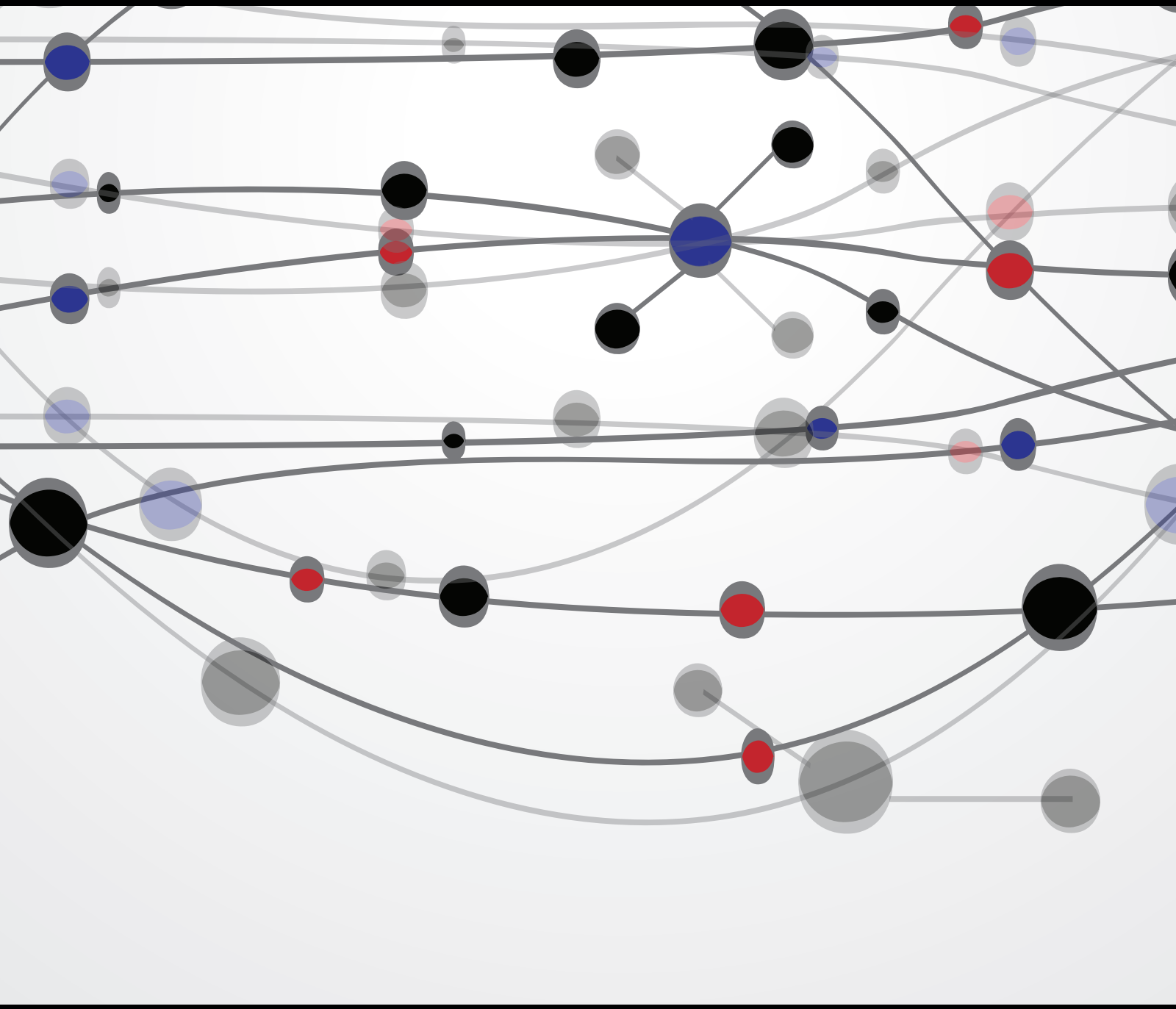
The Scientific World Journal

Wetland Degradation and Ecological Restoration

Guest Editors: Junhong Bai, Baoshan Cui, Huicong Cao, Ainong Li, and Baiyu Zhang



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Editorial

Wetland Degradation and Ecological Restoration

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Wetlands are among the most important ecosystems on earth and functioned as the "kidneys" of the earth, which play an important role in maintaining ecological service functions. However, with the rapid growth in human populations, wetlands worldwide are suffering from serious degradation or loss as affected by wetland pollution, wetland reclamation, civilization and land use changes, and so forth. Wetland degradation has potential influences on human health, biodiversity, regional climate, and regional ecological security. Therefore, it is an urgent task to recover these degraded wetlands. In recent years, wetland protection, restoration, and its reasonable exploitation have been paid much more attention to by most governments and researchers. Moreover, wetland restoration has become the frontier fields of wetlands science, which has been listed as one of important themes in these recent international wetlands and ecological conferences. Understanding wetland degradation processes can contribute to better effective wetland restoration. Therefore, we organized this special issue on "wetland degradation and ecological restoration." The objective of this special issue is to emphasize the effects of human activities on wetland ecosystems, the relationships between soil, water, and plant in wetlands, and wetland restoration issues and applications.

In total 25 submissions have been received from China and other countries until to the deadline date. However, only 11 papers were accepted to be published in this special issue on the basis of paper quality and reviewing process. The studied wetland types cover freshwater wetlands, coastal wetlands, lake wetlands, alpine wetlands and artificial wetlands. Although most contributors among these accepted manuscripts are from China except for 3 from Korea, France and Uganda, we still believe that this special issue can provide a platform to exhibit some related studies in such fields.

Among these accepted articles, Z. Luan et al. analyzed the response of *Carex lasiocarpa* in riparian wetlands to the environmental gradient of water depth by using the Gaussian model. Additionally, Luan et al. also showed readers the ecological impact of agricultural activity on marsh wetlands and assessed the functional efficacy. An ecological network for freshwater wetland conservation in the Central Yangtze Ecoregion was established and optimized based on large-scale gap analysis by X. W. Li et al. The polder effects of water table, residual available water capacity, and salt stress interdependence in coastal wetlands on sediment-to-soil conversion were investigated by R. Radimy et al. T. Seo et al. from Korea testified the functions of the littoral zone could be restored in an oligomesotrophic lake by installing an artificial vegetation island.

Q. Wang et al. reviewed the development of surface water quality models at three stages and gave some effective measures to standardize some surface water quality models. The eutrophication degree was also assessed in Baiyangdian Lake and the relationship between land use change and water quality was identified in Chaohu Lake by Wang et al. and Huang et al., respectively.

J. W. Kakuru et al. assessed the economic value of wetland resources and their contribution to food security in Uganda. A valuable way was established to overcome the gap that exists between the ability to construct computer simulation models to aid integrated land-use plan making and the demand for

them by planning professionals by H. Yu et al. B. Kong and H. Yu developed an estimation model of freeze-thaw erosion and applied it to the Silingco Watershed Wetland of Northern Tibet.

Of course, the wetland degradation and ecological restoration not only contain the above published topics, but also include the following potential themes, for example, biogeochemical processes of carbon, nitrogen, and phosphorous in wetlands; wetland ecohydrological process and ecological water requirement; ecological risk assessment of wetland ecosystems; heavy metal pollution in wetland soils; wetland landscape changes and climate change and so forth. However, this special issue has no way to cover all these topics due to many objective causes. We really hope the future issues can improve them.

Acknowledgments

The guest editors of this special issue would like to express appreciation to the authors for their excellent contributions and patience in assisting us. We also appreciate the fundamental work and valuable comments of all reviewers on these papers.

Junhong Bai
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Research Article

Total Economic Value of Wetlands Products and Services in Uganda

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Wetlands provide food and non-food products that contribute to income and food security in Uganda. This study determined the economic value of wetland resources and their contribution to food security in the three agroecological zones of Uganda. The values of wetland resources were estimated using primary and secondary data. Market price, Productivity, and Contingent valuation methods were used to estimate the value of wetland resources. The per capita value of fish was approximately US\$ 0.49 person^{-1}. Fish spawning was valued at approximately US\$ 363,815 year^{-1}, livestock pastures at US\$ 4.24 million, domestic water use at US\$ 34 million year^{-1}, and the gross annual value added by wetlands to milk production at US\$ 1.22 million. Flood control was valued at approximately US\$ 1,702,934,880 hectare^{-1} year^{-1} and water regulation and recharge at US\$ 7,056,360 hectare^{-1} year^{-1}. Through provision of grass for mulching, wetlands were estimated to contribute to US\$ 8.65 million annually. The annual contribution of non-use values was estimated in the range of US\$ 7.1 million for water recharge and regulation and to US\$ 1.7 billion for flood control. Thus, resource investment for wetlands conservation is economically justified to create incentives for continued benefits.

1. Introduction

Wetlands provide important natural resources, upon which the rural economy in Eastern Africa depends [1]. They provide many substantial benefits not only to local society, but also to the people who live far away from them. They are recognised globally for their vital role in sustaining a wide array of biodiversity and providing goods and services [2] and also as important sources of natural resources, upon which the rural economies depends [3].

In Uganda, wetlands provide a wide range of tangible and non-tangible benefits to various communities [4, 5]. The tangible benefits include water for domestic use and watering of livestock, support to dry season agriculture, provision of handicrafts, building materials, and food resources such as fish, yams, vegetables, wild game, and medicine. The nontangible benefits include flood control, purification of water, maintenance of the water table, microclimate moderation, and storm protection. Wetlands also serve as habitats for important flora and fauna, have aesthetic and heritage values, and contain stocks of biodiversity of potentially high pharmaceutical value [4, 5]. All these benefits have a bearing on food security.

Over 80% of the people living adjacent to wetland areas in Uganda directly use wetland resources for their household food security needs [6]. Besides, they also indirectly contribute to food security by providing services that foster food production such as weather modifications and nutrient retention. Food security exists when all people, at all times, have physical and economic access to sufficient, safe, and nutritious food that meets their dietary needs and food preferences for an active and healthy life [7]. The dimensions of food security include availability, access, and utilisation.

Wetland resources play a vital role in contributing to food security through the following:

(i) enabling direct availability of products such as fish, crops grown along the wetland edges, wild fruits and vegetables, and game meat;
(ii) providing cash income from sale of raw materials and processed products such as crafts, sand, clay, bricks, and ecotourism; which are sold for cash that is used for purchasing/accessing food; and
(iii) contributing to increased crop and livestock yields as a result of improved productivity from use of water, silt, and through climate moderation.

Each of the individual benefits or attributes of wetlands contributes to the household's output, welfare, or utility, thus making wetlands a recognised enabling sector to the economy of Uganda [8]. However, some of the benefits are marketed and can be allocated for monetary values, while others are used at subsistence level and do not have a direct reflection of their monetary values. This often makes it difficult to prioritise allocation of resources for the management and conservation of wetlands. This has led to continued degradation and low economic value attached to sustainable wetland resources management. To guide decisions on wetland management options, it is important to express the benefits derived from wetland resources in quantified monetary terms, as the basis for economic valuation. Wetland economic valuation is defined as a way of attaching quantitative and monetary values to wetland goods and services, whether or not market prices are available, so that they can be directly comparable with other sectors of the economy when activities are planned, policies are formulated, and decisions are made [9]. A better understanding of the benefits and costs of utilising wetland resources will provide important information for understanding and addressing the economic causes of wetland degradation and loss.

This study was undertaken to determine the economic values of wetland resources, to quantify wetland economic benefits and costs, to and determine the economic value of nonmarketed wetland goods and services in Uganda. The study highlights economic benefits in monetary terms for selected key wetland goods and services and demonstrates to wetland users, managers, and policy makers how valuable wetland resources are, as a basis for guiding decision making on wetland conservation.

2. Materials and Methods

2.1. The Study Area. The study was conducted in eight wetland systems located in areas that represent three of the five agroecological zones of Uganda. The wetlands are Nangabo, Mabamba, and Mende in Wakiso district representing the Lake Victoria crescent agroecological zone; Rucece in Mbarara and Lake Nakivale in Isingiro representing Southwestern farmlands; Limoto and Gogonyo in Pallisa and Kibuku Districts representing the Kyoga plains agroecological zone (Figure 1). These wetlands offer different benefits to local communities, have different biophysical characteristics, experience varied socioeconomic conditions and are faced with dissimilar management challenges. This study followed three methods for quantifying the monetary values of wetland services and goods, namely, the market price method [4, 9, 10], the productivity method [11, 12], and the contingent valuation method [11, 13, 14]. The market prices method was applied to quantify direct use values, by estimating the price in commercial markets for such wetland resources as papyrus products, pastures, and fish. The respondents made an estimate of the value of nonmarket goods by utilising direct surveys to solicit responses that reflect each individual resource user's valuation of a nonmarket good. The productivity method was used to quantify the use of water. The contingent valuation method was used for nonuse values such as flood attenuation, water recharge and supply, and habitat and breeding.

2.2. Data Collection and Computation of Wetland Values. Consultative meetings were held with environment and wetlands managers of the selected wetland areas to seek their opinions on the most important wetland resources to the communities, challenges, and opportunities for their sound management. Following discussion and advice from wetland managers, important wetland resources for valuation were selected based on (i) whether a resource met the basic needs of the communities from the study area; (ii) number of users harvesting the resource; (iii) whether the resource represented a range of uses to the different users; and (iv) the likelihood of obtaining sufficient quality data on the resource to enable computation of economic values. The other factor considered was whether harvesting, sale or use of the selected resource were particularly important or widespread or where it generated significant local benefits. A summary of the wetland resources selected for valuation is presented in Table 1.

Opportunistic sampling was made for respondents in areas where different wetland resources were harvested, processed, or marketed. Data were collected through interviewing at least 10 respondents for different resources on the value they attached to the wetland goods and services using structured questionnaire interviews. Focus group discussions were also conducted with different wetland resources users to generate information on the uses and associated costs of the resource under valuation. We also reviewed information from district inventory reports on the crop and animal production, population, prices of the associated goods, and wetland area coverage. Value transfers from previous studies [4, 10, 11, 13] were used to compute the values of wetland goods and services.

Data on economic value of wetlands for crop farming in 2012 were collected by estimating the total farming area in each of the agroecological zone and the area of wetlands under crop production from the district inventory reports. We also collected data on the yields and the number of harvests per year for the key crops grown in the wetlands. The key crops considered were maize in the southwestern farmlands agroecological zone, vegetables in the Lake Victoria crescent, and rice in the Kyoga plains agroecological zone.

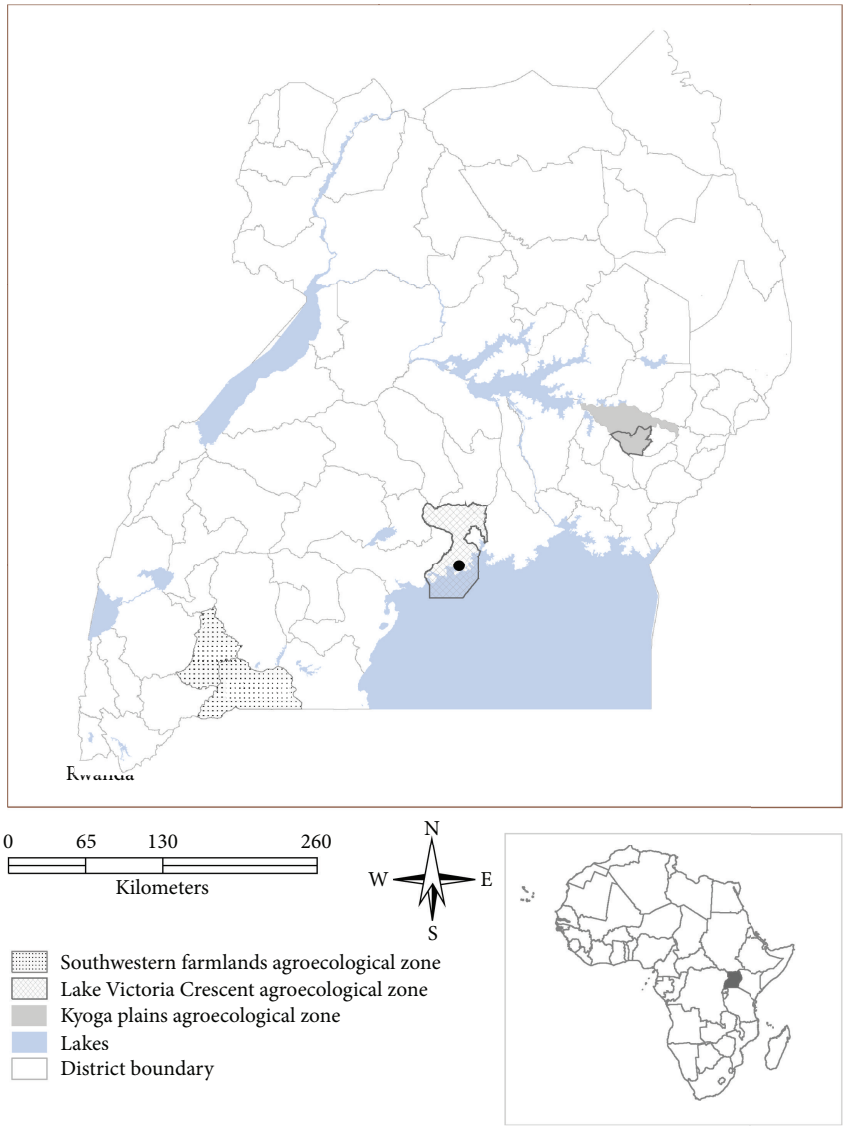

Figure 1: Map of Uganda showing the study sites.

