low flow, the extreme low flow counts, the rise and fall rate of high flow pulse, and the decrease of small flood events. The relative alterations of low flow in January, February, and December were much higher during 1991–2013 with the values of 1.13, 1.22, and 0.66, compared to the relative alteration of 0.13, 0.09, and 0.09 during 1974–1990. And the relative alteration rate of rise rate of high flow pulse during 1991–2013 was high up to −0.81 compared with −0.12 during 1973–1990, similar to the fall rate of high flow pulse. What is more, the small floods in the postdam period all happened before 1990. No floods were observed during 1991–2013. It indicates a more significant change of flow regime after 1990 under the impact of intense human activities.

Figure 3 shows the spectrum of environmental flow component from 1941 to 2013 (except the missing data), which clearly demonstrates a smaller flow process and the attenuating fluctuation after dam was built. Compared with the predam period, the large floods totally disappeared and the small floods were decreased obviously in magnitude and occurrence. The peak of high flow pulses became smaller and low flows became larger. Besides, Figure 3 also shows a substantial change of flow in the intense human disturbance period, such as the disappearing of small foods and low occurrence of high flow pulse.

Accordingly, it is reasonable to believe that the flow regime of Xiangyang hydrological station has been changed to a great degree since dam was built, especially after 1990. The disappearing of flood events was largely caused by the flood control in flood seasons, and the higher monthly low flows were mainly due to the water supply from reservoir in nonflood seasons. The reduced fluctuation of high flow pulse was because of setting reservoir regulation rules formulated by the government. Those great changes in flow regime could be a trigger of changing in the structure of aquatic and riparian macrophytes.

Table 3: Comparison results of the parameters of EFC in the three periods at Xiangyang site.

Parameters	Pre	Post			Relative alteration		
		Post (all)	Post (1974–1990)	Post (1991–2013)	Post (all)	Post (1974–1990)	Post (1991–2013)
January low flow	354	857	971	754	**1.42**	0.13	**1.13**
February low flow	354	852	931	786.3	**1.4**	0.09	**1.22**
March low flow	551	830	883	770	0.51	0.06	0.40
April low flow	724	853	936.8	853	0.18	0.10	0.18
May low flow	826	988	1040	893	0.2	0.05	0.08
June low flow	582	949	1160	889.3	0.63	0.22	0.53
July low flow	761	1130	1245	1125	0.48	0.10	0.48
August low flow	767	1175	1270	1090	0.53	0.08	0.42
September low flow	887	1050	1220	947.5	0.18	0.16	0.07
October low flow	785	790	1050	719	0.01	0.33	−0.08
November low flow	624	764	941.5	749.5	0.23	0.23	0.20
December low flow	422	746	810.5	700	**0.77**	0.09	**0.66**
Extreme low peak	262	264	264	252	0.01	0	−0.04
Extreme low duration	11	2.5	4.5	1.75	**−0.77**	**0.80**	**−0.84**
Extreme low timing	45.5	48	60	22	0.05	0.25	−0.52
Extreme low counts	694	146	136	10	**−1**	−0.07	**−0.99**
High flow peak	2430	1530	1530	1520	−0.37	0	−0.37
High flow duration	5	3	3.5	3	−0.4	0.17	−0.40
High flow timing	182	201	200	205	0.1	0.00	0.13
High flow frequency	9	6	8	4	−0.33	0.33	−0.56
High flow rise rate	758	125	110	145.5	**−0.83**	−0.12	**−0.81**
High flow fall rate	−408	−110	−86.14	−126.2	**−0.73**	−0.22	**−0.69**
Small flood peak	19700	19500	19400		−0.01	−0.01	/
Small flood duration	28	39	46		0.39	0.18	/
Small flood timing	218	278	278		0.28	0.28	/
Small flood counts	42	9	9	0	**−0.78**	**−0.78**	/
Small flood rise rate	1408	4830	3250		**2.43**	−0.33	/
Small flood fall rate	−800	−532	−1165		−0.33	**1.19**	/
Large flood peak	37800				/	/	/
Large flood duration	49				/	/	/
Large flood timing	225.5		281		/	/	/
Large flood counts	4	0	0	0	/	/	/
Large flood rise rate	8814		4830		/	/	/
Large flood fall rate	−916		−532.2		/	/	/

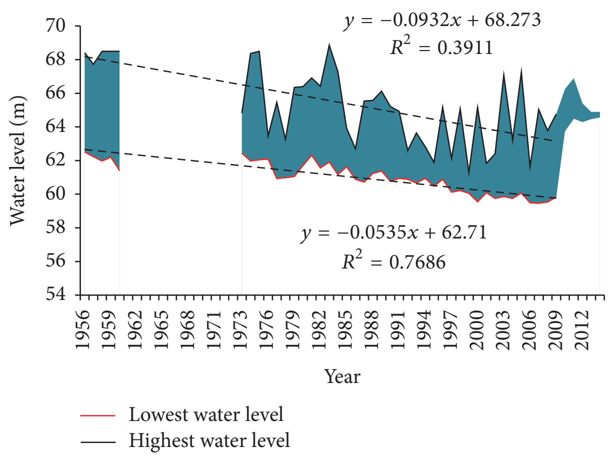

FIGURE 4: The fluctuation range of water level at Xiangyang site between 1973 and 2014.

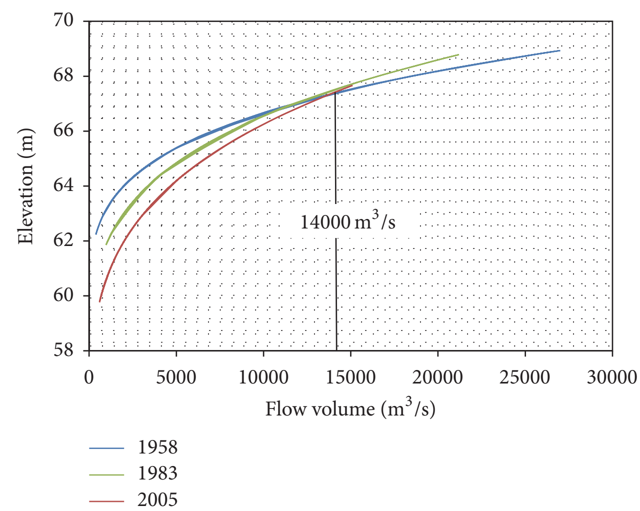

FIGURE 5: The relation curves of water level and flow ($Z \sim Q$) of the three periods.

3.3. The Water Level Variation. Water level is changing with time but has a stable statistical fluctuation path when seeing a long-term measurement data, which is favorable for maintaining the stability of the aquatic plants structure. However, with the change of flow regime and human disturbance, the law of water level fluctuation changed as well. Figure 4 shows the time series of water level fluctuation range of Xiangyang hydrological station from May to October during 1956–2014, the aquatic plant growing period. Due to the same reason, the observed data of water level was missing during 1961–1973. It is clearly seen that both of the two time series present significant decrease during 1973–2009 with the lowest water level decreasing more significantly. Meanwhile, the fluctuation range presented reducing trend compared with the predam period. The decrease of water level and fluctuation range was related to the flood attenuation caused by flood control. What is more, the changing in physical features of the river channel also led to water level decreasing.

Xiangyang is located 109 km away from the Danjiangkou Dam. Affected by the damming upstream, the sediment source running into the downstream channel was changed radically. In the predam period, the sediment in Xiangyang section was basically brought by upstream runoff. But now, due to reservoir, the sediment from upstream regions is zero if without reservoir flood discharge in nonflood seasons, and only 1.98% of reservoir sediment will be deposited into the downstream channel with the flood discharge. Instead, the sediment in the downstream river channel mainly comes from the limited silt scored from the river bed or brought by the tributary. It means that the sediment of Xiangyang site is decreasing year by year and the inflow become much cleaner than before. With the scouring of clean water and less sedimentation, the channel physical characteristics were changed to some degree, and the relationship between water level and river flow was changed correspondingly.

Figure 5 shows the $Z \sim Q$ relation curves of the three years, 1958, 1983, and 2005. It is clear that the three relation curves, respectively, reflect the changes in the shape of river channel in different periods. Hereinto, the relation curve of 1958 represents a natural state of river channel without human disturbance. Under the impact of reservoir operation, the $Z \sim Q$ relation behaved differently with the change of sediment component-amount and developed to a new stability after 1983, when large flood happened and created a strong scour. After 1983, the downstream channel was experiencing the clear water scour. The different behaviors of the three relation curves lead to the same flow volume producing different water levels in different periods. When the river flow was smaller than 14,000 m^3/s, the water level in 1958 behaved highest with the same amount of inflow, while the water level of 2005 behaved lowest. The results were totally inverted when the river flow was greater than 14,000 m^3/s. Based on the flood data, the observed value of flood peak dated from 1984 was rarely greater than 14,000 m^3/s. It seems that the river channel was less and less submerged after dam was built even with the same amount of released water. The truth is that released water was greatly reduced that the study river was experiencing a lower water level period in the postdam period. With the decreasing of water level, some low water level-adaptable macrophytes probably started to grow and develop, such as submerged macrophytes. However, the water level rose suddenly after 2009, which was caused by the flow uplift of downstream hydraulic project. The water level rising further changed the relation curve of $Z \sim Q$ and the macrophytes structure of Xiangyang site.

Based on the evaluation of hydrologic alteration on the flow regime and water level, it is reasonable to believe that the hydrologic features of Xiangyang site have been greatly changed due to reservoir operation and human disturbance. The river flow tended to be smaller and the water level became lower compared with the predam period. The sudden rise of water level from 2010 was caused by the uplift of downstream hydraulic project, which will continue to raise the water level of Xiangyang site in the future. Those changes imposed great influences on the adjustment of aquatic macrophytes structure.

4. Discussions

The study of fresh wetland indicated that productivity and decomposition of aquatic species are positively related to the magnitude of hydrologic inputs [28, 29]. The average gross primary productivity of water column producers in the high flow wetland was generally higher than in low flow wetlands [30]. Compared with predam period, a relative smaller flow process was observed in the aquatic plant growth period (generally between April and October), in terms of the disappearing of large floods and significant decrease of small floods, which partly hindered the growth of aquatic plants and led to the decreasing of biomass at Xiangyang site. Meanwhile, the changing in relation curve of $Z \sim Q$ leads to an even lower water level with the regulated smaller flow input and aggravated the decrease of biomass. According to the water level measurement data from 1956 to 2014, the minimum water level decreased from 62.51 m in 1956 to 59.83 m in 2009, and the maximum water level decreased from 68.41 m in 1956 to 64.76 m in 2009 (see Figure 4). The average decrease rate was about 0.53 m/year and 0.73 m/year, respectively. The shallow zone will be totally exposed when the water level is lower than 59.49 m based on the cross-section of Xiangyang site in 2006. The decrease of water level and smaller inflow resulted in the river channel being less submerged that some part of the river channel was not suitable for the growth of aquatic plants. Accordingly, we speculate that the macrophytes biomass of Xiangyang site presented decreasing trend over a long-term period after dam was built.

However, the lower water level could provide the ideal growth condition for the submerged aquatic plants. Zhu et al. [16] investigated the lake in Yunnan province, located in the southwest of China, and found that the uppermost depth for most submerged plants was approximately 6 m and the highest biomass of an individual plant was within 3 m. From 1973 to 2006, the minimum water level decreased from 62.44 m to 59.49 m, with the corresponding water depth decreasing from 0.08–9 m to 0–6 m based on the cross-section of Xiangyang site (Figure 6). It seems that the water level of the whole section was more suitable for submerged macrophytes during 1995–2006 when there was a lower water level, while only some shallow zones were suitable for the submerged plants in the early period. The phenomenon of submerged macrophytes being the dominant species during 1997–2000 was largely related to the suitable water level.

But the sudden rise of water level after 2009 imposes an unfavorable influence on the growth of aquatic plants. This is because deep water decreases light availability that hinders the growth of aquatic plants [31]. After 2010, the minimum water level rose to 65.29 m sharply in 2010 with a water depth of 11.29 m, which exceeded the uppermost depth of the submerged plants and reduced the growth rate of emergent plants. But the shallow zone was totally submerged again with the water level rising, with a depth of approximately 3 m according to the cross-section. It was the most appropriate living environment for submerged macrophytes. It explains why the submerged plants were just degraded but not disappearing from the macrophyte communities of Xiangyang site after the water level rose to a pretty high level. Considering the geometry of the cross-section of Xiangyang site and irrevocable influences from hydraulic project, the water level within 62.3–65.3 m is appropriate to keep the shallow water zone submerged and make it suitable for the growth of submerged macrophytes. On the other side, after the water level rising, the stable water level in the shallow water zone also provided more chances for the leaves surfacing, which favored the growth of emergent macrophytes and floating plants [32]. It seems that the dominant position of submerged plants will be weakened to a certain degree in the future. It is noteworthy that, influenced by the continuous uplift of downstream hydraulic project, the flow velocity is decreased and the water level fluctuation is reduced, against the water self-purification and pollutant output. The water quality should be highly valued to prevent the deterioration of the living circumstances of submerged plants.

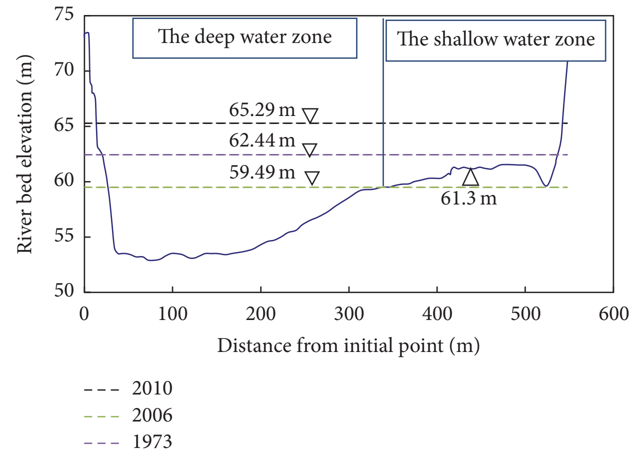

Figure 6: The cross-section of Xiangyang site and the minimum water level in 2010, 2006, and 1973.

5. Conclusions

In this study, the succession of macrophytes communities at Xiangyang site is discussed. Two ecology-related hydrological features at Xiangyang site, flow regime and water level, are evaluated into three periods based on their time sequence characteristics and the Hanjiang River development. The main reasons of changing in those parameters are discussed and explained. Some main findings can be summarized as follows:

(1) The status of macrophyte communities at Xiangyang site is evaluated by comparing the survey results during 1997–2000 and 2013-2014. The results demonstrate an obvious macrophytes succession in terms of the submerged plants disappearing and new species appearing. The macrophytes types changed from submerged macrophytes to a diversity pattern including submerged plants, emergent plants, hygrophyte, and floating plants. And the biomass declined from 12025 g/m^2 to 7789 g/m^2, which indicates the community degradation at Xiangyang site. The main reasons were identified as hydrologic alteration.

(2) The flow regime of Xiangyang site has been greatly changed due to dam being built with the flood events decreasing most significantly. Moreover, the monthly low flow presented increase in dry seasons and the fluctuation of high flow pulse became smaller. The variability of flow regime indicates a homogenization and attenuation tendency. The alteration of flow regime behaved most drastically during 1991–2013, which was influenced by the intense human activities in the end of last century. The smaller flow input partly resulted in the decrease of biomass at Xiangyang site.

(3) The variability of water level during 1973–2013 is evaluated and the results demonstrate a downward trend and a decreased fluctuation range. Beside impact of the smaller flow input, the changing in water level variation was also related to the long-term clear water scour and less sedimentation that changed the physical characteristics of the river channel. The relation curve between water level and river flow ($Z \sim Q$) after dam was built was changed to a low efficient pattern. The same flow volume corresponded to a lower water level in recent period compared with the predam period. It indicates that in most time of postdam period the channel was less submerged which makes it possible for the growth of some shallow water adaptable macrophytes.

(4) The influences of long-term hydrologic alteration on the community succession and macrophytes degradation of Xiangyang site are discussed. The deceasing water level promoted submerged plants being the dominant plants during the field survey period of 1997–2000, while the water level sudden rise after 2009 caused the disappearing of submerged plants in the deep water zone. However, the shallow water zone was totally submerged which reprovided favorable environment for the growth of submerged plants. The appropriate water level of Xiangyang site should be 62.3–63.5 m for the growth of submerged plants and maintaining the shallow water zone submerged.

Based on the evaluation of the influence of hydrologic alteration, we can conclude that hydrologic characteristics, especially water levels, play an important role in the growth of aquatic macrophytes. The restoration of river ecosystem should fully consider the change trend of hydrologic features and formulate more effective and feasible plans.

Competing Interests

The authors declare that there is no conflict of interests regarding the publication of this paper.

Acknowledgments

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References

[1] S. J. Nichols and F. J. Dyer, “Contribution of national bioassessment approaches for assessing ecological water security: An AUSRIVAS Case Study,” *Frontiers of Environmental Science and Engineering*, vol. 7, no. 5, pp. 669–687, 2013.

[2] H. Coops, M. Beklioglu, and T. L. Crisman, “The role of water-level fluctuations in shallow lake ecosystems—workshop conclusions,” *Hydrobiologia*, vol. 506, no. 1, pp. 23–27, 2003.

[3] E. Jeppesen, M. Meerhoff, T. A. Davidson et al., “Climate change impacts on lakes: an integrated ecological perspective based on a multi-faceted approach, with special focus on shallow lakes,” *Journal of Limnology*, vol. 73, no. 1, pp. 88–111, 2014.

[4] G. P. Grabas, A. E. Blukacz-Richards, and S. Pernanen, “Development of a submerged aquatic vegetation community index of biotic integrity for use in Lake Ontario coastal wetlands,” *Journal of Great Lakes Research*, vol. 38, no. 2, pp. 243–250, 2012.

[5] M. E. Ogdahl and A. D. Steinman, “Factors influencing macrophyte growth and recovery following shoreline restoration activity,” *Aquatic Botany*, vol. 120, pp. 363–370, 2015.

[6] K. Zhang, A. D. D. A. Castanho, D. R. Galbraith et al., “The fate of Amazonian ecosystems over the coming century arising from changes in climate, atmospheric CO_2, and land use,” *Global Change Biology*, vol. 21, no. 7, pp. 2569–2587, 2015.

[7] D. H. N. Spence, “The zonation of plants in freshwater lakes,” *Advances in Ecological Research*, vol. 12, pp. 37–125, 1982.

[8] P. Wang, Q. Zhang, Y.-S. Xu, and F.-H. Yu, “Effects of water level fluctuation on the growth of submerged macrophyte communities,” *Flora—Morphology, Distribution, Functional Ecology of Plants*, vol. 223, pp. 83–89, 2016.