To estimate the value of wetlands for fish breeding, data on the total spawning area were collected from the wetland coverage in the three agroecological zones following spatial data from the National Wetlands Information System. The data were used to derive the estimated value of wetlands for fish breeding per hectare per year.

We also collected data on the percentage of the total number of livestock depending on wetlands and the average daily pasture consumption per animal to estimate the economic value of wetlands to food security through livestock production. Data on the total livestock numbers in the three agroecological zones were obtained from the Ministry of Agriculture, Animal Industry and Fisheries (MAAIF) reports. We only took a conservative estimate of the value of wetland pastures only for cattle, leaving out other types of livestock such as pigs, goats, and sheep. The cost of pastures was inferred from the imputed value of the cost of alternative leafy feeds that would be bought, if wetland pastures were not available, estimated at a cost of US$ 0.2 per animal per day following Karanja et al. [4]. Data were also collected on the number of cattle directly using water from wetlands and the daily consumption of water, which was imputed at 40 litres per animal per day. The value added through milk production was derived from data collected on the number of milked cows and the annual milk production in relation to the price of milk in the three agroecological zones.

The value of grass mulch was estimated using data on the total land area under banana production. Only two districts of Isingiro and Mbarara in the southwestern farmlands agroecological zone were considered in estimation of the economic value of grass mulch in bananas because this is only where the resource was used. Reports from the focus group discussions indicated that in the two districts, nearly every household was engaged in banana growing. It was estimated that on average each family in the two districts owned at least one hectare of bananas. This was supplemented

Table 1: Wetland resources considered for economic valuation in Uganda.

Resources contribution	Southwestern farmlands	Lake Victoria crescent	Kyoga plains	Specific sites
Availability				
Fish	√	√	√	Nakivale, Mabamba L. Nakuwa, and Limoto
Paddy rice			√	Limoto, Gogonyo
Vegetables	√	√		Rucece, Nangabo
Yams/Taro		√		Rucece, Mende, and Limoto
Maize	√	√	√	Rucece, Nakivale, Mende, and Gogonyo
Sugar cane	√	√		Rucece, Mende, and Limoto
Livestock grazing	√		√	Nakivale, Gogonyo
Livestock watering	√		√	Nakivale, Gogonyo
Hunting (bush meat)		√	√	Mabamba, Gogonyo/Limoto
Grass for mulching	√	√		Rucece, Nangabo
Wild fruits and vegetables		√	√	Mende, Mabamba, and Gogonyo
Accessibility				
Papyrus		√	√	Mende, Limoto
Crafts		√	√	Mabamba, Gogonyo
Sand	√	√	√	Rucece, Mabamba
Clay		√	√	Rucece, Mende, and Limoto
Grass for thatching	√		√	Nakivale, Gogonyo, and Limoto
Tourism	√	√	√	Mabamba, L. Mburo
Services/functions				
Breeding ground for fish	√	√	√	Nakivale, Mabamba, Limoto, and Gogonyo
Industrial/urban water supply	√		√	Nakivale, Rucece
Flood control	√	√	√	Rucece, Nakivale, Mende, Mabamba, Limoto, and Gogonyo
Weather modification	√	√	√	Rucece, Nakivale, Mende, Mabamba, Limoto, and Gogonyo
Carbon sequestration	√	√	√	Rucece, Nakivale, Mende, Mabamba, Limoto, and Gogonyo
Domestic water supply	√	√	√	Rucece, Nakivale, Mende, Mabamba, Limoto, and Gogonyo
Transport	√	√	√	Nakivale, Mabamba, L. Nakuwa, and Gogonyo

by data on the total acreage of bananas that were mulched with wetland grass, estimated at 50% of banana plantations, and the number of bundles of mulch applied in the banana plantation per hectare per year.

The economic value of papyrus to food security through food accessibility was computed from data collected on the returns to papyrus resource users by either selling raw papyrus materials or after value addition through mat making. Estimates of the total wetland area in the district under papyrus and the productivity per hectare in head loads were made to generate the total productivity per annum. For the craft products from papyrus, data were collected on price of the raw material and of the products, labour costs for harvesting the product, equipment, additives, storage, licenses, transport, hired, and personal time to derive the net returns.

The economic value of wetlands for fresh water storage and supply was estimated by use of data collected on the number of household dependents on wetlands for water supply and annual water use for all the households. This was extrapolated to the midyear total human population projections of 2012. About 80% of the populations depend on wetlands for domestic water supply as used by Karanja et al. [4] and WMD et al. [5]. The contribution of wetland through nonuse values such as flood attenuation, water recharge and supply, and habitat provision was determined by use of value transfers following Karanja et al. [4] and Turpie [10].

Data on the management costs for conserving wetlands to reflect willingness to conserve (WTC), as reflected in

the economic costs for wetland management and conservation, was generated from the existing costs incurred by districts in the wetland management sector. These included costs for staff salaries and allowances, equipment, and their maintenance and monitoring compliance to wetland conservation. Data on estimated income and other benefits foregone from land use, as well as investment and development opportunities precluded or diminished, to maintain wetlands were used to compute opportunity costs. National data from the Ministry of Water and Environment were used to compute the management and operation costs of conservation. In the three agroecological zones, wetland management and conservation were supported by remittances from the central government, Ministry of Water and Environment, and the locally generated revenue from the respective districts.

3. Results

3.1. The Economic Value of Wetlands through Fish Breeding/Spawning and Availability. In terms of spawning habitats for fish, wetlands in Uganda contributed an estimated gross value of US$ 1,091,444 per year (Table 2). Wetlands do not only serve as breeding grounds for fish whose habitat is shallow waters, but were also mentioned as important spawning areas for fish that live in deep open water. On average, fish available for consumption from wetlands in the three agroecological zones of Uganda were equivalent to US$ 0.49 per person (Table 3). During the focus group discussions, fish was reported as a key source of less expensive animal protein, compared to chicken, beef, and goat meat.

3.2. The Economic Value of Wetlands through Crop Farming. The economic value of wetlands to crop farming was estimated to be in the range of US$ 417,536 to 25.09 million (Table 4). In all the three agroecological zones, wetland adjacent communities noted that yields from wetland crop farming were higher, owing to the moisture guaranteed even during the drought periods. This was in addition to fertility replenishment of the wetland ecosystem from sediment trapping and gradual settling of silt particles and rotting of organic matter from wetland vegetation. This is an indicator that guided use of wetland edges for crop farming largely contributes to livelihoods of surrounding communities and can provide incentives for their involvement in wetlands conservation.

3.3. The Economic Value of Wetlands from Grass Mulch. The gross annual contribution of the wetlands to food security, through provision of grass for mulching, was estimated at US$ 8.65 million per annum (Table 5). Wetlands provided grass mulch that enhanced crop productivity, particularly for banana production in the southwestern farmlands agroecological zone.

3.4. The Economic Value of Wetlands from Pastures and Water for Livestock. Wetlands provided livestock pastures worth US$ 4.24 million (Table 6). Focus group discussions revealed that wetlands were vital grazing areas during the drought periods, when alternative pastures were not readily available. The importance of wetlands was also more significant due to the fact that alternative livestock feeds were expensive and were not easily affordable by most farmers, as reported in the focus group discussions of this study.

The total economic value of water from wetland areas for livestock consumption was estimated to be worth US$ 34 million per year (Table 7). During focus group discussions, the wetlands were reported to serve as watering points not only for the wetland adjacent communities but also to distant livestock farmers. For most free range livestock grazing, the most common source of water for livestock in the study areas was wetlands.

The gross annual value of wetlands to milk production was estimated at US$ 1.22 million (Table 8). About 10% of the total production of milk in the study areas was attributed to grazing livestock within wetlands. During the focus group discussions, the respondents reported that wetlands are more vital during the dry periods, when alternative pastures are not readily available in the catchment areas.

3.5. The Total Economic Value Wetlands for Domestic Water Supply and Papyrus. The gross annual value of domestic water supply was estimated to be worth US$ 13.9 million (Table 9). Wetlands were the only source of water for domestic use at both household and community levels in all the study agroecological zones. The total economic value of wetlands for papyrus was assessed by estimating the value of papyrus raw materials or products that were sold for cash such as crafts and mats. The economic value of papyrus raw materials was valued with two options of either selling raw papyrus materials before processing or after value addition through mat making, which was common in all the three agroecological zones. The annual value of papyrus raw materials was estimated to be US$ 4.63 million (Table 10). Papyrus was used for wall construction, thatching houses, and making a number of craft items such as mats and chairs. Papyrus products were also sold to generate income for acquiring different household foodstuffs. The value addition to papyrus into mats was estimated to annually contribute up to US$ 11.5 million (Table 11). Responses during focus group discussions indicated that making and selling of papyrus crafts provide employment to both men and women.

3.6. Contribution of Wetland Nonuse Values. The estimated economic values of wetland nonuse values are presented in Table 12. The annual contribution ranged from US$ 7.06 million for water recharge and regulation to US$ 1.70 billion for flood control. The nonuse values of wetlands considered in this study were micro-climatic regulation, flood control, water regulation/discharge, habitat/refugia, and recreation. The monetary value of these services was more pronounced in the Lake Victoria crescent agroecological zone.

3.7. Economic Costs of Wetland Management. The wetland management costs for the financial year 2011/2012 totaled to US$ 48,668 per year (Table 13). Management costs were computed based on resources from central government

Table 2: Monetary value of fish spawning grounds in the wetlands of Uganda.

Item	Southwestern farmlands	Lake Victoria crescent	Kyoga plains	Overall
Total spawning area (ha)*	21,459	107,833	45,339	174,631
Estimated value (US$ ha^{-1} yr^{-1})**	6.3	6.3	6.3	6.3
Total gross value per year (US$)	**134,119**	**673,956**	**283,369**	**1,091,444**

*Derived from the Uganda National Wetlands Information Systems.
**From Turpie (2000) [10] at U$ US$ 6.25 ha^{-1} yr^{-1}.

Table 3: Per capita fish availability among the local communities in the wetlands areas of Uganda.

Item	Southwestern farmlands	Lake Victoria crescent	Kyoga plains
Number of resource users	21	17	37
Total Revenue (US$)	464,295	372,300	365,000
Human population in 2012	865,800	1,371,600	544,300
Per capita fish revenue (US$)	**0.54**	**0.27**	**0.67**

Table 4: Monetary value of wetlands in terms of crop farming in three agro-ecological zones of Uganda.

Variable	Southwestern farmlands	Lake Victoria crescent	Kyoga plains
Major crop grown in wetlands	Maize	Vegetables (Nakatti)	Rice
Total farming area in wetlands (ha)*	932	3,065	16,335
Area of Wetland under crops (ha)	746	2,452	13,068
Yield per hectare (per season) tonnes	4	8	2
Number of harvests per year	2	3	2
Total Harvest per year (tonnes)	5,219	55,170	52,272
Price per tonne (US$)	80	60	480
Gross annual value of harvest at farm gate prices (US$ per year)	**417,536**	**3,310,200**	**25,090,560**

*Derived from the Uganda National Wetlands Information Systems.

Table 5: Monetary contribution of wetland grass to food security through mulching bananas in the South western Farmlands agro-ecological zone.

Variable	Isingiro district	Mbarara district	Values for southwestern farmlands
Midyear human population projections (2012)	420,200	445,600	865,800
Number of households	60,029	63,657	123,686
Total hectares of bananas (ha)	60,029	63,657	123,686
Total hectares of bananas that are mulched with wetland grass (ha)	30,014	31,829	61,843
Number of bundles of mulch applied per hectare	700	700	700
Number of times mulch is applied per year	2	2	2
Total number of bundles of mulch applied per year	42,020,000	44,560,000	86,580,000
Cost per bundle (US$)	0.10	0.10	0.10
Gross value of mulch applied (US$)	**4,202,000**	**4,456,000**	**8,658,000**

All values reflect estimates of the entire wetlands in Uganda.

funds, locally generated revenues, and salaries and allowances of the wetland staff in each agroecological zone.

3.8. Opportunity Costs for Limiting Access to Wetlands. The opportunity cost was estimated in the range of US$ 1.40 to 6.61 million (Table 14). The value used to estimate the foregone benefits was derived from an estimate by Karanja et al. [4], which indicated that the average benefit for maintaining biodiversity in Uganda was US$ 48.24 per hectare per year. The study considered the opportunity cost, if the current use

Table 6: Monetary value of wetland pastures in three agro-ecological zones in Uganda.

Variables	Southwestern farmlands	Lake Victoria crescent	Kyoga plains	Overall
Total number cattle	330,337	114,769	136,225	581,331
% of total cattle dependant on wetlands*	10	10	10	10
Number of cattle raised in wetlands	33,034	11,477	13,623	58,133
Average value of pasture consumed per day per animal (US$)	0.20	0.20	0.20	0.20
Imputed value of pasture consumed by all animals per day (US$)	6,607	2,295	2,725	11,627
Total value of pasture consumed per year (US$)	**2,411,460**	**837,814**	**994,443**	**4,243,716**

*Estimate 10% of the cattle to directly use wetlands for grazing.

Table 7: Monetary value of wetlands for livestock watering in three agro-ecological zones of Uganda.

Variables	Southwestern farmlands	Lake Victoria crescent	Kyoga plains	Overall
Total number of cattle	330,337	114,769	136,225	581,331
Number of cattle obtaining water from wetlands*	33,034	11,477	13,623	58,133
Amount of water consumed per day per head of cattle (20 litre jerry cans)	2	2	2	2
Total amount of water consumed per year (20 litre jerry cans)	24,114,601	8,378,137	9,944,425	42,437,163
Cost of water per 20 litres (US$)	0.04	0.04	0.04	0.04
Gross annual value of water for livestock production (US$)	**964,584**	**335,125**	**397,777**	**1,697,487**

*Estimate 10% of the cattle to directly use wetlands for watering.

Table 8: Gross monetary value addition from wetlands through milk production in Uganda.

Variables	Southwestern farmlands	Lake Victoria crescent	Kyoga plains	Overall
Total number of cattle	330,337	114,769	136,225	581,331
Number of milked cows*	56,100	22,290	12,600	90,990
Average milk production per cow per week (litres)*	7.1	25.6	5.3	19.0
Total annual milk production (litres)	20,566,260	29,672,448	3,472,560	89,779,833
Percentage attributed to wetlands	10	10	10	10
Milk production assessed to wetlands (litres)	2,056,626	2,967,245	347,256	8,977,983
Price of milk (US$)	0.13	0.13	0.15	0.14
Gross value of milk production per annum (US$)	**257,078**	**398,798**	**51,394**	**1,219,210**

*Derived from MAAIF and UBOS, 2009 [15].

Table 9: The gross annual monetary value of wetlands for domestic water supply in three agro-ecological zones in Uganda.

Variable	Southwestern farmlands	Lake Victoria crescent	Kyoga plains	Overall
Midyear human population projections (2012)*	865,800	1,371,600	544,300	2,781,700
Number of households	123,686	195,943	77,757	397,386
Households dependant on wetlands for water supply**	98,949	156,754	62,206	317,909
Average use of water (20 litre jerrycans)	3	3	3	3
Water use for all the households per year (m^3)	2,166,974	3,432,919	1,362,305	6,962,198
Market price per m^3 (US$)***	2	2	2	2
Gross annual value of water for domestic consumption (US$)	**4,333,947**	**6,865,838**	**2,724,610**	**13,924,395**

*Human population projections were based on midyear values of 2012.
**Estimated at 80% from WMD et al. (2009) [5].
***Computed based on the price of a 20 litre jerrycan at US$ 0.04.

Table 10: Monetary value of papyrus raw materials without value addition in three wetland agro-ecological zones in Uganda.

Variables	Southwestern farmlands	Lake Victoria crescent	Kyoga plains	Overall
Total area under papyrus (ha)*	12,713	20,751	32,095	65,559
Productivity per hectare (head loads)	400	400	400	400
Total productivity per annum (head loads)	5,085,200	8,300,400	12,838,000	26,223,600
Price per head load (US$)	0.17	0.20	0.16	0.18
Total gross value of papyrus production (US$)	**864,484**	**1,660,080**	**2,054,080**	**4,632,836**

*Derived from Uganda National Wetlands Information Systems.

Table 11: The economic value of papyrus after value addition through mat making.

Variables	Southwestern farmlands	Lake Victoria crescent	Kyoga plains	Overall
Total productivity (head loads)	5,085,200	8,300,400	12,838,000	26,223,600
Number of head loads converted into 2.5 m × 3.5 m mats (SW Farmlands: 75%, LVic. Crescent: 80%, Kyoga Plains: 65%)	3,813,900	6,640,320	8,344,700	19,230,640
Number of mats produced (1 : 2 conversion ratio)	7,627,800	13,280,640	16,689,400	38,461,280
Price per mat (US$)	0.40	0.40	0.40	0.40
Gross value of papyrus mats produced (US$)	3,051,120	5,312,256	6,675,760	15,384,512
Cost of processing inputs (US$)	762,780	1,328,064	1,668,940	3,846,128
Gross value addition (US$)	**2,288,340**	**3,984,192**	**5,006,820**	**11,538,384**

Table 12: Monetary contribution of wetland non-use values in three agro-ecological zones of Uganda.

Variable	Monetary value US$ ha^{-1} yr^{-1}	Southwestern farmlands	Lake Victoria crescent	Kyoga plains	Overall
Area (ha)		29,155	137,125	68,932	235,212
Microclimatic regulation	265	7,726,075	36,338,125	18,266,980	62,331,180
Flood control	7,240	211,082,200	992,785,000	499,067,680	1,702,934,880
Water regulation/recharge	30	874,650	4,113,750	2,067,960	7,056,360
Habitat/refugia	439	12,799,045	60,197,875	30,261,148	103,258,068
Recreation/aesthetic	491	14,315,105	67,328,375	33,845,612	115,489,092
Cultural	1,761	51,341,955	241,477,125	121,389,252	414,208,332

*Values derived from Karanja et al. (2001) [4].

Table 13: Costs for wetland management and conservation in three agro-ecological zones of Uganda (Data for 2011/2012).

Item	Southwestern farmlands	Lake Victoria crescent	Kyoga plains	Overall
Central government funding (US$)	8,040	4,840	5,988	18,868
Local revenue (US$)	1,400	3,600	1,400	6,400
Salary and allowances (US$)	9,600	5,760	8,040	23,400
Total management costs (US$)	**19,040**	**14,200**	**15,428**	**48,668**

of the wetlands was to be stopped before any modification or conversion. This would lead to foregoing foodstuffs, income, and other economic opportunities.

3.9. Net Economic Contribution of Wetlands to Food Security. Wetlands in the three agroecological zones provided an average net contribution of about US$ 10,491 per hectare per year (Table 15). This took into consideration the economic value derived from the various uses of the wetlands, the costs involved in the management, and the fact that the benefits of wetlands can be sustained by good management interventions. This is in line with the study by Karanja et al. [4], which estimated the net total economic value of wetlands in Pallisa District at US$ 10,861 per hectare per year, and

Table 14: Opportunity costs of limiting community access to wetlands in three agro-ecological zones in Uganda.

Variable	Southwestern farmlands	Lake Victoria crescent	Kyoga plains
Area (ha)	29,110	137,125	68,932
Opportunity cost (US$ ha^{-1} yr^{-1})	48.24	48.24	48.24
Total opportunity cost (US$)*	**1,404,266**	**6,614,910**	**3,325,280**

*Based on national average of US$ 48.24 ha^{-1} derived from Karanja et al. 2001 [4].

Maclean et al. [16], which estimated benefits from Lake Bunyonyi in the range of US$ 11,200 to US$ 24,000 per hectare per year.

4. Discussion

4.1. The Economic Values of Wetlands

4.1.1. Fish Availability and Breeding/Spawning. Results show that wetlands were valued as major breeding grounds for fish. Wetlands are important for the reproduction of certain fish species like *Protopterus*, *Clarias*, *Schilbe*, *Labeo*, *Alestes* spp., and *Oreochromis niloticus* [17]. They are also important habitats for a number of fish species including *Clarias* spp., mudfish, *Protopterus* spp., and various *Haplochromis* spp. In addition to serving as breeding grounds, the contribution of wetlands through provision of fish is most significant for species that have respiratory systems that are adapted to seasonal flooding and can withstand reduced water levels in wetland areas such as *Clarias* spp. [18]. However, in some of the pilot areas, the spawning grounds for fish species that reproduce in wetlands are under threat as result of the ever-progressing encroachment. This justifies the need to protect wetlands for increased fisheries resources, given the fact that most fishes breed in shallow waters along wetland areas, as noted by WMD et al. [5].

Results from this study further indicate the value of wetlands through fish catch as food and source of proteins and employment to the fishing communities. During the focus group discussions, it was noted that *Clarias* sp. (catfish) are commonly harvested from wetland areas in the Kyoga plains and Southwestern farmlands agroecological zone, and provide a cheap source of animal protein and are one of the main commercial activities during the dry season when water levels of wetlands reduce, providing easy harvesting. The findings from this study indicate that the contribution of wetlands to food security through provision of fish is significant, and this is supported by other findings that in Uganda fish provides up to 50% of all animal protein [19]. The results are supported by other studies such as Akwetaireho [20], in which wetlands support livelihoods of people engaged in fishing such as fishers, boat owners, crew, and employees in fish processing factories. Wetlands are thus of importance to socioeconomic development from the fisheries sector, whose contribution in 2009 was estimated at about 2.8% of Uganda's national GDP [21]. Loss of wetlands will therefore have a significant impact on the livelihoods of local communities and will have a negative impact on the availability of fish. Benefits from fish harvest to local communities can serve as incentives for involvement in the conservation of wetlands in different areas and should therefore be enhanced.

4.1.2. Crop and Livestock Farming. The economic value of wetlands through crop production was enormous. The economic contribution of wetlands through crop farming is locally and globally recognised as indicated by one of the crops valued during this study (paddy rice), regarded as a staple diet of more than half the world's population, Uganda inclusive [22]. Successes in socioeconomic development to local communities from use of wetlands for crop farming have also been reported in Ethiopia [23]. Given the current impacts of climate change on unpredictable rainfall [24], use of wetlands for crop farming will keep increasing, considering the fact that wetlands have all year round reliable moisture for crop growth. The availability of moisture and nutrients provides an opportunity for use of wetlands edges for production of different crops throughout the year, if clear guidelines for different practices are provided to minimise increased drainage of wetlands for crop farming.

During focus group discussions, the farmers noted that over time the fertility their soils has declined, which has necessitated the use of inorganic fertilizers and pesticides, which represent a potential threat to the wetland ecosystems. These agrochemicals alter the ecological balance of wetlands and can indirectly eliminate important faunas that play a role in wetland functions and services. As noted by Dixon and Wood [23] and FAO [25], the wise use of wetland for crop farming should therefore be guided by well-defined policies and legislation to limit the amount of areas to be drained, quantities of agrochemicals to be used, in addition to use of appropriate agronomic practices. Lessons on the wise use of wetlands for crop farming are available from FAO [25] and Heimlich et al. [26]. Successes in socioeconomic development to local communities from use of wetlands for crop farming have also been reported in Ethiopia [23] and are expected to be possible in Uganda [5].

Results further show that wetlands contribute significantly to crop farming through grass mulch. Mulching helps retain the moisture, controls soil erosion, and acts as a source of organic manure in the banana plantations. Wetlands are the major remaining sources of mulch, comprising mainly of sedges including *Typha* spp. and *Cyperus* spp. [5]. During focus group discussions, it was reported that wetland grass mulch adds value to banana productivity through moisture retention and erosion control and also acts as a source of organic manure in the banana plantations. The farmers indicated that without mulch, banana yields can even reduce by 50%.

Table 15: Summary of total economic contribution of wetlands in three agro-ecological zones of Uganda.

Resource contribution (US$)	Southwestern farmlands	Lake Victoria crescent	Kyoga plains
Availability			
Fish breeding/spawning	134,119	673,956	283,369
Fish production	464,295	372,300	365,000
Crop farming	417,536	3,310,200	25,090,560
Livestock grazing/pastures	2,411,460	837,814	994,443
Livestock watering	19,291,681	6,702,510	7,955,540
Value added through milk production	7,717	2,681	3,182
Wetland grass for mulching	4,202,000	4,456,000	8,658,000
Accessibility			
Papyrus	864,484	1,660,080	2,054,080
Papyrus crafts	2,288,340	3,984,192	5,006,820
Services/functions			
Domestic water supply	4,333,947	6,865,838	2,724,610
Nonuse values	298,139,030	1,402,240,250	704,898,632
Total economic value to food availability	**26,928,808**	**16,355,461**	**43,350,094**
Total economic value to food accessibility	**3,152,824**	**5,644,272**	**7,060,900**
Total economic value through services and functions	**302,472,977**	**1,409,106,088**	**707,623,242**
Total economic value of wetlands to food security	**332,554,609**	**1,431,105,821**	**758,034,236**
Costs of management and maintenance of wetlands			
Management costs	19,040	14,200	15,428
Opportunity costs	1,404,266	6,614,910	3,325,280
Total economic cost to maintain the wetlands	**1,423,306**	**6,629,110**	**3,340,708**
Net economic value of wetlands for food security	**331,131,303**	**1,424,476,711**	**754,693,528**
Net benefits per hectare per year (US$)	**11,358**	**10,388**	**10,948**

Wetlands were also valued for provision of fodder, especially during the drought periods, when alternative pastures were not readily available. Pastures from wetlands not only provided fodder but also enhanced milk production, thus contributing to food security. The importance of wetlands is also more significant due to the fact that alternative livestock feed is expensive and may not be easily affordable by most farmers in Uganda. This is more significant with the current challenges of climate change and unpredicted weather conditions [27]. However, most wetlands suffer from overgrazing. Overgrazing harm wetlands through soil compaction, removal of vegetation, and river bank or lake shore destabilization [28]. These changes in turn affect wetlands' filtering capacity, flood control capabilities, water recharge, and wildlife habitat. Other studies have identified the direct effects of livestock grazing to include the consumption of plant biomass, trampling of plants, including belowground parts and soil, nutrient inputs and bacterial contamination from dung and urine, and the introduction and dispersal of seeds and other propagules [28, 29]. Similar effects are likely to be experienced in the study wetlands where livestock grazing is the key livelihood activity. However, there is limited information on the effects of livestock grazing on water and soil quality in wetlands in the study area [15]. Studies are ongoing in Uganda that will give evidence-based information for formulation of livestock grazing guidelines.

Wetlands were valued as the most reliable water sources for livestock grazers. The importance of wetlands for livestock watering is more pronounced during dry seasons when most water sources dry up and large herds of cattle concentrate in few wetlands. However, information from focus group discussions indicated that watering livestock usually leads to grazing the livestock nearby, and when kept near streams and wetlands, they trampled river banks and lake shores, damaging vegetation resulting in increased erosion and sedimentation. This in most cases leads to soil compaction, removal of vegetation, and river bank or lake shore destabilization. It also directly adds animal waste, which most often leads to pollution. Similar impacts of livestock watering on wetlands have been noted by Belsky et al. [30], Robertson and Rowling [31], and Staton and O'Sullivan [32]. Uncontrolled grazing and watering of livestock in wetland areas also often results in increased stream turbidity, as well as increased input of nutrients and bacteria into the stream, which affects the quality of water available to downstream users. Impacts of livestock wastes contaminating streams with faecal organisms contained in the wastes, which lead to health problems for humans, have been noted by Miner et al. [33]. Such effects are very significant in Uganda, where more than 80% of the population directly use water from wetlands [5]. The impacts of livestock watering can be minimised by providing alternative livestock watering facilities as proposed by Jansen and Robertson [28], Staton and O'Sullivan [32], and Miner et al. [33].

4.1.3. Papyrus. Harvesting of papyrus is one of the sustainable wetland uses of wetlands that would provide multiplier

effects, in addition to direct income to papyrus harvesters and processors. This has been confirmed by studies elsewhere, such as Karanja et al. [4], Emerton et al. [11], Maclean et al. [16], and Muthuri et al. [34]. Moreover, the benefits from papyrus can motivate the users not to clear the papyrus wetlands, which would provide a relatively less degraded wetland that can provide other ecological services such as climate modulation and water purification and filtration. For example, it is known that papyrus swamps are significant sinks of carbon as they have a high net primary productivity and large amounts of detritus that can accumulate below the living mat of rhizomes and roots [16, 35–37].

4.1.4. Domestic Water Supply. Another important resource provided by the wetlands in all the agroecological zones was water for domestic use. Wetlands are the main sources for the spring wells, boreholes, shallow wells, valley dams, and natural wells where local communities draw water for domestic use. As noted by Akwetaireho [20], wetland meets the daily water requirements of around 18, 885 people living close to wetlands and that about 119,249.333 litres cubic meters of water per year are collected from the watering sources scattered around the wetlands. Availability of water from wetlands enables the disadvantaged groups particularly women and children to easily access water rather than walking long distances which is an additional burden to their domestic cores.

4.1.5. Nonuse Values. Wetlands were valued for the nonuse values such as micro-climatic regulation, flood control, water regulation/discharge, habitat/refugia, and recreation/tourism. Though rarely appreciated, the nonuse values contribute to the benefits that directly or indirectly play a role in food security. Thus, providing monetary figures for wetland nonuse values gives a basis for planning and decision making on the importance of leaving some wetlands intact. This is critical because the loss of most nonuse values is not easily recognised, compared to the direct resources, whose loss can be realized by lost incomes.

It is worth to note that wetland resource utilisation activities are carried out almost exclusively by the people who live in settlements which directly border relevant wetlands. However, the benefits associated with nonuse values accrue over a much larger area, to rural and urban residents, and most of them are of public goods nature and deserve special consideration. This is recognised by different studies as one of the strong justification of leaving some wetland areas intact, with minimal disturbance as justified by Emerton et al. [11], Balmford et al. [38], Bullock and Acreman [39], and Korsgaard and Schou [40].

4.2. Net Economic Contribution of Wetlands to Food Security. The findings from this study indicates that if wetland resources were used unsustainably, or in a manner which reduces societal net benefits, local people's income would decline. This is likely to affect their perceived value of wetlands and would further encourage even more unsustainable levels of resource use, ultimately leading to the destruction of wetland ecosystems as observed by Korsgaard and Schou [40] and Bai et al. [41, 42]. The estimates of wetland benefits as for this study illustrate the magnitude of the economic value of wetlands in addition to their biodiversity, scientific value, climate regulation, potential tourism, social, cultural and other important wetland values. They further represent one more tool to raise awareness with decision makers about the economic year.

5. Conclusions and Recommendations

Results from this study provide evidence of the economic benefits derived from wetland goods and services. The study points out that many rural people's livelihoods depend directly on wetlands in addition to wetlands provision of ecosystem services. Often, these people are resource poor and they have few alternatives once the ecosystems deteriorate. It is also appreciated that the increasing human and animal populations and uncertain climatic conditions are exerting immense pressure on the different wetland resources, leading to varying levels of wetland degradation, which may lead to loss of the benefits.

One of the causes of wetland degradation is information failure, which most often is caused by lack of understanding of the values of wetlands, including the economic values. For such reasons, the protection of wetlands does not appear to be a serious alternative for resource users and cannot be advocated for by planners and policy makers.

Findings from the study therefore hold great potential for raising awareness about the roles and economic values of wetland benefits and ecosystem services. There is need to disseminate results from this study to resource users, policy makers and implementers, and to make them recognize the economic value of wetlands and put their efforts in sustainable management of the important resources by drawing strategies to sustain the wetland benefits to society.