[9] C. M. Duarte and J. Kalff, “Patterns in the submerged macrophyte biomass of lakes and the importance of the scale of analysis in the interpretation,” *Canadian Journal of Fisheries and Aquatic Sciences*, vol. 47, no. 2, pp. 357–363, 1990.

[10] G. R. Zhu, M. Zhang, T. Cao, L. Y. Ni, A. W. Zhong, and H. Fu, “Effects of sediment type on stem mechanical properties of the submerged macrophyte *Hydrilla verticillata* (L.F.) Royle,” *Fresenius Environmental Bulletin*, vol. 21, no. 2, pp. 468–474, 2012.

[11] K. E. Havens and A. D. Steinman, “Ecological responses of a large shallow lake (Okeechobee, Florida) to climate change and potential future hydrologic regimes,” *Environmental Management*, vol. 55, no. 4, pp. 763–775, 2015.

[12] C. H. A. van Leeuwen, J. M. Sarneel, J. van Paassen, W. J. Rip, and E. S. Bakker, “Hydrology, shore morphology and species traits affect seed dispersal, germination and community assembly in shoreline plant communities,” *Journal of Ecology*, vol. 102, no. 4, pp. 998–1007, 2014.

[13] W. J. Mitsch and J. G. Gosselink, *Wetlands*, Van Nostrand Reinhold, New York, NY, USA, 2nd edition, 1993.

[14] M. Mjelde, S. Hellsten, and F. Ecke, “A water level drawdown index for aquatic macrophytes in Nordic lakes,” *Hydrobiologia*, vol. 704, no. 1, pp. 141–151, 2013.

[15] K. M. Wantzen, K.-O. Rothhaupt, M. Mörtl, M. Cantonati, L. G. Tóth, and P. Fischer, “Ecological effects of water-level fluctuations in lakes,” in *Developments in Hydrobiology*, vol. 204, pp. 21–31, Springer, Berlin, Germany, 2008.

[16] G. Zhu, W. Li, M. Zhang, L. Ni, and S. Wang, “Adaptation of submerged macrophytes to both water depth and flood intensity as revealed by their mechanical resistance,” *Hydrobiologia*, vol. 696, no. 1, pp. 77–93, 2012.

[17] R. E. Sparks, "Risks of altering the hydrologic regime of large rivers," in *Predicting Ecosystem Risk*, J. B. Cairns, B. R. Niederlehner, and D. R. Orvos, Eds., pp. 119–152, Princeton Scientific Publishing, Princeton, NJ, USA, 1992.

[18] J.-T. Shiau and F.-C. Wu, "Assessment of hydrologic alterations caused by Chi-Chi diversion weir in Chou-Shui Creek, Taiwan: opportunities for restoring natural flow conditions," *River Research and Applications*, vol. 20, no. 4, pp. 401–412, 2004.

[19] J. H. Larson, A. S. Trebitz, A. D. Steinman et al., "Great Lakes rivermouth ecosystems: scientific synthesis and management implications," *Journal of Great Lakes Research*, vol. 39, no. 3, pp. 513–524, 2013.

[20] X. U. Xin-Wei, W. U. Zhong-Hua, Y. U. Dan, L. I. U. Chun-Hua et al., "Diverstiy of aquatic plants in Hanjiang River Basin and possible effects of the engineering of transferring water from south China to North China on it," *Acta Ecologca Sinica*, vol. 22, no. 11, pp. 1933–1938, 2002 (Chinese).

[21] M. T. Casanova and M. A. Brock, "How do depth, duration and frequency of flooding influence the establishment of wetland plant communities?" *Plant Ecology*, vol. 147, no. 2, pp. 237–250, 2000.

[22] B. D. Richter, J. V. Baumgartner, J. Powell, and D. P. Braun, "A method for assessing hydrologic alteration within ecosystems," *Conservation Biology*, vol. 10, no. 4, pp. 1163–1174, 1996.

[23] J. T. Mobley, T. B. Culver, and R. W. Burgholzer, "Environmental flow components for measuring hydrologic model fit during low flow events," *Journal of Hydrologic Engineering*, vol. 17, no. 12, pp. 1325–1332, 2012.

[24] S. M. Thomaz, T. A. Pagioro, L. M. Bini, and K. J. Murphy, "Effect of reservoir drawdown on biomass of three species of aquatic macrophytes in a large sub-tropical reservoir (Itaipu, Brazil)," *Hydrobiologia*, vol. 570, no. 1, pp. 53–59, 2006.

[25] W. T. Z. Sousa, S. M. Thomaz, and K. J. Murphy, "Response of native Egeria najas Planch. and invasive Hydrilla verticillata (L.f.) Royle to altered hydroecological regime in a subtropical river," *Aquatic Botany*, vol. 92, no. 1, pp. 40–48, 2010.

[26] C. E. Korschgen, W. L. Green, and K. P. Kenow, "Effects of irradiance on growth and winter bud production by *Vallisneria americana* and consequences to its abundance and distribution," *Aquatic Botany*, vol. 58, no. 1, pp. 1–9, 1997.

[27] X. Liu, Y. Zhang, K. Shi, J. Lin, Y. Zhou, and B. Qin, "Determining critical light and hydrologic conditions for macrophyte presence in a large shallow lake: the ratio of euphotic depth to water depth," *Ecological Indicators*, vol. 71, pp. 317–326, 2016.

[28] W. H. Conner and J. W. Day, "Productivity and composition of a baldcypress-water tupelo site and a bottomland hardwood site in a louisiana swamp," *American Journal of Botany*, vol. 63, no. 10, pp. 1354–1364, 1976.

[29] W. J. Mitsch and B. C. Reeder, "Modelling nutrient retention of a freshwater coastal wetland: estimating the roles of primary productivity, sedimentation, resuspension and hydrology," *Ecological Modelling*, vol. 54, no. 3-4, pp. 151–187, 1991.

[30] J. K. Cronk and W. J. Mitsch, "Aquatic metabolism in four newly constructed freshwater wetlands with different hydrologic inputs," *Ecological Engineering*, vol. 3, no. 4, pp. 449–468, 1994.

[31] S. J. Blanch, G. G. Ganf, and K. F. Walker, "Growth and resource allocation in response to flooding in the emergent sedge *Bolboschoenus medianus*," *Aquatic Botany*, vol. 63, no. 2, pp. 145–160, 1999.

[32] M. P. Cooling, G. G. Ganf, and K. F. Walker, "Leaf recruitment and elongation: an adaptive response to flooding in *Villarsia reniformis*," *Aquatic Botany*, vol. 70, no. 4, pp. 281–294, 2001.

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Research Article

Numerical Response of Migratory Shorebirds to Prey Distribution in a Large Temperate Arid Wetland, China

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Wuliangsuhai Lake provides important breeding and stopover habitats for shorebirds. The health of this wetland ecosystem is rapidly deteriorating due to eutrophication and water pollution and environmental management is urgently needed. To explore the connections among ecosystem health, prey density, and shorebird populations, we conducted surveys of both the benthic macroinvertebrates and shorebirds in the shorebird habitat of the wetland during the 2011 autumn migration season. The abundance of both shorebirds and benthic macroinvertebrates varied significantly in both space and time. Our data showed a clear association between shorebird populations and the density of benthic macroinvertebrates, which explained 53.63% of the variation in shorebird abundance. The prey density was strongly affected by environmental factors, including water and sediment quality. Chironomidae were mainly found at sites with higher total phosphorus, but with lower sediment concentrations of Cu. Lymnaeidae were mainly found at sites with a higher pH, lower salinity, and lower concentrations of total phosphorus and Cu. Habitats with very high concentrations of total phosphorus, heavy metals, or salinity were not suitable for benthic macroinvertebrates. Our findings suggest that the reductions of nutrient and heavy metal loadings are crucial in maintaining the ecological function of Wuliangsuhai as a stopover habitat for migratory shorebirds.

1. Introduction

Shorebirds are known to forage extensively on benthic macroinvertebrates [1, 2] and the abundance of benthic macroinvertebrates is likely to be crucial for the short- and long-term survival of shorebirds given their high energy demands during migration and the breeding season [3–5]. Shorebirds are therefore predicted to show numerical and functional responses to changes in the abundance of their prey [6, 7]. Migratory shorebirds depend on stopover sites along their migration routes to rest and replenish their energy reserves [8, 9]. Many factors affect the distribution of their prey species, including water and sediment quality and aquatic plants [10, 11], which, in turn, affects the aggregation patterns of foraging shorebirds.

Wuliangsuhai Lake is a key wetland in the vast, arid region of northwest China and provides important breeding and staging habitats for shorebirds in the East Asian–Australasian Flyway [12]. However, as the major area for water storage and the discharge of agricultural drainage in the Yellow River Bend Region [13], the wetland is highly eutrophic [14]. Subsequently, the structure and function of this wetland have gradually changed—for example, the rapid expansion of *Phragmites* into open water areas has been well documented [15, 16]. Structural changes in the flora may modify the distribution and abundance of benthic macroinvertebrates [17] and ultimately affect the wetland's function as a feeding ground for shorebirds. However, the relationship between shorebirds and their prey at Wuliangsuhai Lake has not been investigated on any scale and the numerical response of shorebirds to the variation in benthic macroinvertebrates remains unknown.