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Research Article

The Ecological Response of *Carex lasiocarpa* Community in the Riparian Wetlands to the Environmental Gradient of Water Depth in Sanjiang Plain, Northeast China

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The response of *Carex lasiocarpa* in riparian wetlands in Sanjiang Plain to the environmental gradient of water depth was analyzed by using the Gaussian Model based on the biomass and average height data, and the ecological water-depth amplitude of *Carex lasiocarpa* was derived. The results indicated that the optimum ecological water-depth amplitude of *Carex lasiocarpa* based on biomass was [13.45 cm, 29.78 cm], while the optimum ecological water-depth amplitude of *Carex lasiocarpa* based on average height was [2.31 cm, 40.11 cm]. The intersection of the ecological water-depth amplitudes based on biomass and height confirmed that the optimum ecological water-depth amplitude of *Carex lasiocarpa* was [13.45 cm, 29.78 cm] and the optimist growing water-depth of *Carex lasiocarpa* was 21.4 cm. The TWINSPAN, a polythetic and divisive classification tool, was used to classify the wetland ecological series into 6 associations. Result of TWINSPAN matrix classification reflected an obvious environmental gradient in these associations: water-depth gradient. The relation of biodiversity of *Carex lasiocarpa* community and water depth was determined by calculating the diversity index of each association.

1. Introduction

Water regime, as distinct from instantaneously measured water depth, has been implicated in affecting the composition, diversity, and distribution of macrophyte communities [1–6]. While the influence of water level fluctuation on the germination and establishment of wetland seed banks has been well documented, there is few research on the impact of water level on the growth of mature plants [2]. Water regime (depth, duration, and frequency of flooding) is the principal factor determining plant species distribution along the the land-water interface in wetlands [1, 6–17]. *Carex lasiocarpa* wetland is the main wetland type in the mire wetlands in Sanjiang Plain, Northeast China [18]. *Carex lasiocarpa*, a perennial Cyperaceae moss grass, is a clonal perennial which can form nearly monospecific stands on shorelines and lakesides [2, 19–23]. Where water conditions permit, such as in bays protected from waves, the species sometimes forms thick, floating mats. These floating mats often support a rich array of other plant life adapted to wet infertile conditions. Hence, this particular species of *Carex* is important in producing distinctive plant communities along lakes and rivers [2, 19, 20]. In wetlands in Sanjiang Plain it is generally considered to be an indicator species for wetlands [18, 19, 21, 22].

Responses of diversity of assemblages and individual species to water depth are necessary to be considered in the management and restoration efforts of wetland ecosystems; therefore, understanding plant response to hydrologic conditions is important both to the maintenance of native biodiversity and to the design of management strategies appropriate to a specific wetland [8, 24–26]. At present, research on the vegetation in Sanjiang plain wetlands still mostly focuses on the traditional classification and description, while little attention paid on the ecological pattern and the process of quantitative research, especially the quantitative research on

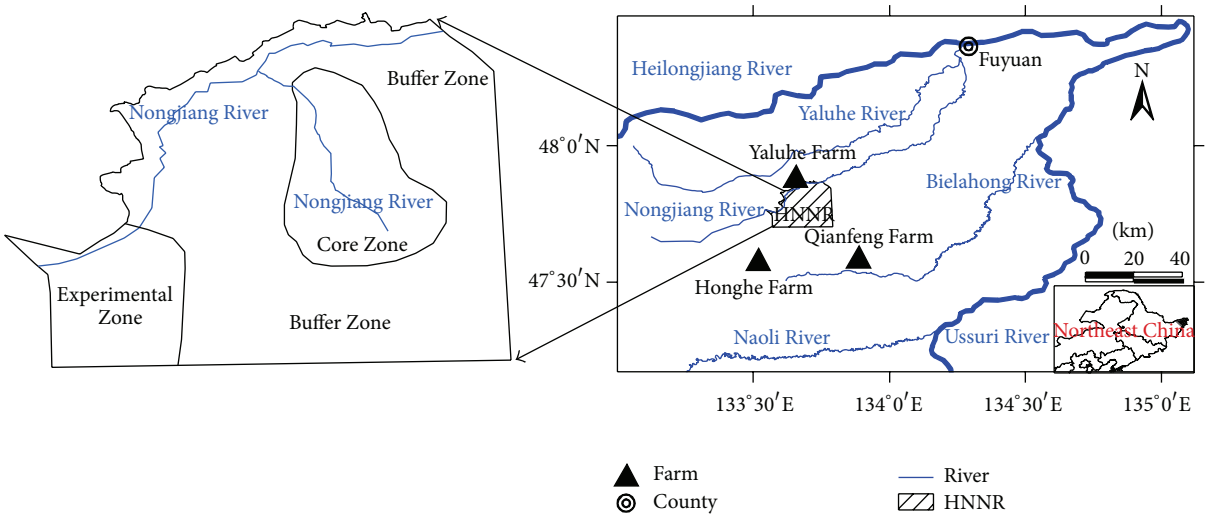

Figure 1: Location of the study area.

the relationship between wetland hydrological and vegetation [2, 20, 21, 27–31]. The present study sought to achieve the following two objectives: (1) analyze the response of *Carex lasiocarpa* populations to the environmental gradient of water depth using the biological characters of biomass and height based on Gaussian Model and figure out the ecological amplitude of reed populations to water depth; (2) explore the relationship between the ecological characteristics of *Carex lasiocarpa* communities and water-depth variables. Based on the relationship between the ecological characteristics of wetland vegetation and hydrological regime, the suggestion of ecohydrological management for wetland ecosystem is proposed.

2. Materials and Methods

2.1. Study Area. The Honghe National Nature Reserve (HNNR) (47°42′18″N–47°52′N and 133°34′38″E–133°46′29″E) is located in the northeast of Sanjiang Plain, Northeast China, with an area of 250.9 ha (Figure 1).

HNNR has been listed as the International Important Wetland (Ramsar wetland) since 2001 for being a typical inland wetland and fresh water ecosystem in the north temperate zone [32]. Presently, HNNR has 16 orders, 43 families, and 174 species of waterfowl, including ten species of nationally rare and endangered waterfowl. In addition, 1012 species of plants are founded in HNNR, including six species of nationally endangered plants. With a very low topographic gradient (average slope grade less than 1 : 10,000), this area is favorable to the formation of wetland ecosystems [33].

2.2. Methods

2.2.1. Sampling Method. Samples were collected in 28 sampling spots (Figure 2) from May to September of 2011 and 47 sampling spots in 2012. Three quadrats ($50 \times 50\,\text{cm}^2$) of plants samples were collected with scissors at each sample point. Plants naturally growing in the quadrats were recorded with their names, abundances, coverage, heights, and aboveground biomass (dry weight). Coordinate of each sampling quadrat was also recorded by using GPS.

2.2.2. Data Analysis. (1) Gaussian Model was adopted to describe the species-environmental relations. The study on the response of reed to water depth based on the Gaussian Model has been achieved good effect [34, 35]. The Gaussian Model was shown as the following equation:

$$y = ce^{[-(1/2)(x-u)^2/t^2]}, \tag{1}$$

where y represents an indicator of biological characteristics of plant species, which can be abundance, coverage, density or biomass, and so on; c is the maximum of y; x is the value of environmental factor, u is the optimum ecological amplitude of species to environmental factor; and t is species tolerance. Generally optimum ecological amplitude of species to environmental factor change within $2t$ range. The analysis was performed by Excel 2003.

(2) Two-Way Indicator Species Analysis (TWINSPAN) was used to classify the plant community in the study area based on the number of species in all quadrats [36]. The TWINSPAN analysis was performed by using winTWINS 2.3 [37].

(3) The ecological characteristics of plant community were reflected with the following indices [38–40].

Species richness: Margalef index (MA) was adopted, which is expressed as

$$\text{MA} = \frac{(S-1)}{\ln N}, \tag{2}$$

where S is the number of species and N is the number of individuals of all species in a community.

Species diversity: Shannon-Weaver index (H) was used, which can be calculated as

$$H = -\Sigma Pi \ln Pi, \tag{3}$$

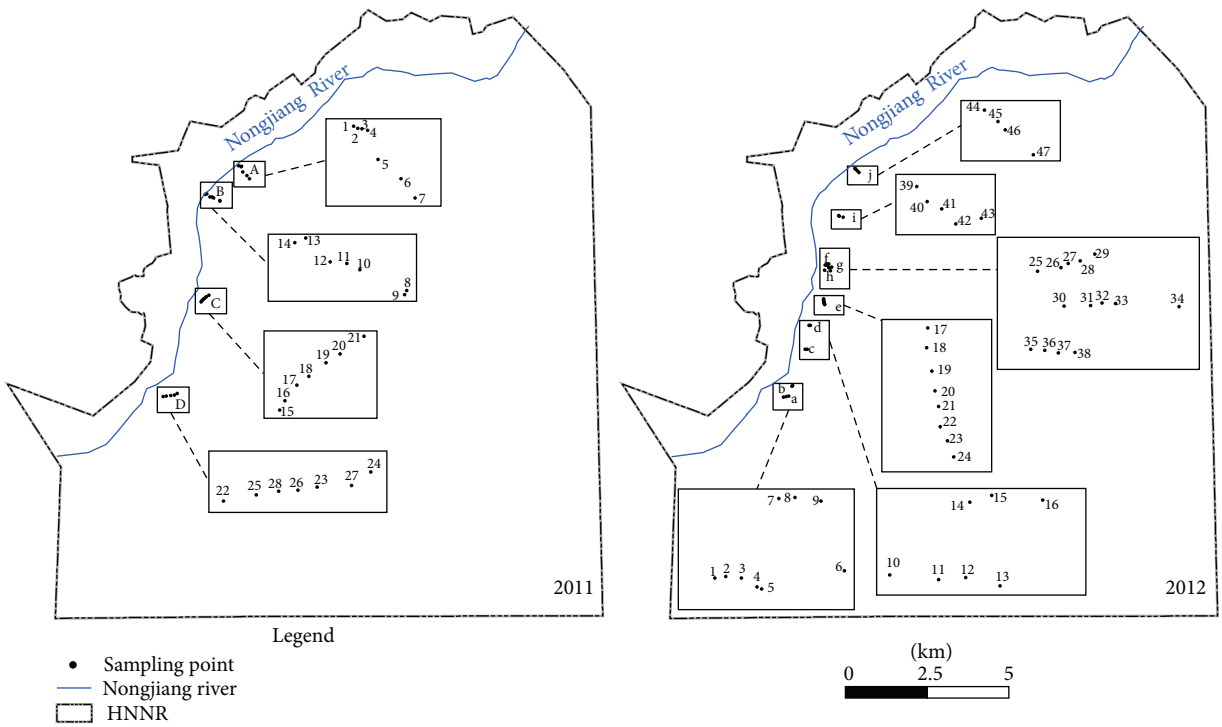

Figure 2: Location of sampling spots.

where $Pi = ni/N$, ni is the importance value of species i, and N is the sum of the importance value of all species in a community.

Species evenness: Pielou evenness index (E) was used, which is expressed as:

$$E = \frac{H}{\ln(S)}, \quad (4)$$

where S is the number of species and H is the Shannon-Weaver index.

(4) Data was analyzed by using SPSS17.0 and Microsoft Excel.

3. Results and Discussion

3.1. Response of Carex lasiocarpa to Water Depth

3.1.1. Result of Statistical Analyses. 17 *Carex lasiocarpa* community sampling data (spots without *Carex lasiocarpa* removed) from May to September in 2011 were statistically analyzed (Table 1). The water depth ranged within 2.5–37.5 cm with a mean value of 17.18 cm. Average population height ranged within 38–73.25 cm, and the average value was 59.64 cm. Range of populations biomass was 1.3–47.68, and the mean value was 25.37. All the sampling data was of normal distribution.

Table 1: The statistical description of *Carex lasiocarpa* community in different sampling points.

Sampling points	Water depth (cm)	Population height (cm)	Population biomass (g/m^2)
2	22.70	53.25	33.54
4	12.80	64.00	24.96
5	8.20	52.50	26.86
6	5.70	38.00	1.30
7	3.10	43.00	2.63
8	26.80	73.25	28.10
9	27.90	69.50	47.68
10	20.10	69.00	36.90
12	18.80	68.75	35.92
13	13.50	60.75	29.94
17	31.50	54.50	10.92
19	37.50	48.75	9.15
21	22.20	66.67	23.58
23	9.60	69.25	28.41
25	15.00	66.00	39.00
26	14.20	71.25	46.05
28	2.50	45.50	6.36
Mean	17.18	59.64	25.37
Max	37.50	73.25	47.68
Min	2.50	38.00	1.30

3.1.2. Response of the Population Biomass of Carex lasiocarpa to Water Depth. Population biomass of *Carex lasiocarpa* was strongly correlated to water depth ($R^2 = 0.7229$, $P < 0.01$). Quadratic curve fitting was used to fit the relationship between population biomass data of *Carex lasiocarpa* (after natural logarithm transformation) and water-depth

data, and the obtained quadratic curve was fit with gaussian regression (Figure 3). With regression analyses, the Gaussian regression equation and regression curve were obtained, which was expressed as the following equation:

$$y = 39.82 \exp\left[-\frac{(1/2)(x-21.61)^2}{8.16^2}\right]. \quad (5)$$

The results indicated that the optimum ecological amplitude of *Carex lasiocarpa* to water depth based on population biomass was [13.45 cm, 29.78 cm] and the optimist growing point is 21.6 cm.

3.1.3. Response of the Population Height of Carex lasiocarpa to Water Depth. Population height of *Carex lasiocarpa* was strongly correlated to water depth ($R^2 = 0.6685$, $P < 0.01$). Quadratic curve fitting was used to fit the relationship between population height data of *Carex lasiocarpa* (after natural logarithm transformation) and water-depth data, and the obtained quadratic curve was fit with Gaussian regression (Figure 4). With regression analysis, the Gaussian regression equation and regression curve were obtained, which was expressed as the following equation:

$$y = 68.43 \exp\left[-\frac{(1/2)(x-21.21)^2}{18.90^2}\right]. \quad (6)$$

The results indicated that the optimum ecological amplitude of *Carex lasiocarpa* to water depth based on population average height was [2.31 cm, 40.11 cm] and the optimist growing point is 21.2 cm.

3.1.4. The Optimum Ecological Amplitude of Carex lasiocarpa to Water Depth. An intersection of the ecological amplitudes based on biomass ([13.45 cm, 29.78 cm]) and height ([2.31 cm, 40.11 cm]) was carried out to figure out the ecological amplitude of *Carex lasiocarpa* to water depth in general. The final result confirmed that the optimum ecological amplitude of *Carex lasiocarpa* to water depth was [13.45 cm, 29.78 cm] and the optimist growing point of *Carex lasiocarpa* to water depth was 21.4 cm.

3.2. Response of Community Diversity of Carex lasiocarpa to Water Depth. By using TWINSPAN, the 47 sampling spots in 2012 were classified into 6 groups at the end of division (Figure 5).

Association Group I: Association *Carex pseudocuraica + Carex lasiocarpa*. This was also a hygrophyte association group including 3 sampling spots (S19, S20, S24). *Carex pseudo-curaica* and *Carex lasiocarpa* were dominant species, while the others were companion species. *Carex pseudo-curaica* occurred in relatively deep water conditions.

Association Group II: Assoc. *Carex pseudo-curaica + Carex lasiocarpa + Glyceria spiculosa*. This was also a hygrophyte association group including 19 sampling spots (S2, S3, S4, S5, S7, S8, S9, S10, S12, S13, S14, S15, S16, S18, S21, S23, S25, S42, S43). The group occured in relatively moderate water-depth conditions.

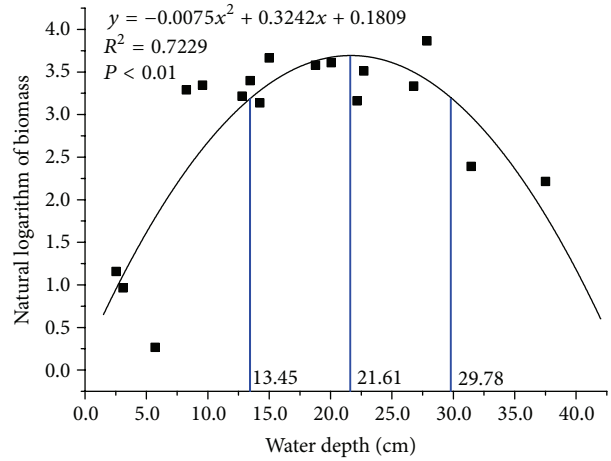

Figure 3: The secondary nonlinear regression based on Gaussian Model of *Carex lasiocarpa* population biomass and water depth.

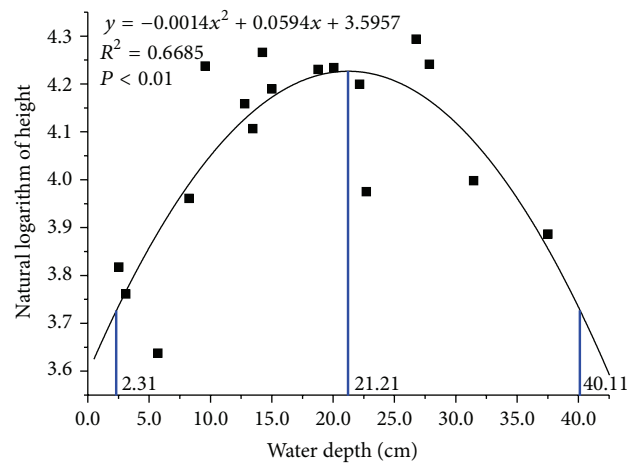

Figure 4: The secondary nonlinear regression based on Gaussian Mode of *Carex lasiocarpa* populations height and water depth.

Association Group III: Assoc. *Carex lasiocarpa + Carex pseudo-curaica + Glyceria spiculosa + Carex dispalata*. This was also a hygrophyte association group including 4 sampling spots (S31, S32, S33, S44). The group occured relatively in moderate water depth conditions.

Association Group IV: Assoc. *Glyceria spiculosa + Carex lasiocarpa + Carex pseudo-curaica + Calamagrostis angustifolia*. This was also a mesophyte association group including 12 sampling spots (S1, S11, S22, S26, S27, S34, S35, S36, S39, S40, S45, S47).

Association Group V: Assoc. *Carex lasiocarpa + Calamagrostis angustifolia + Carex pseudo-curaica*. This was also a mesophyte association group including 7 sampling spots (S6, S17, S30, S37, S38, S41, S46). *Carex lasiocarpa*, *Calamagrostis angustifolia* and *Carexpseudo-curaica* were dominant species, and the others were companion species.

Association Group VI: Assoc. *Calamagrostis angustifolia + Carex lasiocarpa*. This was also a mesophyte association

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Figure 5: TWINSPAN analyses. Note: *1-Carex lasiocarpa, 2-Glyceria spiculosa, 3-Carex pseudo-curaica, 4-Calamagrostis angustifolia, 5-Galium manshuricum Kitag., 6-Galium dahuricum Turcz, 7-Comarum palustre L., 8-Equisetum fluviatile, 9-Carex humida, 10-Phragmites australis, 11-Anemone dichotoma, L. 12-Menyanthes trifoliate, 13-Achillea acuminate, 14-Lathyrus quinquenervius., 15-Carex dispalata, 16-Salix rosmarinifolia, L. 17-Caltha palustris var, sibirica.*

group including 2 sampling spots (S28, S29). *Calamagrostis angustifolia* occurred in relatively shallow water conditions.

TWINSPAN classification matrix results reflected an obvious environmental gradient: water depth. The weighted-average wetland indicator status for each community type reflected the distribution of community types along the hydrologic gradient. The matrix diagram reflected that from association 1 to association 6 the water depth was gradually reduced, which determined the distribution range of these species.

Based on the inquisitional data of sampling sites, the biodiversity of the *Carex lasiocarpa* community in HNNR was analyzed by adopting diversity index, richness index, and evenness index.

The characteristics of the plant community can reflect vegetation functions and ecological niche. In our research, the characteristics of plant community (species richness MA, species diversity index H, and species evenness index E) were supposed to be different among different water-depth areas. Therefore, 47 quadrats were divided into 5 groups based on different water depth, and then the biodiversities indices of plant community were analyzed (Figure 6). The results demonstrated that the water depth was distinct in different vegetation types, and the optimal water depth for *Carexpseudo-curaica* was the deepest, followed by *Glyceria spiculosa*, *Carex lasiocarpa*, and *Calamagrostis angustifolia*. It was observed that the evenness index and diversity index of *Carex lasiocarpa* communities were low when water depth was too high or too low, while species evenness was poorer. When the water depth was moderate, *Carex lasiocarpa* species distributes evenly (Figure 5). Species richness of association I was high because of relatively more species, but when water depth was too high, *Carex pseudocuraica* and *Carex lasiocarpa* occured as dominant species, and other associated species were rare. *Carex pseudocuraica*, *Carex lasiocarpa,* and *Glyceria spiculosa* in association II were main dominant species, accompanying species was less, therefore species richness was low. The number of species in association III, IV, and V was more, and evenness and abundance of plant communities were higher. The growth of *Carex lasiocarpa* community vegetation was significantly correlated with water depth, which can be found from biodiversity of each association. With the increase or decrease of water depth, density of *Carex lasiocarpa* community decreased while biodiversity increased.

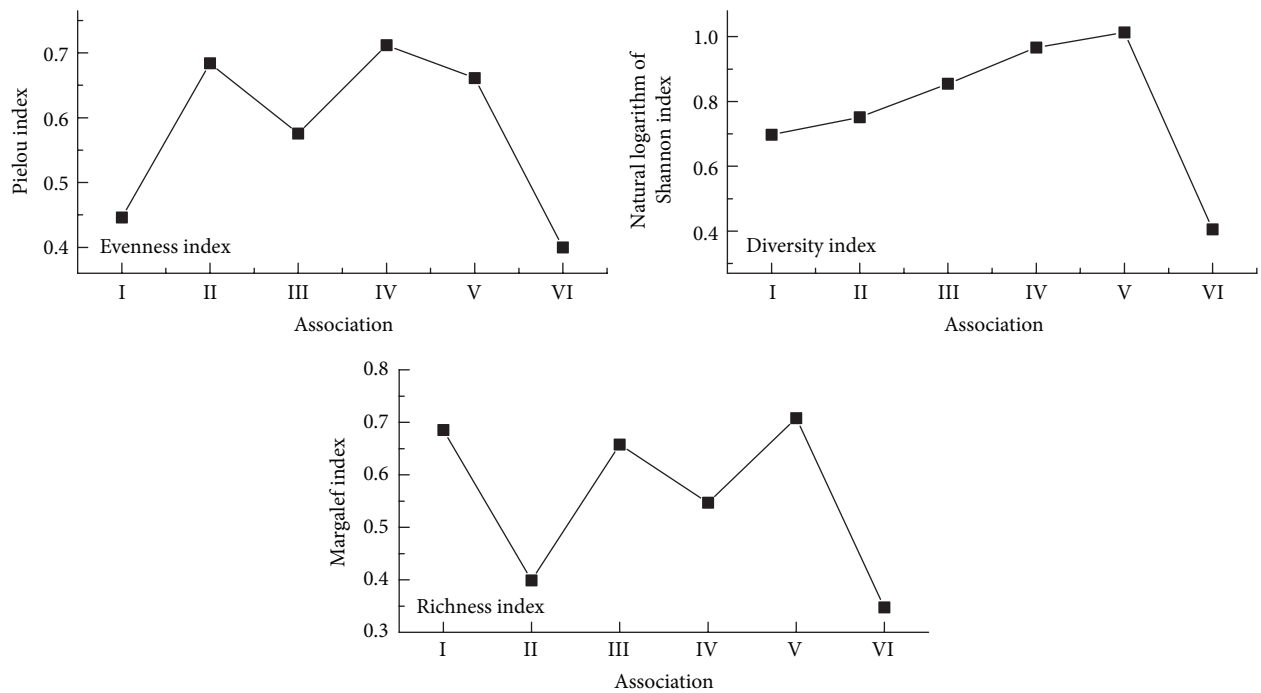

Figure 6: *Carex lasiocarpa* community biodiversity index.

4. Conclusions

The results indicated that the optimum ecological amplitude of *Carex lasiocarpa* to water depth based on population biomass was [13.45 cm, 29.78 cm], while the optimum ecological amplitude of *Carex lasiocarpa* to water depth based on average height was [2.31 cm, 40.11 cm]. The optimum ecological amplitude of *Carex lasiocarpa* to water depth was [13.45 cm, 29.78 cm] and the optimist growing point of *Carex lasiocarpa* to water depth was 21.4 cm.

TWINSPAN classification matrix results reflected an obvious environmental gradient for wetland plant species: water-depth gradient. *Carex lasiocarpa* maintains high cover across most water-depth gradients but requires high variation at the wettest conditions. Water depth for plant species in the freshwater marsh showed the order as *Carex pseudocuraica* > *Carex lasiocarpa* > *Glyceria spiculosa* > *Calamagrostis angustifolia*.

The growth of *Carex lasiocarpa* community was significantly correlated with water depth. With the increase or decrease of water depth, the densityof *Carex lasiocarpa* decreased, and the evenness index and diversity index of *Carex lasiocarpa* communities were low. In moderate water depth condition, the density of *Carex lasiocarpa* was the highest.

Acknowledgments

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Research Article

Gap Analysis and Conservation Network for Freshwater Wetlands in Central Yangtze Ecoregion

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The Central Yangtze Ecoregion contains a large area of internationally important freshwater wetlands and supports a huge number of endangered waterbirds; however, these unique wetlands and the biodiversity they support are under the constant threats of human development pressures, and the prevailing conservation strategies generated based on the local scale cannot adequately be used as guidelines for ecoregion-based conservation initiatives for Central Yangtze at the broad scale. This paper aims at establishing and optimizing an ecological network for freshwater wetland conservation in the Central Yangtze Ecoregion based on large-scale gap analysis. A group of focal species and GIS-based extrapolation technique were employed to identify the potential habitats and conservation gaps, and the optimized conservation network was then established by combining existing protective system and identified conservation gaps. Our results show that only 23.49% of the potential habitats of the focal species have been included in the existing nature reserves in the Central Yangtze Ecoregion. To effectively conserve over 80% of the potential habitats for the focal species by optimizing the existing conservation network for the freshwater wetlands in Central Yangtze Ecoregion, it is necessary to establish new wetland nature reserves in 22 county units across Hubei, Anhui, and Jiangxi provinces.

1. Introduction

Freshwater ecosystems provide considerable amount of the earth's global biodiversity and substantial ecosystem services, creating a strong imperative for their protection and restoration [1–3]. Although this unique ecosystem has been exposed to higher pressures and threats than adjacent terrestrial ecosystem, freshwater ecosystems have received less attention than terrestrial ecosystems from the conservation community [1, 4–6]. In recent years, freshwater wetlands have internationally received a growing attention due to its globally continuing decline, and freshwater conservation planning has become a newly emerged research field, especially at ecoregional scale [6–13]. However, related case study for freshwater conservation planning is still rare, more research throughout the world is needed to establish scientific conservation strategies for freshwater wetlands worldwide, especially in China where the freshwater ecosystem is unique and diverse globally [8, 14–19].

In the two past decades, gap analysis has emerged in North America as a valuable technique to assist land managers in formulating regional biodiversity conservation planning and building regional conservation networks; numerous gap analysis projects have been developed [1, 20–29]. Gap analysis is also considered to be applicable and valuable in large-scale biodiversity conservation efforts and has been receiving increased attention in China [30]. However, so far there have been few such documented studies for freshwater wetlands at ecoregional scale.

The Central Yangtze and its floodplain cover a large area with some of the world's most important and unique freshwater ecosystem, supporting a wide range of important freshwater biotas and associated habitats [31, 32]. Specifically, these habitats act as crucial staging and breeding areas for many globally endangered waterbirds. The Central Yangtze and floodplain currently hosts four Ramsar sites (e.g., Poyang Lake, West Dongting Lake, South Dongting Lake, and East Dongting Lake) and has been designated by the World

Wildlife Fund (WWF) as one of the Global 200 Ecoregions (i.e., Central Yangtze Ecoregion), which can be defined as a distinct assemblage of natural communities sharing a large majority of species, dynamics, and environmental conditions that effectively function as a conservation unit [33]. Also, its wetlands play an important role in supplying ecological services, such as microclimate stabilization, flood control, waste and pollutant mitigation, and securing a supply of ground water [34, 35]. Further, the large area of rice growing land in the Central Yangtze provides an essential contribution to regional and national sustainable development needs [34].

In the half past century, the freshwater wetlands in Central Yangtze Ecoregion and the biodiversity they support have been under the constant threat of degradation, mostly associated with human developmental pressures such as large-scale agricultural practices, land reclamation, water and flood control projects, and rapid urbanization [36]. This has resulted in significantly negative consequences, for example, increased flooding, loss of lake and wetland areas, and declines in biodiversity [31, 35, 37–39]. To reverse this trend, the WWF, in collaboration with the Chinese government, has launched a large-scale conservation initiative in Central Yangtze, which is listed as one of the 12 key protection projects in the WWF's global protection network [34]. Following the rapid socioeconomic development, the freshwater wetlands of Central Yangtze have largely fragmented into isolated habitats, and the focus of wetland conservation should therefore shift from those individual-based reserve patterns to the regional conservation network for freshwater wetlands, to ensure the long-term survival of those endangered species and the persistence of these unique freshwater habitats in Central Yangtze Ecoregion.

Due to its internationally important wetlands and globally valuable freshwater biodiversity, the Central Yangtze Ecoregion has already become the target of numerous research projects, including those aimed at detecting changes in habitats and biodiversity, analyzing underlying anthropogenic driving forces, and formulating conservation strategies [31, 37–39]. However, these previous research mainly focused on particular hotspots, such as the Ramsar Sites or nature reserves at a local scale (e.g., Dongting Lake, Poyang Lake, and the Jianghan floodplain, etc.) hence; the conservation strategies were generated based on local and site-based protection and cannot adequately be used as guidelines for a broad-scale conservation initiatives for freshwater wetlands across Central Yangtze Ecoregion. This study therefore aims at establishing a complete and efficient conservation network for freshwater wetlands by refining existing protective system based on a gap analysis of the Central Yangtze, in response to these conservation needs and with a view of ecoregion-based conservation.

2. Study Site and Methods

2.1. Site Description. WWF delineated the Central Yangtze Ecoregion mainly based on the habitat distribution of focal species (e.g., endangered waterbirds and migratory fish) as well as the ecological integrity of its river and lake basins. Geographically located between E 106°02′–118°36 and N 24°22′–34°16′, the freshwater wetlands of Central Yangtze Ecoregion are mainly composed of some large lake and river subcatchments along its mainstream, for example, Dongting Lake Basin, Poyang Lake Basin, Hanjiang River Basin, and Wujiang River Basin, (Figure 1). Its administrative units include all of Hunan Province, much of Hubei and Jiangxi Provinces, the western part of Anhui Province, and small parts of Fujian, Guangdong, Guangxi, Guizhou, Shanxi, Sichuan, and Zhejiang provinces, covering a total area of $7.55 \times 10^5\,\text{km}^2$ and accounting for 41.92% of the overall Yangtze River watershed.

Table 1: Grading and coding the habitat suitability factors of elevation, slope, and landuse/landcover types. Codes of habitat suitability are composed of elevation, slope, and landuse/landcover and are designed to facilitate grid-based operations in GIS. For example, the code 11125 represents an area of habitat with an elevation code of 11 (i.e., 0–20 m), a slope code of 1 (0–5°), and the landuse/landcover code of 25 (i.e., surface water).