This study investigated the spatial association of shorebirds and their food resources within Wuliangsuhai Lake. We conducted regular shorebird surveys at three main foraging sites along environmental gradients during the 2011 autumn migration season and sampled the benthic macroinvertebrates and collected data on environmental variables along

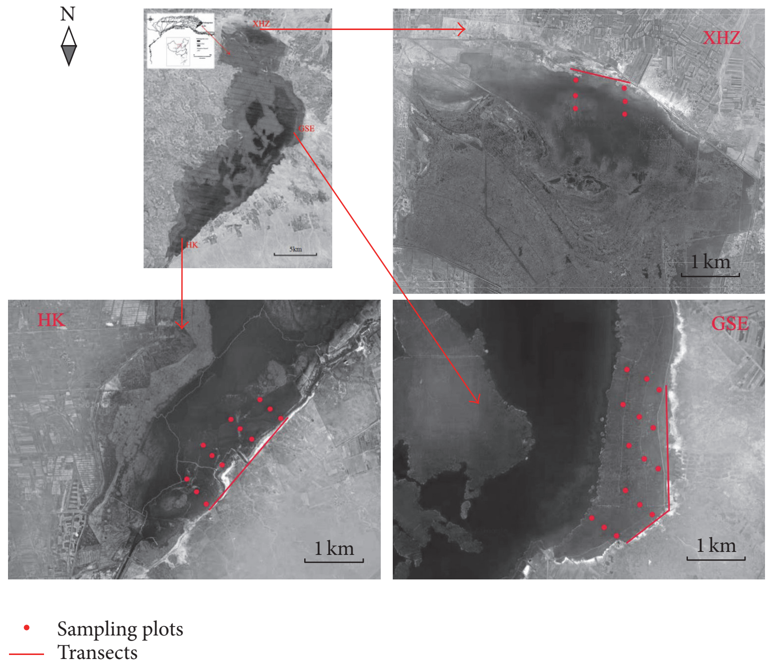

Figure 1: Location of Wuliangsuhai Lake and the arrangement of shorebird survey transects (lines) and sampling plots for macroinvertebrates and environmental variables (dots) within the study areas of Gesuer (GSE), Hekou (HK), and Xiaohaizi (XHZ).

the bird survey transects. We aimed to answer two specific questions: (1) what are the important environmental factors for benthic macroinvertebrates and (2) how do shorebirds respond to the changes in their food sources (i.e., the benthic macroinvertebrate density)? This information is crucial in the management of the wetland, particularly as a habitat for migratory waterbirds.

2. Materials and Methods

2.1. Study Site. Wuliangsuhai Lake is located in Inner Mongolia, China (40°46′–41°05′N, 108°42′–108°58′E) (Figure 1), and has an area of 293 km^2 and a mean water depth of about 1 m [17]. The prevailing climate is continental with a low mean annual rainfall (220 mm), high annual evaporation (1,502 mm), and a mean annual temperature of 7.0°C [18]. The wetland is frozen from November to March and the frost-free period averages 152 days [19]. The main water source is agricultural drainage from the Hetao Irrigation Area. Recent studies have shown that the concentrations of total nitrogen (TN) and total phosphorus (TP) range from 1.44 to 19.31 and 0.024 to 0.057 mg/L, respectively, indicating that the wetland is highly eutrophic [20, 21].

About 50.9% of the wetland surface area is covered by emergent plants and 43.7% by submerged plants; the remaining 5.4% consists of sandbars and shoals [22]. The wetland is recognized as one of the most important areas for birds in the vast, arid region of northwest China. A total of 241 bird species has been recorded from 17 orders and 49 families, including black stork *(Ciconia nigra)*, relict gull *(Larus relictus)*, and mute swan *(Cygnus olor)* [23]. Shorebirds of global conservation importance include far eastern curlew *(Numenius madagascariensis)*, Asian dowitcher *(Limnodromus semipalmatus)*, black-tailed godwit *(Limosa limosa)*, and Eurasian curlew *(Numenius arquata)* [12].

2.2. Bird Survey. To investigate the possible associations between shorebirds and benthic macroinvertebrates and to test whether there were variations in the abundance of shorebirds in space and time, data on shorebirds were collected from biweekly surveys during the 2011 autumn migration (late August to early November 2011; eight observation periods) [12]. For the shorebirds, we focused on the families Scolopacidae and Charadriidae because these are the main groups of shorebirds that visit Wuliangsuhai Lake annually [12, 24]. The surveys were conducted at three main shorebird foraging sites in the wetland: Xiaohaizi (XHZ), Gesuer (GSE), and Hekou (HK) (Figure 1). The sites were chosen to include a range of environmental gradients (e.g., water and sediment quality), although the choice of sites was also influenced by accessibility and safety considerations. XHZ is located in the northernmost part of the wetland, near where drainage water flows into the lake. This area is largely covered by emergent plants, with small patches of shallow water and sandbars. GSE is located on the eastern shore and is mainly covered by emergent and submerged plants; it has the largest proportion of shallow water (depth < 1 m) among the three sampling sites. HK is located in the southernmost part of the wetland where the water flows into the Yellow River. It consists of a large area of open water with an area of shallow water near the shore. All three sampling sites provide foraging and resting habitats for shorebirds [12, 22]. A fixed transect with variable lengths was established at each sampling site. Based on the size of the habitat (about 5 km^2 at GSE, 4 km^2 at HK, and

2 km^2 at XHZ), the transect length was 4, 3, and 1 km for GSE, HK, and XHZ, respectively. Each survey was conducted by walking the length of the transect parallel to the lakeshore at a constant speed; the perpendicular searching distance was 0.6 km. Using binoculars (8 × 42) and telescopes (Swarovski ATS 80 HD 20–60 × 80), we counted and recorded all the shorebirds (in sight or heard). To increase detectability, the surveys were carried out in daytime on clear days and there were at least two fully trained observers for each transect. The variability in observer error was minimized by using the same observers whenever possible throughout the study period. To avoid repeated counting, the three transects were surveyed simultaneously. Visual and/or verbal communication enabled us to avoid duplicate recordings of the same flock of shorebirds by at least two observers. Shorebirds that flew forward were excluded and all the shorebirds present were identified to species level.

2.3. Benthic Macroinvertebrate Survey. Benthic macroinvertebrate samples were collected in the shallow water area of the lake (along the shorebird survey transects) using PVC pipes (7.14 cm diameter) on 17–19 September and 16-17 October 2011. The sampling dates were chosen to reflect the shorebird phenology and represent the "peak" and "postpeak" dates in the autumn migration. As most shorebird species at Wuliangsuhai Lake are unable to forage on prey that is distributed in sediment deeper than 10 cm [24], only the top sediment layer was collected and analyzed. This depth was sufficient to capture most if not all benthic macroinvertebrates that serve as shorebird prey at Wuliangsuhai Lake. Sediment cores were taken parallel to the transects at regular 100 m intervals and sampling was repeated every 1 km (Figure 1). The number of sediment cores collected during each sampling period for the three sampling sites was six, 15, and 12 for XHZ, GSE, and HK, respectively, giving a total of 66 samples. Duplicate samples were collected and the sediment cores were sliced into top (0–5 cm) and bottom (5–10 cm) layers, washed, and sorted using a 63 μm sieve [25]. The remaining material was preserved in 95% ethanol and examined microscopically. All individual organisms were counted and because the counts of benthic macroinvertebrate species other than Chironomidae and Lymnaeidae (e.g., the larvae of Tabanidae) were very low, they were grouped together and labeled as "Others" to give three major groups of benthic macroinvertebrate: Chironomidae, Lymnaeidae, and Others. We followed the methods of Epler to identify chironomid larvae [26]. The methods of Merritt et al. [27] and Liu et al. [28] were used to identify other species.

2.4. Substrate Quality and Water Quality. We collected substrate samples at each benthic sampling location using PVC pipes with the same diameter as the macroinvertebrate sampling tubes. The substrate sampling was conducted from 17 to 19 September 2011 and a total of 66 substrate samples were collected, the same number of samples as for the benthic macroinvertebrates. The substrate cores were also divided into top (0–5 cm) and bottom (5–10 cm) layers. The samples were analyzed in the laboratory of the Institute of Geographic Sciences and Natural Resources Research, Chinese Academy of Sciences (Beijing, China) for nitrogen (N), phosphorus (P), organic matter (OM), and heavy metals (As, Co, Cu, Li, Ni, and Hg). The N was determined using an elemental analyzer (Vario MACRO cube, Elementar, Germany). The OM was determined by titration with $K_2Cr_2O_7$. The concentrations of P, Co, Cu, Li, and Ni were determined by inductively coupled plasma optical emission spectrometry using an Optima 5300 DV spectrometer (Perkin-Elmer, USA) and the concentrations of As and Hg were determined by inductively coupled plasma mass spectrometry (Elan DRC-e, USA). The substrate was classified into three types based on the particle diameter: sand, sand-mud, and mud.

Water quality measurements were taken at the same time as the benthic macroinvertebrate samples from 17 to 19 September 2011. The in situ water temperature, pH, and dissolved oxygen were measured with a YSI Professional Plus handheld multiple parameter meter (YSI, USA). Chlorophyll a (Chl a) was measured on-site with a Hydrolab MS 5 instrument (HACH, USA). Water samples were collected and preprocessed on site for other parameters, including salinity, TN, and TP. All water samples were processed in the laboratory of Urat Front Banner Environmental Protection Monitoring Station (Inner Mongolia, China) on the sampling day using a TU-1810 UV-VIS spectrophotometer (Persee Incorporated, China).

2.5. Statistical Analyses. We used the nonparametric Kruskal-Wallis test to compare the differences in the abundance of shorebirds, benthic macroinvertebrates, and environmental variables between sampling sites. The Mann–Whitney U test was applied to the benthic macroinvertebrates and quality of substrates collected from different layers to compare the distribution of the variables at different depths.

We applied multiple permutation linear regression to explore the relationship between the abundance of shorebirds and benthic macroinvertebrates. Model selection was based on the resultant p value and the Akaike information criterion [29]. A one-sample Kolmogorov-Smirnov test was used to test the normality of shorebird abundance. The test indicated that the data slightly violated normality ($p = 0.087$). In consideration of the small sample size and possible outliers, we used the permutation test instead of the normal theory test for statistical inference [30].

The distribution variation of benthic macroinvertebrates with environmental gradients was examined using canonical correspondence analysis (CCA) [31] on the $\log(x + 1)$-transformed benthic macroinvertebrate density because this allows a quick appraisal of how the community composition varies with environmental gradients. The densities of the three groups of benthic macroinvertebrates at GSE, HK, and XHZ in relation to environmental data were included in the analysis. We pooled the benthic macroinvertebrate density data of the three sediment cores in one sampling transect and used it as the density of the site in the CCA analysis. Thus, we had a total of five sites at GSE, two sites at XHZ, and four sites at HK. Before the CCA, we tested the correlations among these variables using Pearson's correlation coefficient and the strongly correlated variables (i.e., Pearson's $r > 0.75$)

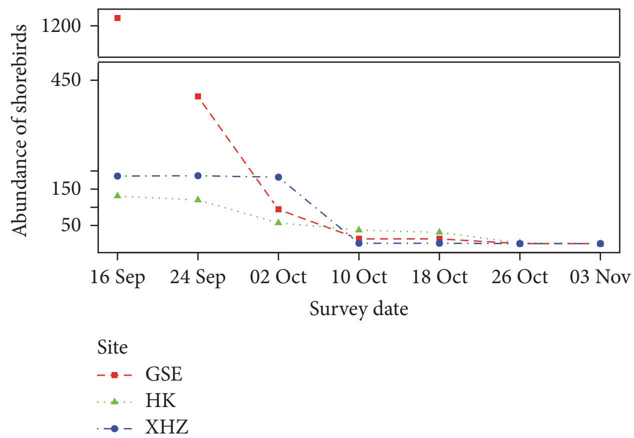

Figure 2: Number of shorebirds observed at the three sampling sites of Gesuer (GSE), Hekou (HK), and Xiaohaizi (XHZ) during the autumn migration.

Table 1: Mean ± SD benthic macroinvertebrate densities from the counts at Wuliangsuhai Lake during the autumn migration in 2011 ($n = 33$).

Time period	Chironomidae	Lymnaeidae	Others
Peak migration	5,168 ± 13,392	205 ± 490	90 ± 182
Postpeak migration	1,303 ± 1,842	0 ± 0	0 ± 0

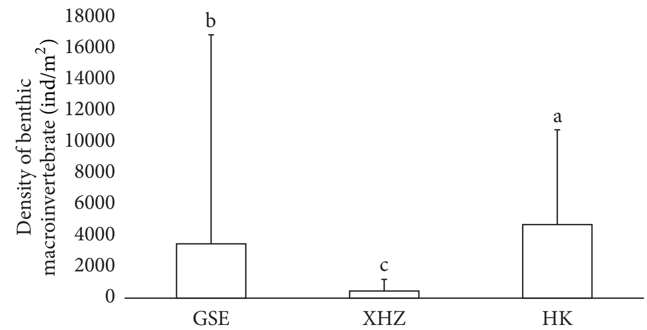

Figure 3: Density of benthic macroinvertebrate (mean ± SD) at the three sampling sites of Gesuer (GSE), Xiaohaizi (XHZ), and Hekou (HK) ($n = 30$, 12, and 24 for GSE, XHZ, and HK, respectively).

were excluded, including dissolved oxygen, Chl a, TN, OM, N, As, Co, Li, Ni, and Hg. A permutation test was used to investigate the correlations between benthic macroinvertebrates and environmental variables. All statistical analyses were performed in R3.2.2 [32].

3. Results

3.1. Spatial and Temporal Distribution of Shorebirds. A total of 27 shorebird species with over 3,300 individuals were recorded in the 2011 autumn migration season at Wuliangsuhai Lake. The three most abundant species, black-tailed godwit, spotted redshank *(Tringa erythropus)*, and Pacific golden plover *(Pluvialis fulva)* comprised 83% of the shorebird population; Northern lapwing *(Vanellus vanellus)*, Kentish plover *(Charadrius alexandrinus)*, and black-winged stilt *(Himantopus himantopus)* made up another 13% of the shorebirds. The total number of shorebirds decreased sharply after September and then decreased gradually until the end of migration season on 6 November 2011 (Figure 2).

The number of shorebirds at the three sampling sites was significantly different (Kruskal-Wallis test, $p = 0.030$). At the early stage of the study (in September), the highest number of shorebirds was observed at GSE, followed by XHZ and HK, with 68.57, 16.65, and 14.78% of the total count, respectively. More birds were counted at GSE and XHZ than at HK before 10 October 2011, after which the opposite distribution pattern was observed (Figure 2).

3.2. Spatial and Temporal Distribution of Benthic Macroinvertebrates. The most abundant prey in the wetland during both the peak and postpeak shorebird migration was Chironomidae (Table 1).

There were large spatial (horizontal and vertical) and temporal variations in the distributions of benthic macroinvertebrates. Spatially, the Kruskal-Wallis test showed that the density of benthic macroinvertebrates varied significantly between the three sampling sites ($p < 0.001$), with HK having the highest density and XHZ the lowest (Figure 3). The Mann–Whitney U test showed a significant vertical difference, with a significantly higher density of benthic macroinvertebrates in the top layer of sediment than in the bottom layer ($p < 0.001$) (Figure 4). Temporally, the average density of benthic macroinvertebrates in mid-September 2011 (5460 indiv./m^2) was substantially higher than in mid-October 2011 (1000 indiv./m^2) ($p = 0.005$). Although Lymnaeidae were largely found in coarse sediments (sand), Chironomidae were mainly found in fine sediments (mud). Sandy-mud could still support a certain amount of Chironomidae (Figure 5).

3.3. Response of Shorebirds to the Distribution of Benthic Macroinvertebrates. Permutation linear regression analyses showed that among the three groups of benthic macroinvertebrates, only Chironomidae showed a positive correlation with shorebird abundance ($p = 0.049$). The model selection procedure using the Akaike information criterion indicated that Lymnaeidae were needed in the regression, although the permutation test suggested that its effect was nonsignificant ($p = 0.074$). The densities of Chironomidae and Lymnaeidae together explained 53.63% of the variation in the abundance of shorebirds at Wuliangsuhai Lake during the autumn migration ($r^2 = 0.536$, $p = 0.046$).

3.4. Environmental Factors Influencing the Abundance and Distribution of Benthic Macroinvertebrates

3.4.1. Water Quality. A wide range of environmental gradients was evident across the three sampling sites, most noticeably salinity ($p = 0.022$, Table 2), ranging from brackish to supersaline (5,887–42,600 mg/L). A range of

Table 2: Mean ± SD values of the principal water quality characteristics across the three sampling sites within the lake: Gesuer (GSE), Hekou (HK), and Xiaohaizi (XHZ) ($n = 11$).

Sampling site	Water temperature (°C)	pH	Dissolved oxygen (%)	Chlorophyll a (mg/L)	Salinity (mg/L)	Total nitrogen (mg/L)	Total phosphorus (mg/L)
GSE	19.32 ± 0.96	8.87 ± 0.10	96.92 ± 15.34	12.41 ± 2.30	29,660 ± 13,076	4.36 ± 0.59	0.282 ± 0.032
HK	15.98 ± 1.15	9.34 ± 0.54	76.8 ± 15.0	4.04 ± 1.04	5,887 ± 537	1.47 ± 0.09	0.070 ± 0.011
XHZ	19.15 ± 0.75	9.04 ± 0.02	83.3 ± 16.7	7.80 ± 3.02	42,600 ± 2,500	12.4 ± 1.0	0.356 ± 0.000

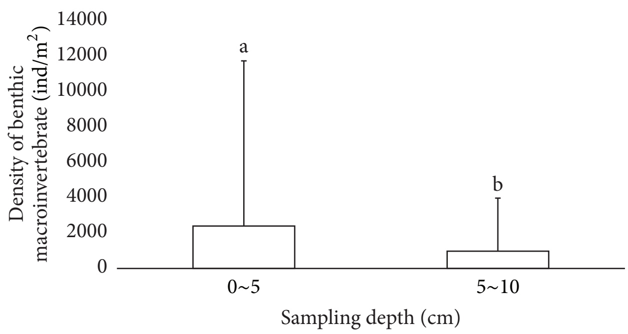

Figure 4: Density of benthic macroinvertebrates (mean ± SD) at sediment depths of 0–5 and 5–10 cm ($n = 66$).