Elevation (m)	Code	Slope	Code	Landuse/landcover type	Code
0–20	11	0–5	1	Evergreen needleleaf forest	11
20–50	12	5–10	2	Evergreen broadleaf forest	12
50–100	13	10–15	3	Deciduous needleleaf forest	13
100–200	14	15–20	4	Deciduous broadleaf forest	14
200–500	15	20–25	5	Mixed forest	15
500–1000	16	25–30	6	Closed shrub	16
1000–1500	17	30–35	7	Open shrub	17
1500–2000	18			Shrub and meadow	18
2000–2500	19			Grasslands	19
2500–3000	20			Marsh	20
>3000	21			Cropland	21
				Urbanized area	22
				Cropland/natural vegetation mosaic	23
				Sparsely vegetated area	24
				Surface water	25

2.2. Data Source. The landuse/landcover data were derived from the International Geosphere-Biosphere Programme (IGBP) Global Land cover Database [40] with a grid resolution of 1 km × 1 km (Figure 2). A digital elevation model (DEM) of the Central Yangtze Ecoregion was obtained and extracted from SRTM 90m Digital Elevation Database Version 4.0 [41] (Figure 3), from which the main topographic data (i.e., elevation and slope gradient) were extracted and classified according to their differences in ecological influence based on WWF guidelines (Table 1). Slope gradients were categorized into 6 levels, 0–5°, 5–10°, 15–20°, 20–25°, 25–30°, and >30° (sites with a slope gradient higher than

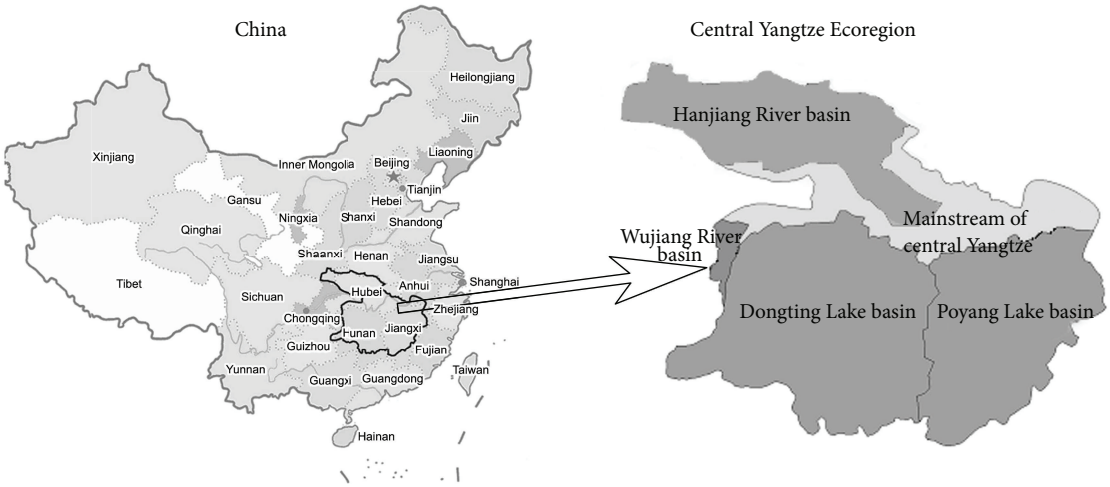

Figure 1: The geographical boundary of Central Yangtze Ecoregion, including its mainstream area and associated large lake and river subcatchments, that is, Dongting Lake Basin, Poyang Lake Basin, Hanjiang River Basin, and Wujiang River Basin, covering an area of 7.55×10^5 km^2 and accounting for 41.92% of the overall Yangtze River watershed.

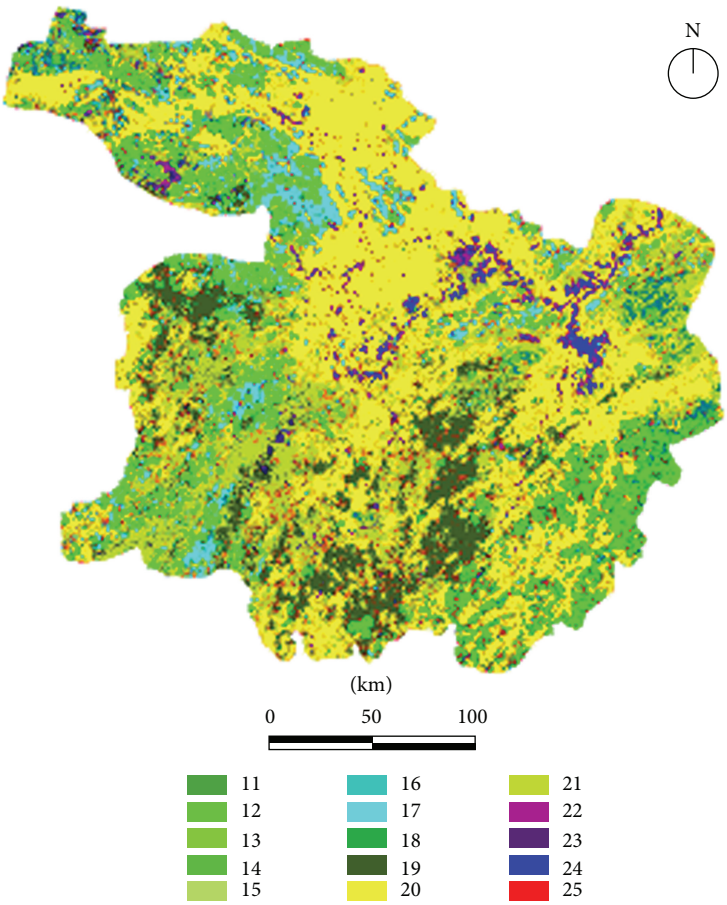

Figure 2: The landuse/landcover data of Central Yangtze Ecoregion obtained and extracted from IGBP 2000 Global landuse/landcover database (11: evergreen needleleaf forest; 12: evergreen broadleaf forest; 13: deciduous needleleaf forest; 14: deciduous broadleaf forest; 15: mixed forest; 16: closed shrub; 17: open shrub; 18: shrub and meadow; 19: grasslands; 20: marsh; 21: cropland; 22: urbanized areas; 23: cropland/natural vegetation mosaic; 24: sparsely vegetated area; 25: surface water).

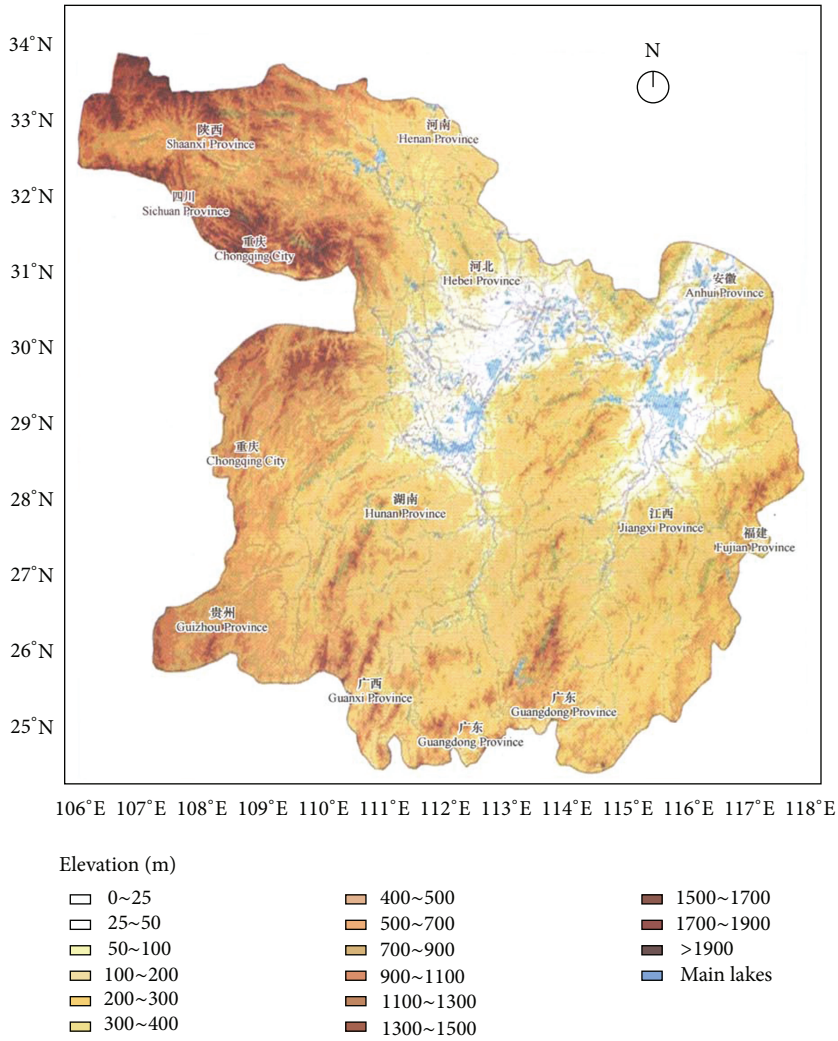

Figure 3: A digital elevation model (DEM) of Central Yangtze Ecoregion derived from SRTM 90m Digital Elevation Database Version 4.1.

30° can rarely be used as habitats for the focal species considered here). Elevations were divided into 11 levels, 0–20 m, 20–50 m, 50–100 m, 100–200 m, 200–500 m, 500–1000 m, 1000–1500 m, 1500–2000 m, 2000–2500 m, 2500–3000 m, and >3000 m.

2.3. Gap Analysis. To facilitate habitat analysis, the Habitat Suitability Unit (HSU) Index was used to characterize the main habitat types of the focal species. The HSU was defined as the combination of dominant ecogeographical factors affecting habitat suitability (i.e., elevation, slope gradient, and landuse/landcover types). To support the analysis of potential habitats and conservation gaps, a GIS-based spatial dataset of HSUs was built in which a combination of these three factors was used to create a specific HSU type.

A group of focal species was used in the habitat assessment and gap analysis. The focal species were identified by the following criteria suggested by the WWF and some previous research studies [42, 43]: (1) internationally important species (e.g., species listed in the International Union for Conservation of Nature Red Data Book or famous flagship species that draw considerable public attention); (2) nationally important species, listed in the top or second class of the National Conservation Inventory; (3) endemic species exclusively living in certain areas or habitat types; and (4) umbrella species, whose habitat requirements incorporate the needs of other species. Conserving habitats sufficiently to protect umbrella species can often concurrently protect other species residing within the same ecoregion [43, 44]. Based on these standards, four endangered waterbirds were identified as focal species: Siberian Crane (*Grus leucogeranus*), Oriental White Stork (*Ciconia boyciana*), Lesser White-fronted Goose (*Anser erythropus*), and Chinese Merganser (*Mergus squamatus*). To conduct habitat analysis, the existing habitat distribution data of the focal species was identified by WWF experts in a series specific symposiums organized by WWF in China in 2004 (Figure 4).

The habitat analysis was based on the hypothesis that if field records indicated that a focal species could be found within a certain area, then the dominant combinations of ecogeographic factors (i.e., elevation, slope gradients, and landuse/landcover types) would constitute the main types of

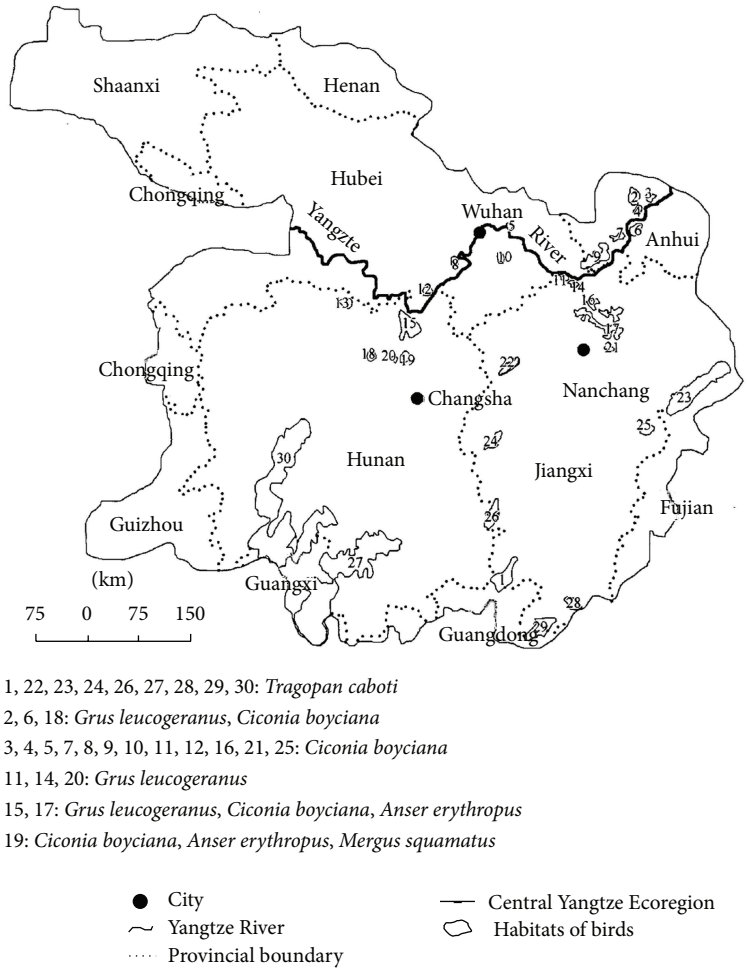

Figure 4: Existing habitats of the focal species in the Central Yangtze Ecoregion identified by the WWF.

HSU preferred by the focal species. Through overlaying the spatial data of the ecogeographic factors using GIS, the spatial linkages between the HSUs and the focal species were thus built by detecting and identifying the dominant HSUs for each focal species within their existing habitats. These main HSU types of the focal species were then further screened and refined based on their detailed habitat information obtained from the literature review and expert consultation to minimize errors in the analysis. In this manipulation, a large variety of combinations of ecogeographic factors were produced, but most were deleted from the HSU spatial dataset because of insignificant spatial linkages to the habitats as demonstrated by their minor area contribution to the existing habitats. Here, considering the concept of minimum critical area for the long-term survival of focal species, we chose HSU types from these combinations of ecogeographic factors based on the criteria that their area contribution should exceed 5% of the total existing habitat area.

The key objective of gap analysis is identifying those potential habitats (i.e., biodiversity hotspots) which have been excluded from existing conservation systems. To analyze potential habitats, we supposed that if a habitat of a focal species can be represented by a group of HSUs, then other unsurveyed habitats sharing the same HSUs within a certain ecoregion can be considered the potential habitats for the focal species. Accordingly, potential habitats can be extrapolated and predicted at a larger scale by GIS-based spatial linkage between the species and the associated types of HSU (Figure 5). To verify the predicted results of the potential habitats, the existing status of the wetlands in those counties with unprotected potential habitats was examined through a literature review and the latest monitoring data from the State Forestry Administration of China.

Based on the above potential habitat analysis, conservation gaps (i.e., unprotected potential habitats) were located by comparing protected areas (i.e., existing nature reserves) with potential habitats. Then, the conservation network of the Central Yangtze Ecoregion could be established by combining the conservation gaps with the existing conservation system (Figure 6).

Because precise spatial data of the nature reserves is lacking (especially at provincial and local levels), the county units were therefore used as the basic spatial units for the gap analysis and conservation network planning in our research.

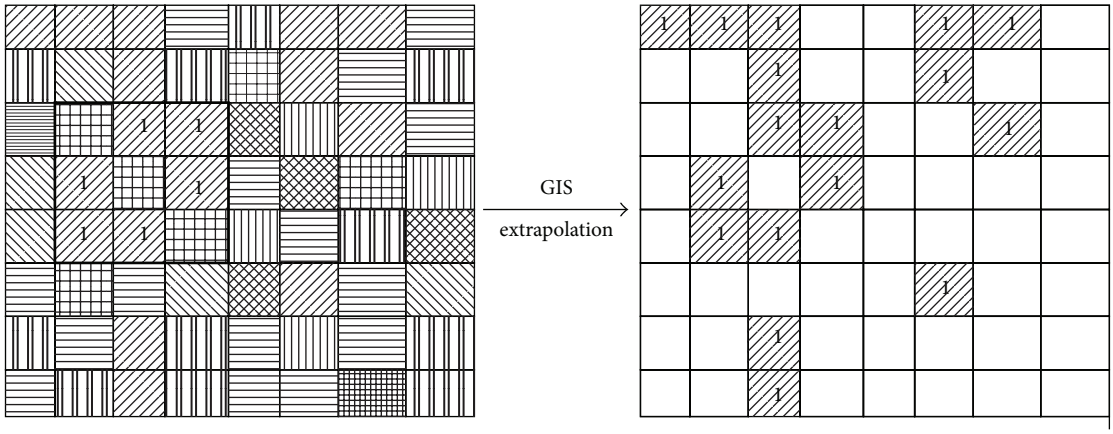

Figure 5: The sketch map to analyze the potential habitats of the focal species.

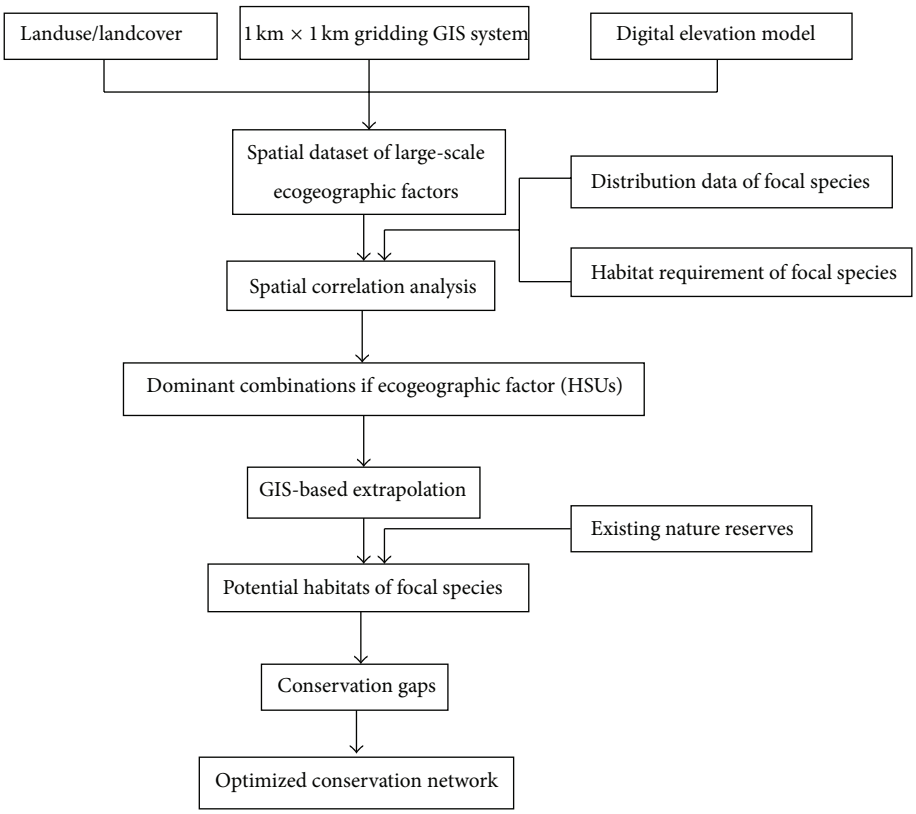

Figure 6: The flow chart to develop optimized wetland conservation network in Central Yangtze Ecoregion.

We assumed if only one county has at least one wetland nature reserve, then the freshwater wetlands and the associated biodiversity in the county could be effectively protected.

3. Results

The results of habitat analysis showed that the surveyed distribution area of Siberian Crane is 2201.4 km^2, including 33 types of ecogeographic combinations, of which four types were identified as HSUs with total area contribution of 81.1% to the potential habitat, including the types of 11125 (44.2%), 12123 (16.4%), 12125 (11.8%), and 11123 (8.7%). 45 types of ecogeographic combinations of oriental white stork were generated from its surveyed distribution area (3567.1 km^2), of which four types, that is, 11125, 12123, 11123, and 12125 were extracted as HSUs, accounting for 49.2%, 14.6%, 9.4%, and 9.1% of the potential habitat, respectively, and totally contributed 82.3% to the potential habitat. With regard to lesser white-fronted goose, there are 20 ecogeographic types within its distribution area (1761.5 km^2), of which five types were extracted as HSUs (i.e., 11125, 12123, 12125, 11123, and 11117), accounting for 49.9%, 14.8%, 12.7%, 6.7%, and 6.6%

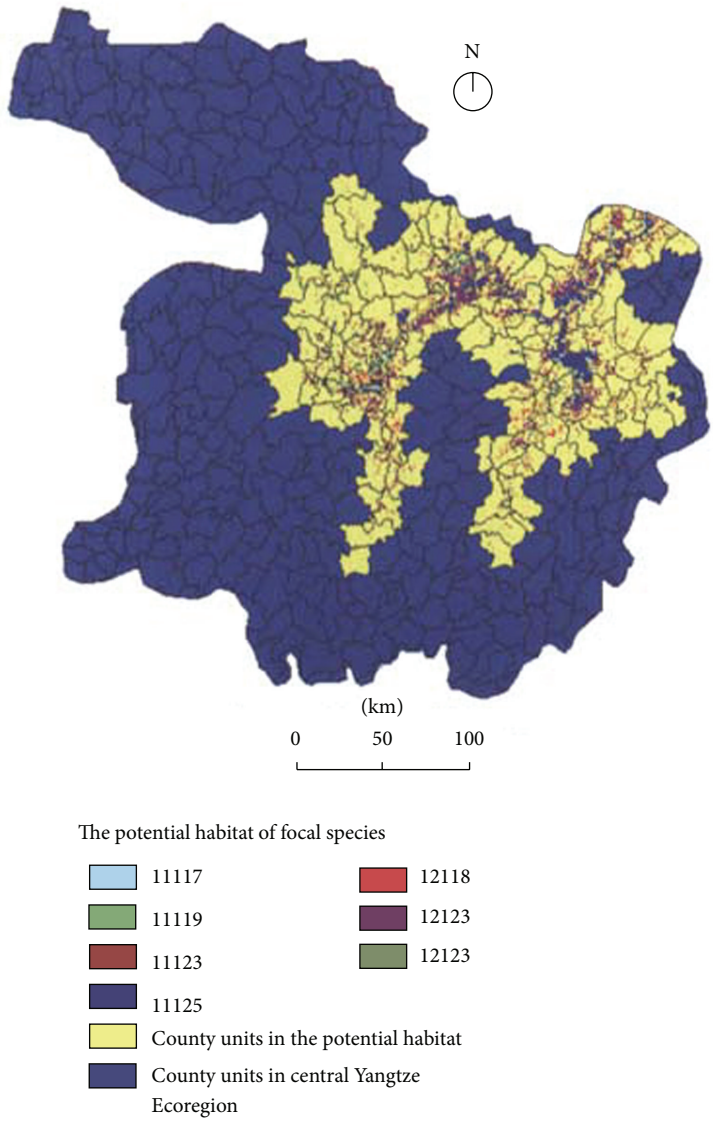

Figure 7: Potential habitats of focal species in the Central Yangtze Ecoregion. All potential habitats cover an area of 32,050 km^2 and are mainly composed of seven types of Habitat Suitability Units (HSUs), including 132 county units of four provinces, for example, Hubei, Hunan, Jiangxi, and Anhui provinces.

of the potential habitat, respectively, and totally contributed 90.7% to the potential habitat. With the smallest distribution area of 161.2 km^2, Chinese Merganser had only 13 ecogeographic types, of which the 6 types were recognized as HSUs (i.e., 11123, 11125, 11119, 12123, 11117, and 12118), accounting for 24.2%, 19.9%, 16.1%, 11.8%, 10.6%, and 5.0% of the potential habitat, individually, and totally contributed 87.6% to the potential habitat.

Through habitat analysis for the focal species, seven HSU types were identified from the large varieties of combinations of ecogeographic factors, 11125, 12123, 12125, 11123, 11119, 11117, and 12118, accounting for 84.0% of the overall potential habitat area (Figure 7 and Table 1 explain these codes for HSUs). The results revealed the core potential habitats represented by the dominant HSUs (i.e., 11125 and 12125) are characterized by surface water in the Central Yangtze floodplain, which occupies 47.30% of the total potential habitats and is mainly made up of shallow wetlands of the main lakes (e.g., Dongting Lake, Poyang Lake and Honghu Lake) in the Central Yangtze Ecoregion. Various waterbirds prefer these HSUs and core potential habitats as foraging and resting habitats. Areas dominated by sedges, meadows, and open shrubs (i.e., 11119, 11118, and 11117) form secondarily important HSUs of potential habitats, occupying 26.68% of the total potential habitats. These HSUs are especially preferred by wintering waterbirds as their core habitats. Also, the ecotones between croplands and natural vegetation (i.e., 11123 and 12123) function as complimentary potential habitats for focal species; in particular, they provide important forging habitats for some endangered large wading birds, such as white crane and oriental white stork. The overall potential habitats of these dominant HSUs include parts of 132 county units with

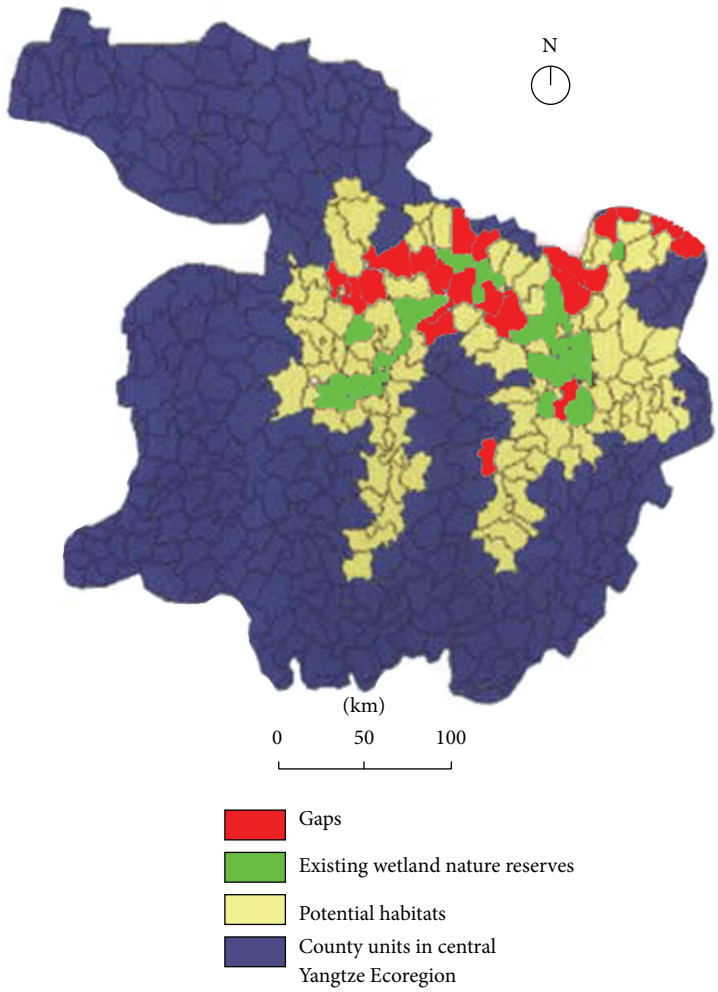

Figure 8: Existing wetland nature reserves, potential habitats, and conservation gaps in the Central Yangtze Ecoregion. The conservation gaps were identified by comparing potential habitats with existing wetland nature reserves, while optimized conservation network can be established by combining the existing nature reserves with the proposed nature reserves based on the gap analysis.

a total area of 32,050 km^2. Of these, the top four counties rich in potential habitats are Wuchang (1575 km^2, Hubei Prov.), Susong (1476 km^2, Anhui Prov.), Jinxian (1217 km^2, Jiangxi Prov.), and Poyang (1143 km^2, Jiangxi Prov.), which account for 78.4%, 61.7%, 62.3%, and 27.1% for the administrative areas of these counties and contribute 4.9%, 4.6%, 3.8%, and 3.6% to the total potential habitats, respectively.

Currently, 16 wetland nature reserves have been established in the Central Yangtze Ecoregion, which incorporate the most ecologically valuable parts of the potential wetland habitats, with a total area of 7530 km^2, including national nature reserves (NNRs) such as the Dongting Lake NNR and the Poyang Lake NNR. However, our analysis revealed that most of potential habitats are still exposed to human impacts, of which only 23.49% was included into the existing wetland nature reserves. Also, the existing conservation pattern presented by the county units seems disorganized and fragmented and can hardly provide a long-term and large-scale conservation in Central Yangtze. The result underscores the urgent need to optimize the existing conservation pattern by filling the conservation gaps and establishing a conservation network in Central Yangtze.

The significant conservation gaps for freshwater wetlands in Central Yangtze could be identified in counties rich in potential habitats but unprotected by existing nature reserves. These conservation gaps were further refined based on the following criteria: (1) unprotected potential habitats adjacent to existing nature reserves should be selected as gaps, as existing nature reserves act as the core habitats in the Central Yangtze Ecoregion, (2) the selected gaps should be ecologically integrated with existing nature reserves so as to form an optimized conservation network with an interlinked conservation pattern, and (3) those county units sharing a larger proportion of potential habitats should be given priority so that the conservation network can provide effective protection for the potential habitats with the least land cost.

After screening, 22 county units were categorized as conservation gaps, including 13 in Hubei Province, eight in Anhui Province, and one county of Jiangxi Province (Figure 8 and Table 2). Thus, the combination of existing (23 county units)

Table 2: Conservation gaps of wetlands in the Central Yangtze Ecoregion.

No.	County	Wetland list	Province
1	Tongling	Dayehu lake	Anhui
2	Tongcheng	Caizihu lake	Anhui
3	Congyang	Lakes of Baidanghu, Chengyaohu, and Caizihu	Anhui
4	Taihu	Hualiangting reservoir	Anhui
5	Susong	Pohu lake	Anhui
6	Wangjiang	Lakes of Wuchanghu and Pohu	Anhui
7	Lujiang	Lakes of Caohu and Huangpihu	Anhui
8	Wuwei	Caohu lake	Anhui
9	Nanchang	Jiang'an Valley Wetlands	Jiangxi
10	Daye	Dayehu lake	Hubei
11	Wuchang	Lakes of Qingxunhu, Luhu, and Futouhu	Hubei
12	Huangpi	Wuhu lake	Hubei
13	Xinzhou	Lakes of Wuhu and Zhangduhu	Hubei
14	Shashi	Hujiang Wetlands	Hubei
15	Hanchuan	Chahu lake	Hubei
16	Chibi	Huanggaihu lake	Hubei
17	Jiayu	Xilianghu lake	Hubei
18	Yangxin	Lakes of Wanghu and Dayehu	Hubei
19	Xiantao	Paihu lake	Hubei
20	Tianmen	Chahu lake	Hubei
21	Qianjiang	Changhu lake	Hubei
22	Jiangling	Changhu lake	Hubei

and proposed (22 county units) conservation systems constitutes an ecologically optimized conservation network for freshwater wetlands in the Central Yangtze Ecoregion, which can be expected to effectively conserve 84% of total potential habitats of focal species in Central Yangtze Ecoregion.

4. Conclusions

Our research indicated that a number of wetland nature reserves have been established in the Central Yangtze, but the existing wetland nature reserve system is still far from being effective in conserving the freshwater biodiversity represented by the focal species in Central Yangtze Ecoregion. Our habitat analysis shows that the potential habitats for the focal species in the Central Yangtze Ecoregion include parts of 134 county units, of which the existing wetland conservation system only covers 23.49%. Large parts of these potential habitats are not included in the current protection system and are exposed to human activities such as agricultural development, hydrological projects, and urbanization. Moreover, the existing conservation areas are fragmented and isolated from each other, and so the existing conservation system in Central Yangtze must be adjusted and optimized.

In consideration of maximized representativeness (e.g., proportion of potential habitats) and connectivity of the conservation system with minimized land cost, the optimized conservation network for the freshwater wetlands in Central Yangtze Ecoregion could be established by integrating the existing wetland conservation system with the identified conservation gaps. This optimized wetland conservation network would effectively protect over 80% of the potential habitats for the focal species in Central Yangtze Ecoregion. It would comprise 45 county units across the Central Yangtze, of which 22 would need to establish new protected areas to fill their conservation gaps, including 13 counties in Hubei Province, eight in Anhui Province, and one county in Jiangxi Province.

In our research, although the accuracy of the current habitat analysis may have been restricted by the resolution (1 km × 1 km) of the landuse/landcover classification due to the hugescale of the research area, the potential habitats revealed by the research, especially those counties with larger proportion of conservation gaps, can be explained and verified by existing documentation. Also, our results show that all of the existing wetland natural reserves were included in the identified potential habitats, indicating that it is applicable to employ Habitat Suitability Units (HSU) and GIS-based habitat extrapolation in large-scale gap analysis.

Acknowledgments

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Research Article

Impacts of Intensified Agriculture Developments on Marsh Wetlands

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A spatiotemporal analysis on the changes in the marsh landscape in the Honghe National Nature Reserve, a Ramsar reserve, and the surrounding farms in the core area of the Sanjiang Plain during the past 30 years was conducted by integrating field survey work with remote sensing techniques. The results indicated that intensified agricultural development had transformed a unique natural marsh landscape into an agricultural landscape during the past 30 years. Ninety percent of the natural marsh wetlands have been lost, and the areas of the other natural landscapes have decreased very rapidly. Most dry farmland had been replaced by paddy fields during the progressive change of the natural landscape to a farm landscape. Attempts of current Chinese institutions in preserving natural wetlands have achieved limited success. Few marsh wetlands have remained healthy, even after the establishment of the nature reserve. Their ecological qualities have been declining in response to the increasing threats to the remaining wetland habitats. Irrigation projects play a key role in such threats. Therefore, the sustainability of the natural wetland ecosystems is being threatened by increased regional agricultural development which reduced the number of wetland ecotypes and damaged the ecological quality.