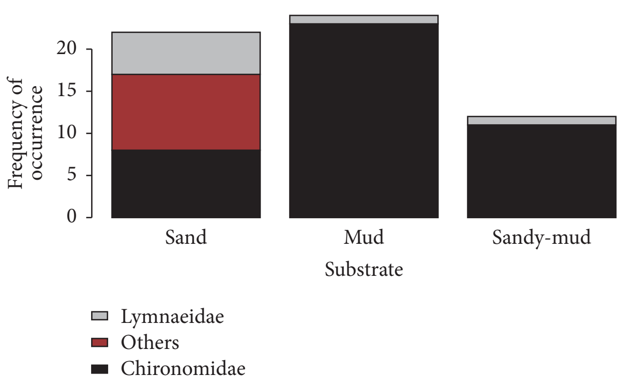

Figure 5: Frequency of occurrence of benthic macroinvertebrates in different substrate types during benthic coring within each sample plots ($n = 66$).

concentrations of Chl a ($p = 0.024$), TN ($p = 0.014$), and TP ($p = 0.013$) was also evident, with the concentrations at GSE (12.41, 4.36, and 0.282 mg/L for Chl a, TN, and TP, respectively) and XHZ (7.80, 12.4, and 0.356 mg/L for Chl a, TN, and TP, respectively) significantly higher than those at HK (4.04, 1.47, and 0.070 mg/L for Chl a, TN, and TP, respectively). The water temperature at GSE and XHZ was 3.00°C higher than at HK ($p = 0.030$). The range of dissolved oxygen concentrations was not significantly different among three sampling sites (76.8–96.92%, $p = 0.377$), nor was the pH (8.87–9.34, $p = 0.187$) (Table 2).

3.4.2. Substrate Quality. The concentrations of N, P, and OM in the sediment varied significantly among the three sampling sites ($p = 0.042$, $p < 0.001$, and $p < 0.001$, respectively), with the concentration of N and OM highest at HK and the concentration of P highest at GSE. The concentrations of all the heavy metals ($p < 0.001$), except Hg ($p = 0.051$), were significantly different among the three sampling sites, with the concentrations at XHZ and GSE higher than those at HK. The concentrations of Li in the upper layer of the sediments were significantly higher than in the bottom layer ($p = 0.010$), whereas the concentrations of Co and Ni were higher in the deeper sediments ($p = 0.036$) (Table 3).

3.4.3. Effects of Environmental Variables on Benthic Macroinvertebrates. Six environmental variables were included in the final CCA (Figure 6). The first two canonical axes collectively explained 87.5% of the variance in benthic macroinvertebrate distribution (60.2 and 27.3% for axes 1 and 2, respectively). Water temperature, salinity, TP, and Cu were significantly positively correlated with axis 2 (Table 4) and axis 2 was positively associated with Chironomidae (Figure 6). From Figure 6, it is clear that Chironomidae were mainly found at sampling sites with a higher water temperature and a higher concentration of both water TP and sediment P, but with a lower concentration of Cu and a lower water pH. By contrast, Lymnaeidae were largely found at sites with a higher water pH, but with a lower water temperature, lower water TP and salinity, and sediment P. Other macroinvertebrate species (e.g., the larvae of Tabanidae and Tubificidae) mainly occurred at sites with higher concentrations of Cu and higher salinity (Figure 6). The permutation test also indicated that there was a significant relationship between macroinvertebrate density and water/sediment quality ($p = 0.013$).

4. Discussion

4.1. Factors Influencing the Abundance and Distribution of Benthic Macroinvertebrates. The abundance of benthic macroinvertebrates was generally lower at sites that had poorer water and sediment quality, with the abundance decreasing from HK (near the outlet with higher water and sediment quality than other sites), to XHZ (near the drainage inlet with the highest concentrations of nutrients and pollutants). A similar tendency has been reported within the wetland, with the density and biomass of macroinvertebrates increasing with increasing distance from the main sources of pollution [33]. Among the recorded benthic macroinvertebrates, Chironomidae and Lymnaeidae had higher abundances at sites close to the outlet of the wetland where the salinity, TN, TP, and heavy metal (in particular Cu) concentrations were much lower than at XHZ and GSE. The pattern

Table 3: Mean ± SD values of the principal substrate quality characteristics across the three sampling sites within the lake (Gesuer (GSE), Hekou (HK), and Xiaohaizi (XHZ)) in sediments from depths of 0–5 and 5–10 cm (n = 15, 12, and 6 for GSE, HK, and XHZ, respectively).

Substrate characteristic	GSE		HK		XHZ	
	0–5 cm	5–10 cm	0–5 cm	5–10 cm	0–5 cm	5–10 cm
Organic matter (%)	1.33 ± 0.62	1.17 ± 0.43	3.22 ± 1.36	1.97 ± 0.85	2.30 ± 1.03	1.59 ± 0.54
N (mg/kg)	1362 ± 703	717 ± 202	1508 ± 637	1403 ± 863	1134 ± 370	1001 ± 242
P (mg/kg)	916.2 ± 69.6	972.30 ± 83.84	835.85 ± 59.16	818.98 ± 54.06	804.53 ± 29.72	866.23 ± 54.65
As (mg/kg)	21.14 ± 5.81	25.98 ± 7.56	9.20 ± 1.99	12.30 ± 3.60	29.68 ± 4.26	35.30 ± 5.98
Co (mg/kg)	10.67 ± 2.30	12.20 ± 2.44	8.92 ± 1.39	9.46 ± 1.13	11.49 ± 0.52	12.81 ± 0.59
Cu (mg/kg)	26.63 ± 6.19	30.93 ± 10.82	19.69 ± 4.10	21.12 ± 3.51	33.26 ± 3.84	37.20 ± 11.71
Li (mg/kg)	40.60 ± 5.47	44.46 ± 6.72	37.08 ± 8.14	37.44 ± 6.51	49.78 ± 2.20	51.97 ± 3.27
Ni (mg/kg)	30.28 ± 6.72	34.61 ± 7.59	26.74 ± 4.64	27.94 ± 4.29	31.25 ± 1.48	34.63 ± 2.89
Hg (mg/kg)	0.01 ± 0.00	0.02 ± 0.00	0.01 ± 0.01	0.01 ± 0.00	0.01 ± 0.00	0.01 ± 0.00

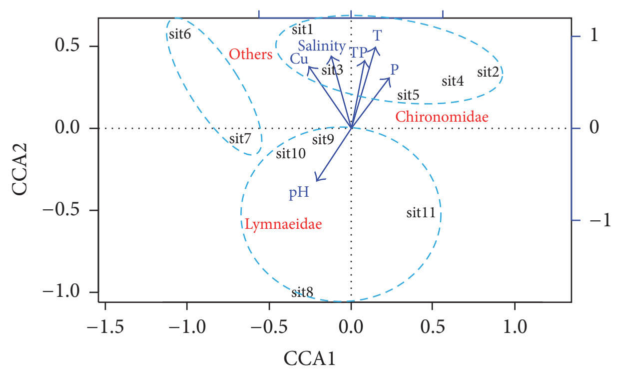

Figure 6: Plot of first two axes of canonical correspondence analysis (CCA) ordination based on the log-transformed benthic macroinvertebrate density data and environmental variables (arrows) at different sites.

Table 4: Correlation coefficients of environmental variables with the first two axes of canonical correspondence analysis.

Environmental variable	Axis	
	1	2
pH	−0.58418	−0.81162
Water temperature	0.31746	0.94827**
Salinity	−0.29459	0.95562*
Total phosphorus	0.20732	0.97827*
P	0.63746	0.77048
Cu	−0.60334	0.79748*

Significance at $^{*}p < 0.05$ and $^{**}p < 0.01$.

was particularly clear for Lymnaeidae, which was absent from the majority of sampling plots at XHZ. This indicates that Lymnaeidae have a relatively higher requirement for water and substrate quality than Chironomidae and that high levels of salinity and TP or heavy metals could restrict the survival of Lymnaeidae [34]. Some species of Chironomidae larvae can tolerate very high levels of pollution [35, 36] and they often become the dominant group of benthic invertebrates in polluted water bodies such as Wuliangsuhai Lake.

Benthic invertebrates are strongly affected by environmental stress, such as inorganic contaminants [37]—for example, the accumulation of heavy metals in sediments could lead to the death of benthic invertebrates [38]. Several effects of metal contamination on benthic communities have been documented, including decreased density [39, 40], a reduction in the number of sensitive taxa [41], and changes in the distribution patterns of species [42]. Our study suggested that water temperature, salinity, TP, and Cu concentrations in sediments were the most important variables in determining the density and distribution of benthic macroinvertebrate. Other variables, such as water pH and the P concentration in the sediment, might be less important for benthic macroinvertebrate at Wuliangsuhai Lake. pH does not play an important part in determining the distribution of Lymnaeidae in natural water bodies [34]. Although the CCA indicated that the sediment P concentration did not have a positive relationship with the density of benthic macroinvertebrates, this result should be interpreted with caution. Because the sediment P and water TP were highly positively correlated, which is consistent with the study of Sun et al. [43], the negative relationship between the water column TP and the abundance of benthic macroinvertebrates might imply a negative relationship between the sediment P and the abundance of benthic macroinvertebrates. Previous studies have shown that the amount of sediment P at Wuliangsuhai Lake may be a threat to the benthic communities and may have exacerbated the eutrophic level of the water body [44].

The concentration of Cu had a more significant effect on the benthic macroinvertebrates than other heavy metals. In a study of benthic fauna and pollutant levels in Norwegian fjords, Brage [45] also found that Cu had the highest deleterious effect. The high Cu level in the top 5 cm of sediments sampled from GSE and XHZ was evidence of slight to moderate heavy metal pollution [46]. The substrate type also played an important part in determining the species, abundance, and distribution of benthic invertebrates [47]. Chironomidae larvae preferred muddy substrates, whereas Lymnaeidae preferred sandy substrates at Wuliangsuhai Lake.

The loading of nutrients (especially TP) and heavy metals (in particular Cu) into the aquatic environment has increased with the intensity of human activities. The input of nutrients into Wuliangsuhai Lake is mainly from irrigation drainage water [20] and the source of heavy metals is mainly from industrial wastewater and domestic sewage around the wetland [48]. As a result, the diversity and distribution of benthic macroinvertebrates may have changed. Some species may even disappear locally when pollution exceeds their tolerance level. For example, Lymnaeidae were absent from most sampling plots at XHZ. As demonstrated in this and many previous studies [49–51], shorebirds positively follow the distribution pattern of benthic macroinvertebrates and therefore changes in the benthic macroinvertebrate communities might eventually lead to the functional loss of Wuliangsuhai Lake as a stopover site for migratory shorebirds. Reducing the nutrient loadings and controlling water pollution are crucial for waterbird conservation at Wuliangsuhai Lake.

4.2. Distribution of Shorebirds and Relationship with Benthic Macroinvertebrates. Our results showed that the distribution of shorebirds at Wuliangsuhai Lake was not random, with the greatest abundance of shorebirds at GSE, the main foraging habitat during peak migration. However, in mid-October 2011, after the peak season when most shorebirds had left, HK provided the main foraging habitat for late migrants [12] (Figure 2). These two main foraging areas also had the highest abundance of benthic macroinvertebrates (Figure 3). Numerous studies have shown positive relationships between shorebirds and prey abundance or biomass, but the strength of the relationship varies among studies [52, 53]. Our study confirmed the numerical relationship between shorebirds and benthic macroinvertebrates within Wuliangsuhai Lake, even though the relationship was not very strong (the r^2 value of the multiple linear regression was moderate 53.63%), which may be due to the relatively small sample size (only during autumn migration). Prey density alone is unlikely to account for all the variation in bird density [54]—for example, GSE had the highest average shorebird density, but not the highest average prey density. Other variables, such as vegetation cover [55] and prey availability, which are associated with water depth [56], should be incorporated into future studies. Determination of the dynamic interaction between shorebirds and the density of benthic macroinvertebrates (e.g., sites with more abundant macroinvertebrates would attract more shorebirds, which, in turn, could suppress the abundance of macroinvertebrates) would give a better insight into the ecological communities and help in the conservation of the wetland ecosystem.

Many shorebird species are long-distance migrants and require high-quality stopover sites to rest and refuel for their next journey. One of the most important factors is the availability of food at the stopover sites [57]. It is also known that the interactions among shorebirds, their prey, and the environment are important [6, 58–61] and provide basic information for the management of shorebird habitats. The populations of long-distance migratory shorebirds around the world are decreasing [62] and some of the steepest and most widespread declines have been observed in the East Asian-Australasian Flyway [63]. Most of the earlier studies have focused only on shorebirds and their habitats in the coastal zones of the flyway [5, 64, 65] and there is little information available on inland wetlands. Our study focused on the numerical relationship between shorebirds and benthic macroinvertebrates in a temperate arid wetland and provides important data for future studies.

5. Conclusions

The abundance of both shorebirds and benthic macroinvertebrates at Wuliangsuhai Lake varied significantly in space and time. Environmental factors, including the water and substrate quality, had strong impacts on the abundance and distribution of benthic macroinvertebrates. The shorebirds positively followed the distribution patterns of the benthic macroinvertebrates. Thus, the numerical relationship between shorebirds and benthic macroinvertebrates within Wuliangsuhai Lake was confirmed. Our findings suggest that reducing the nutrient and heavy metal loadings is crucial in maintaining the ecological function of Wuliangsuhai Lake as a foraging and stopover habitat for migratory shorebirds.

Competing Interests

The authors declare that there is no conflict of interests regarding the publication of this paper.

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References

[1] P. Evans and P. Dugan, *Coastal Birds: Numbers in Relation to Food Resources*, Cambridge University Press, Cambridge, UK, 1983.

[2] K. Grond, Y. Ntiamoa-Baidu, T. Piersma, and J. Reneerkens, “Prey type and foraging ecology of Sanderlings *Calidris alba* in different climate zones: are tropical areas more favourable than temperate sites?” *PeerJ*, vol. 3, no. 8, Article ID e1125, 2015.

[3] P. R. Evans, “Energy balance and optimal foraging strategies in shorebirds: some implications for their distributions and movements in the non-breeding season,” *Ardea*, vol. 64, pp. 117–139, 1976.

[4] T. Piersma, “Energetic bottlenecks and other design constraints in avian annual cycles,” *Integrative and Comparative Biology*, vol. 42, no. 1, pp. 51–67, 2002.

[5] H.-Y. Yang, B. Chen, Z.-J. Ma et al., “Economic design in a long-distance migrating molluscivore: how fast-fuelling red knots in Bohai Bay, China, get away with small gizzards,” *Journal of Experimental Biology*, vol. 216, no. 19, pp. 3627–3636, 2013.

[6] M. A. Colwell and S. L. Landrum, "Nonrandom shorebird distribution and fine-scale variation in prey abundance," *The Condor*, vol. 95, no. 1, pp. 94–103, 1993.

[7] C. Holling, "A model of the functional response of predator to prey density involving the hunger effect," *The Canadian Entomologist*, vol. 91, pp. 385–398, 1959.

[8] J. A. Alves, T. G. Gunnarsson, D. B. Hayhow et al., "Costs, benefits, and fitness consequences of different migratory strategies," *Ecology*, vol. 94, no. 1, pp. 11–17, 2013.

[9] I. Newton, *The Migration Ecology of Birds*, Academic Press, 2010.

[10] J. A. Alves, W. J. Sutherland, and J. A. Gill, "Will improving wastewater treatment impact shorebirds? Effects of sewage discharges on estuarine invertebrates and birds," *Animal Conservation*, vol. 15, no. 1, pp. 44–52, 2012.

[11] H. Rousi, H. Peltonen, J. Mattila, S. Bäck, and E. Bonsdorff, "Impacts of physical environmental characteristics on the distribution of benthic fauna in the northern Baltic sea," *Boreal Environment Research, Supplement B*, vol. 16, p. 521, 2011.

[12] Z. Yamian, J. Yifei, J. Shengwu et al., "Wuliangsuhai wetlands: a critical habitat for migratory water birds," *Journal of Resources and Ecology*, vol. 3, no. 4, pp. 316–323, 2012.

[13] W. P. Li, C. Y. Li, X. H. Shi, and B. Cui, "Analysis of distribution features of nitrogen and phosphorus nutritous elements and their geochemical environment for Wulangsu Lake, Inner Mongolia," *Resources Survey & Environment*, vol. 29, no. 2, pp. 131–138, 2008.

[14] Q. Luo and C. Y. Li, "Seasonal variation and distribution of chlorophyll-a in WuliangsuHai Lake," *Water Saving Irrigation*, no. 2, pp. 36–39, 2011.

[15] Y. Cao, S. Shang, J. Yang, L. Wu, H. Liu, and K. Luan, "Research on Spatial and Temporal Dynamic Evolution of Wuliangsuhai Lake Wetland," *Progress in Geography*, vol. 29, no. 3, pp. 307–311, 2010.

[16] Y. Ruihong, L. Tingxi, X. Youpeng, and L. Changyou, "The impacts of human activities on the Wuliangsuhai wetland environment," *Journal of Lake Sciences*, vol. 19, no. 4, pp. 465–472, 2007.

[17] Y. Q. Pan, L. L. Xing, and G. S. Yang, "A preliminary study on Avifauna's evolution in Wuliangsuhai Wetland during the last 10 years," *Acta Scientiarum Naturalium Universitatis NeiMongol*, vol. 37, pp. 170–174, 2006.

[18] Z. Y. Liu, *Research about Estimating the Input to the Lake Wuliangsuhai from Farmland Surface Pollution*, Inner Mongolia University, Hohhot, China, 2004.

[19] Y. X. Qiu, *The Study on the Issue of Harnessing Wuliangsu Lake by International Co-Operation*, Inner Mongolia Universtiy, Hohhot, China, 2007.

[20] Z. Wang, "Water body nitrogen and phosphorus pullution and control measures at Wuliangsuhai Lake," *Inner Mongolia Water Resources*, no. 2, pp. 8–9, 2013.

[21] B. W. Xiao, W. L. Cheng, R. Yao, H. M. Liu, and Y. H. Liu, "Study on variation trend of nitrogen and phosphorus in Wuliangsuhai lake of Inner Mongolia," *Journal of Water Resources & Water Engineering*, vol. 26, pp. 43–46, 2015.

[22] Q. Zeng, Y. Zhang, Y. Jia et al., "Zoning for management in wetland nature reserves: a case study using wuliangsuhai nature reserve, China," *SpringerPlus*, vol. 1, no. 1, p. 23, 2012.

[23] Y. Q. Pan, L. Li, and H. Y. Zhang, "The study of the ecological service functions assessment of awes habitat in Wuliangsuhai Wetland," *Northern Environment*, vol. 23, no. 1, pp. 36–38, 2011.

[24] L. L. Xing and G. S. Yang, *The Avifauna of Wuliangsuhai in Inner Mongolia*, Inner Mongolia Press, 1996.

[25] T. F. Nalepa, D. L. Fanslow, S. A. Pothoven, A. J. Foley III, and G. A. Lang, "Long-term trends in benthic macroinvertebrate populations in Lake Huron over the past four decades," *Journal of Great Lakes Research*, vol. 33, no. 2, pp. 421–436, 2007.

[26] J. H. Epler, "Identification manual for the larval Chironomidae (Diptera) of North and South Carolina," *Febs Letters*, vol. 81, no. 2, pp. 427–430, 2001.

[27] R. W. Merritt, K. W. Cummins, and M. B. Berg, *An Introduction to the Aquatic Insect of North America*, Kendall/Hunt, Dubuque, Iowa, USA, 2008.

[28] Y. Y. Liu, W. Z. Zhang, Y. X. Wang, and S. Y. Wang, *China's Economic Animals. Freshwater Mollusks*, Science Press, Beijing, China, 1979.

[29] H. Akaike, "A new look at the statistical model identification," *IEEE Transactions on Automatic Control*, vol. 19, no. 6, pp. 716–723, 1974.

[30] R. E. Wheeler, "Permutation tests for linear models in R," 2010.

[31] C. J. F. Ter Braak, "Canonical correspondence analysis: a new eigenvector technique for multivariate direct gradient analysis," *Ecology*, vol. 67, no. 5, pp. 1167–1179, 1986.

[32] R Develoment Core Team, *R: A Language and Environment for Statistical Computing*, R Foundation for Statistical Computing, Vienna, Austria, 2014.

[33] Z. H. He, Z. H. Xie, Y. Z. Lei, D. L. Sun, and Z. H. Liu, "A hydrochemical & hydrobiological survey of the lake Wuliang Su Hai," *Journal of Dalian Fisheries University*, vol. 1, no. 9, p. 84, 1984.

[34] N. Pullan, F. Climo, and C. Mansfield, "Studies on the distribution and ecology of the family lymnaeidae (Mollusca: Gastropoda) in New Zealand," *Journal of the Royal Society of New Zealand*, vol. 2, no. 3, pp. 393–405, 1972.

[35] S. K. Mousavi, R. Primicerio, and P.-A. Amundsen, "Diversity and structure of Chironomidae (Diptera) communities along a gradient of heavy metal contamination in a subarctic watercourse," *The Science of the Total Environment*, vol. 307, no. 1–3, pp. 93–110, 2003.

[36] G. Winberg, *Experimental Application of Various Systems of Biological Indication of Water Pollution [in USSR]*, PB US National Technical Information Service, 1978.

[37] J. S. Richardson and P. M. Kiffney, "Responses of a macroinvertebrate community from a pristine, Southern British Columbia, Canada, stream to metals in experimental mesocosms," *Environmental Toxicology and Chemistry*, vol. 19, no. 3, pp. 736–743, 2000.

[38] S.-E. Lee, D.-H. Yoo, J. Son, and K. Cho, "Proteomic evaluation of cadmium toxicity on the midge *Chironomus riparius* Meigen larvae," *Proteomics*, vol. 6, no. 3, pp. 945–957, 2006.

[39] T. P. Diggins and K. M. Stewart, "Chironomid deformities, benthic community composition, and trace elements in the Buffalo River (New York) area of concern," *Journal of the North American Benthological Society*, vol. 17, no. 3, pp. 311–323, 1998.

[40] R. W. Winner, M. W. Boesel, and M. P. Farrell, "Insect community structure as an index of heavy metal pollution in lotic ecosystems," *Canadian Journal of Fisheries and Aquatic Sciences*, vol. 37, no. 4, pp. 647–655, 1980.

[41] T. W. LaPoint, S. M. Melancon, and M. K. Morris, "Relationships among observed metal concentrations, criteria, and benthic community structural responses in 15 streams," *Journal of the Water Pollution Control Federation*, vol. 56, no. 9, pp. 1030–1038, 1984.

[42] W. H. Clements, D. S. Cherry, and J. H. Van Hassel, “Assessment of the impact of heavy metals on benthic communities at the Clinch River (Virginia): evaluation of an index of community sensitivity,” *Canadian Journal of Fisheries and Aquatic Sciences*, vol. 49, no. 8, pp. 1686–1694, 1992.

[43] H. M. Sun, J. He, X. D. Gao et al., “Distribution of total phosphorus in sediments of Wuliangsuhai Lake,” *Acta Sedimentologica Sinica*, vol. 24, no. 4, pp. 579–584, 2006.

[44] H. M. Sun, J. He, X. D. Gao, C. W. Lv, Q. Y. Fan, and H. X. Xue, “Distribution of total phosphorus in sediments of Wuliangsuhai Lake,” *Acta Sedimentologica Sinica*, vol. 24, no. 4, pp. 579–584, 2013.

[45] R. Brage, “Effect of sediment copper on benthic fauna,” *Marine Ecology Progress Series*, vol. 25, pp. 83–89, 1985.

[46] EPA (U.S. Environment Protection Agency), “Toxicological benchmarks for screening contaminants of potential concern for effects on sediment-associated biota,” Report of the Sediment Criteria Subcommittee, Science Advisory Board ES/ER/TM-95/R4.61, 1997.

[47] D. M. Bezmaternykh and O. N. Zhukova, “Composition, structure, and formation factors of bottom invertebrate communities in lakes of the southern Ob-Irtysh interfluve,” *Russian Journal of Ecology*, vol. 44, no. 2, pp. 170–177, 2013.

[48] S. N. Zhao, C. Y. Li, X. H. Shi, H. M. Zhang, and S. Wang, “Bioavailability and environment pollution evaluation of sediments heavy metals in Wuliangsuhai Lake,” *Ecology and Environmental Sciences*, vol. 22, no. 3, pp. 481–489, 2013.

[49] S. Guareschi, P. Abellán, A. Laini et al., “Cross-taxon congruence in wetlands: assessing the value of waterbirds as surrogates of macroinvertebrate biodiversity in Mediterranean Ramsar sites,” *Ecological Indicators*, vol. 49, pp. 204–215, 2015.

[50] R. J. Safran, C. R. Isola, M. A. Colwell, and O. E. Williams, “Benthic invertebrates at foraging locations of nine waterbird species in managed wetlands of the northern San Joaquin Valley, California,” *Wetlands*, vol. 17, no. 3, pp. 407–415, 1997.

[51] S. Werner, M. Mörtl, H.-G. Bauer, and K.-O. Rothhaupt, “Strong impact of wintering waterbirds on zebra mussel (*Dreissena polymorpha*) populations at Lake Constance, Germany,” *Freshwater Biology*, vol. 50, no. 8, pp. 1412–1426, 2005.

[52] W. Wolff, “Distribution of non-breeding waders in an estuarine area in relation to the distribution of their food organisms,” *Ardea*, vol. 57, pp. 1–28, 1969.

[53] L. Zwarts, A.-M. Blomert, B. J. Ens, R. Hupkes, and T. M. Van Spanje, “Why do waders reach high feeding densities on the intertidal flats of the Banc d’Arguin, Mauritania?” *Ardea*, vol. 78, no. 1-2, pp. 39–52, 1990.

[54] G. M. Puttick, *Foraging and Activity Patterns in Wintering Shorebirds*, 1984.

[55] C. S. Lopes, J. A. Ramos, and V. H. Paiva, “Changes in vegetation cover explain shifts of colony sites by little terns (*Sternula albifrons*) in Coastal Portugal,” *Waterbirds*, vol. 38, no. 3, pp. 260–268, 2015.

[56] E. Sebastián-González and A. J. Green, “Habitat use by waterbirds in relation to pond size, water depth, and isolation: lessons from a restoration in Southern Spain,” *Restoration Ecology*, vol. 22, no. 3, pp. 311–318, 2014.

[57] T. Piersma, “Using the power of comparison to explain habitat use and migration strategies of shorebirds worldwide,” *Journal of Ornithology*, vol. 148, no. 1, pp. S45–S59, 2007.

[58] J. E. Dugan, D. M. Hubbard, M. D. McCrary, and M. O. Pierson, “The response of macrofauna communities and shorebirds to macrophyte wrack subsidies on exposed sandy beaches of southern California,” *Estuarine, Coastal and Shelf Science*, vol. 58, pp. 25–40, 2003.

[59] S. E. A. L. V. D. Durell, R. A. Stillman, R. W. G. Caldow, S. Mcgrorty, A. D. West, and J. Humphreys, “Modelling the effect of environmental change on shorebirds: a case study on Poole Harbour, UK,” *Biological Conservation*, vol. 131, no. 3, pp. 459–473, 2006.

[60] G. L. Hamer, E. J. Heske, J. D. Brawn, and P. W. Brown, “Migrant shorebird predation on benthic invertebrates along the Illinois river, Illinois,” *Wilson Journal of Ornithology*, vol. 118, no. 2, pp. 152–163, 2006.

[61] F. L. Spruzen, A. M. M. Richardson, and E. J. Woehler, “Influence of environmental and prey variables on low tide shorebird habitat use within the Robbins Passage wetlands, Northwest Tasmania,” *Estuarine, Coastal and Shelf Science*, vol. 78, no. 1, pp. 122–134, 2008.

[62] International Wader Study Group, “Are waders world-wide in decline? Reviewing the evidence,” *Wader Study Group Bulletin*, vol. 101-102, pp. 8–12, 2003.

[63] T. Amano, T. Székely, K. Koyama, H. Amano, and W. J. Sutherland, “A framework for monitoring the status of populations: an example from wader populations in the East Asian-Australasian flyway,” *Biological Conservation*, vol. 143, no. 9, pp. 2238–2247, 2010.

[64] M. A. Barter and A. Riegen, “Northward Shorebird Migration Through Yalu Jiang National Nature Reserve,” vol. 46, pp. 9–14, 2004.

[65] J. K. Szabo, P. F. Battley, K. L. Buchanan, and D. I. Rogers, “What does the future hold for shorebirds in the East Asian–Australasian Flyway?” *Emu Official Organ of the Australasian Ornithologists Union*, vol. 116, article 2, 2016.