1. Introduction

Natural ecosystems, especially freshwater ecosystems in the inland flood plain, are undergoing profound and extensive disturbances by humans worldwide [1–5]. A key indicator of these disturbances is that humans extensively reclaim natural wetlands to expand their economic benefits. Therefore, most habitats of natural ecosystems have been changed into farms or urban areas rapidly and continuously [6–8]. The disturbances have been representatively observed in China, the largest developing country in the world. A good example is the shrinking process of the marsh wetland landscapes on the Sanjiang Plain in Northeast China [9, 10].

With its rapid development, China can be regarded as a typical country of most other developing countries in the world. China has experienced high-speed development in the past 30 years. Scientifically assessing or even imagining the impact of urbanization and agricultural reclamation on natural ecosystems is difficult because few countries have comparably rapid and extensive development [11, 12]. During the past 30 years, a large number of natural habitats in China have been reclaimed into cropland, and numerous farmlands have been occupied and then urbanized into towns or cities [7]. With this progress, the Chinese population has rapidly increased and is currently 1.3 billion. The most natural habitats of the wetland ecosystems have been encroached upon during this progress [13]. Though food security is always the top priority for the massive Chinese population [10], the continuous reclamation of the few remaining natural habitats has difficulty meeting the demand of grain production.

Some developing countries, such as China, have published various administrative policies for natural resource protection during their rapid developmental stages. Many natural reserves have been established in the past few years. China has listed the most natural reserves in the world [14]. However, the institutional efficacy of these reserves remains questionable from a scientific perspective [15, 16]. In this paper, the Honghe National Nature Reserve (HNNR) was included within our study area as a wetland reserve. It is also an international wetland listed by the Ramsar Convention.

The institutional efficacy of this Chinese natural reserve was the topic of the present study. Researchers analyzed the spatiotemporal changes of the inner and outer landscapes of the reserve and reached some interesting scientific conclusions.

Chinese scholars have recently become concerned about the great changes in the natural marsh wetlands in China. Many papers have reported research results in this field [9, 13, 17–30]. In these studies, some researchers [9, 13, 31–33] analyzed the marsh landscape on the Sanjiang Plain over periods of 20 or even 50 years. Most research approaches were based on theories of landscape ecology. The integration of remote sensing techniques and geographical information systems was applied for the spatiotemporal analysis of marsh landscape segments. Landscape investigators obtain dynamic information on marsh landscapes with the support of remote sensing techniques [34]. However, these studies lack an analysis on the profound driving forces that impact the wetlands and especially lack a correlational analysis of the linkage between policy issues and regional characteristics that deal with the spatiotemporal dynamics of the marsh wetlands. These previous studies focused more on obtaining data and analyzing dynamic wetland landscapes on large regional scales (e.g., 10000 km^2), which is suitable for the application of remote sensing techniques [35]. Liu and Ma descriptively studied the changes in the natural environments on the entire Sanjiang Plain and its regional ecological response to such changes [9]. Rich survey data and historical statistics of wetlands were used in their study, but the spatiotemporal dynamics of the wetland landscapes were poorly assessed.

Many papers have studied the issue of land use and cover change caused by regional and international urbanization in the past few decades. An abundance of literature has addressed the impact of urbanization and regional development that have encroached on cropland or the reclamation of wild fields in China [10, 11, 36]. Most studies have focused on the spatiotemporal characteristics of changing land use or land cover or have analyzed the relative driving forces. Ecological impact issues related to agricultural activity have long been neglected [14]. Little research has focused on the impact on wetland ecology, linked the dynamics of the marsh landscape over the long term, and studied the driving forces of regional agriculture with a background analysis of historical national policies [7]. This paper provides a case study of the Sanjiang Plain in Northeast China and demonstrates the shrinking process of the typical marsh wetland and other natural landscapes driven by agricultural activity. The ecological impacts on the wetland ecosystems were also analyzed from a regional development perspective. This research will help better understand the gradual evolution of the disturbed natural ecosystems and elucidate the dependence of these natural ecosystems in developing countries. The goal is to help resource administrators determine the evolutionary direction of these ecosystems in the future [37, 38]. An identification of the common characteristics of these natural ecosystems will significantly impact decision making in the management of surviving natural ecosystems in developing countries [38, 39].

The present study sought to achieve three objectives: (1) present the spatiotemporal process of the encroachment of expanding farmland on wild marsh landscapes in the core area on the Sanjiang Plain since 1975, which is a microcosm of shrinking natural wetland ecosystems worldwide; (2) analyze the characteristics of the driving forces that continuously reduce the marsh wetland area in this region, with an emphasis on discussing Chinese policies related to intensified agricultural development on a local scale; and (3) study the negative impact of marsh reclamation on natural ecosystems. An international wetland is used as a typical example to show readers the ecological impact of agricultural activity on marsh wetlands and assess the functional efficacy of this natural reserve.

2. Materials and Methods

2.1. Study Area. The HNNR and its three surrounding farms (Yaluhe Farm, Honghe Farm, and Qianfeng Farm) were selected as our study area. The study area is located in the northeast region of Heilongjiang (47°25′N-48°1′N, 133°18′E-134°5′E), the core area of the Sanjiang Plain (Figure 1). It covers 2416.8 km^2 in the neighboring area of Tongjiang County and Fuyuan County. This area was a unique marsh wetland landscape 30 years ago. The establishment of local farms coincided with a gradual loss of the marsh wetlands. The establishment of the HNNR was useful for obtaining data on the later progression [40]. Therefore, our study area selection of both the HNNR and its surrounding farms was helpful for comparing and analyzing marsh wetland loss and the negative impacts of neighboring agricultural activity on the marsh landscape in the HNNR.

2.2. Methods. The database for this research derived mostly from LANDSAT satellite images. It included one MSS image from July 25, 1975, and two TM images from June 12, 1989, and August 30, 2006. Additional materials used for this research included a geographical map (1 : 100000 scale) and a QuickBird image with a high spatial resolution of 0.61 m from May 16, 2004. All of the landscape maps in raster format that were interpreted from the images were inputted into the ArcGIS 9.2 platform, in which a spatial resolution of less than 0.5 pixels was attained with the aid of a 1 : 10000 scale geographical map. The statistical analysis was complemented with the dynamics of local landscapes during the past 30 years using Excel 2003 software after careful topological examination in the ArcGIS. Data sources about current wetland plant survey and water fowl survey came from our field survey, and the comparable historic data source came from previous research publication (see details in Section 3.4).

A classification system of the landscapes needs to be based on the specific objectives of the research, and the hierarchical characteristics of a classification system need to match the corresponding spatial scale of the research. This research focused on the historical exchange between the natural landscapes and artificial landscapes according to the spatiotemporal information generated from the satellite

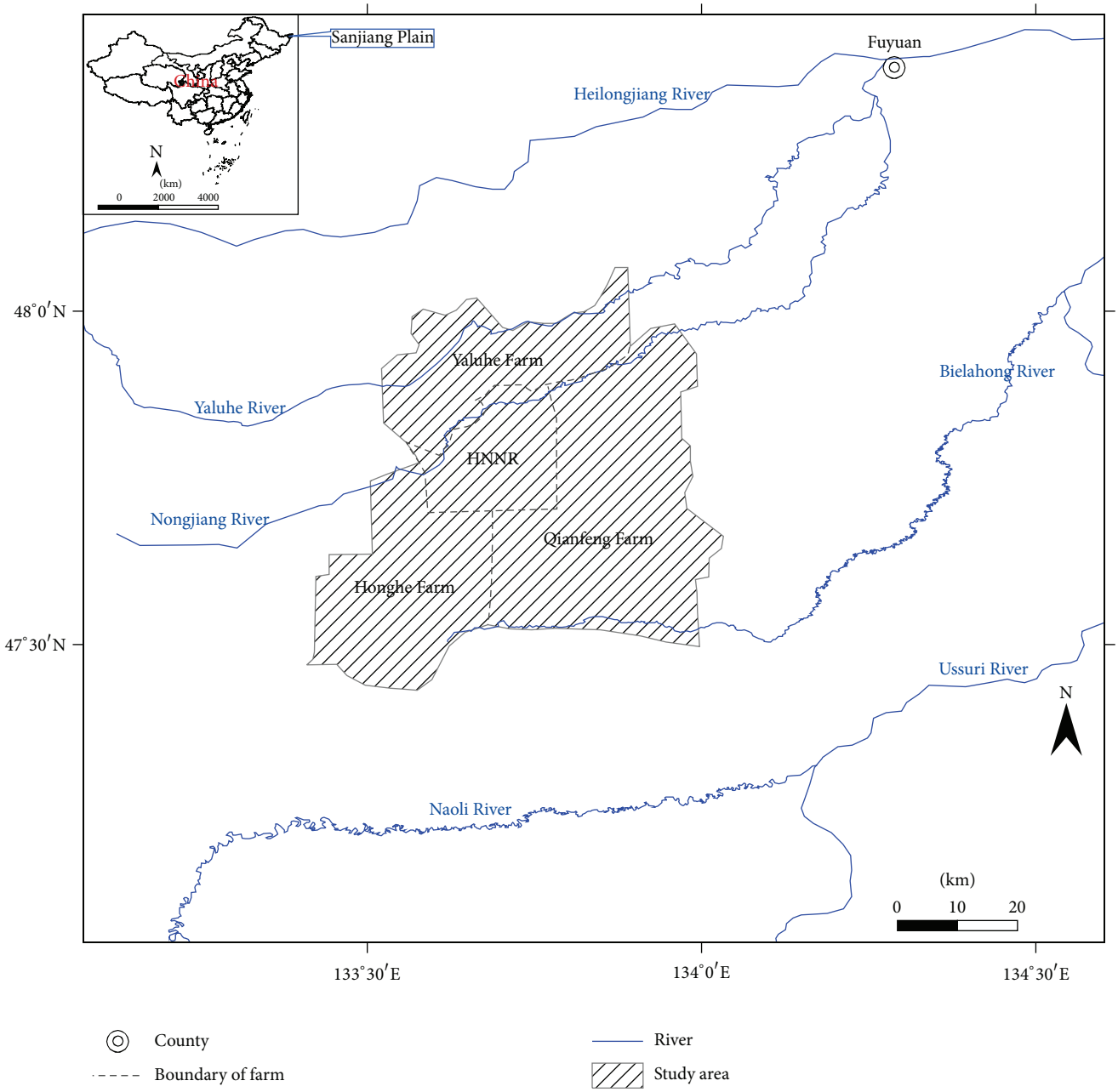

Figure 1: Location of the study area.

images on three different dates. The landscapes were classified into seven basic classifications that included three ecotypes to analyze the various landscape information in the images. The seven landscape classifications included marsh, river pond, meadow, forest, paddy field, dry farmland, and others. Among these, the river pond classification comprised natural rivers, ponds, and all other artificial water bodies. Few areas included residences in the study area between 1975 and 1989, although this increased in 2006. For an easier historical comparison of the different landscape classifications, residential areas, road areas, and other types of small landscapes were merged into one landscape classification termed "other."

The data processing method for this research included constructing a new multiple-band file for georeferenced remote sensing images and a mask for the boundary of the study area within the ENVI 4.0 platform. The mask was applied to the imagery data for the purpose of creating an image-based region of interest in the three specific dates. We utilized the layer stacking tool to construct a new file and then performed rapid filter enhancement on the images to meet the needs of image interpretation. The interpretation signs were then established, based on the images according to different colors, shapes, textures, and field investigation photographs. Manual interpretation was used to obtain the

classification maps in raster format to describe the regional wetland landscapes in 1975, 1989, and 2006. The QuickBird image was used for reducing the uncertainty while manually delineating the similar landscapes, such as marsh and meadow. After resetting the digital boundaries of four inner units as the HNNR and three farms within the study area, the three thematic maps of the wetland landscapes were reproduced for dynamic analysis purposes (Figures 2(a), 2(b), and 2(c)). An accuracy estimation was made based on the confusion matrices generated from the database of ground truth and a variety of relevant maps (e.g., the previous land-use maps and a previous classification map of the wetlands) [16, 41]. The results of the accuracy assessment showed that the total classification accuracies reached 92.33%, 92.60%, and 90.41% in 1975, 1989, and 2006, respectively. The kappa coefficients ($N = 365$) were 86.66%, 89.47%, and 86.93%, respectively. Finally, a statistical analysis was performed to present the temporal and spatial changes of the regional dynamic landscapes using Excel 2003 software [41].

3. Results and Discussion

3.1. Basic Changes of the Landscapes in the Study Area. The progression of gradual marsh landscape loss could be described quantitatively in the study area by comparing and analyzing the dynamic information from the three landscape maps in 1975, 1989, and 2006 (Figures 2(a), 2(b), and 2(c)). The basic marsh landscape in purple changed into farm landscapes in yellow as the present basic landscapes. A very substantial change of the landscapes occurred in the study area, from the 67.1% of the marsh wetland area in 1975 to 73.1% farmland area in 2006. In 1989, the typical marsh wetland loss was 47.4% compared with 1975, and the loss was 89.8% in 2006. The marsh landscape shrank in the HNNR, with a few odd marsh wetlands in the farm areas.

In the past 30 years, a large loss of rivers and ponds occurred during the progression of marsh loss. The landscape in blue lost 53%, and the natural forest loss was 58.2% since 1975. During the progression of the basic natural landscape of the marsh wetlands changing into an agricultural landscape, the dry farmland landscape changed to an increasing number of paddy fields. No paddy fields existed in the study area in 1975, but this landscape comprised one-third of the study area in 2006. A large amount of dry farmland was replaced by paddy fields with the extensive development of agricultural irrigation, which had a very negative impact on the regional marsh wetlands. The few remaining marsh wetlands degraded into meadows because of the loss of healthy habitats attributable to irrigation activity. Therefore, the area of the meadow landscape has seen a nearly 32.3% increase even after most of the original meadows were reclaimed into croplands in the past 30 years.

3.2. Progression and Characteristics of Encroachment on Marsh Wetlands. Two matrices of the landscape changes were made for the two periods according to the three landscape maps in 1975, 1989, and 2006 based on interpretations of the satellite images (Table 1, Table 2). From these, we analyzed how the marsh wetlands shrunk while the farm landscapes increased in the study area.

Table 1 shows the apparent loss of marsh wetlands from 1975 to 1989, during which a large amount of marsh wetlands were reclaimed into dry farmland or paddy fields. A 47% loss of the marsh area occurred, and the area of dry farmland increased by 380%, a four-fold increase compared with 1975. In 1989 paddy fields comprised 15% of the study area, while in 1975 almost no paddy fields existed. Twenty percent of the forest area was reclaimed into crop land or for other purposes. The originally existing marsh wetlands were the basic landscape in the study area in 1975, and natural marsh, river, and pond landscapes comprised nearly 90% of the area. Few dry farmlands existed during that time. However, the basic marsh landscape was replaced by a landscape pattern consisting of nearly 40% farmlands in 1989, with dry farmlands being the principle landscape. No significant changes occurred to the other landscapes during this period.

Table 2 shows that the marsh wetlands continued to be lost with a change ratio of over 80%, and the area decreased from 35.3% in 1989 to 6.9% in 2006. At the same time, the other natural landscapes, such as river, pond, and forest, also continuously decreased, with an average loss ratio of 50%. The progression of shrinking natural landscapes coincided with the expansion of farmlands, similar to what happened during the previous period, but some new trends appeared in the change of the landscapes from 1989 to 2006. A substantial change in the farm pattern was a 131% increase in the paddy fields during that period. The dual progression occurred as natural landscapes changed to farm landscapes while dry farmlands were replaced by paddy fields.

3.3. Impacts on the Marsh Wetland Habitat due to the Intensified Agriculture Development. The uniformity of the changing landscapes in the study area includes the three surrounding farms that experienced a rapid change from a basic marsh landscape to an agricultural landscape, although they experienced different agricultural progressions and retain different landscape structures as a result of regional development (Figure 3). We concluded that the impacts on the natural wetland habitat caused by marsh reclamation have two characteristics. First, it reduced the area of the marsh wetland habitat directly. Wetland habitats for wildlife and plants were lost largely because of the rapid decrease in the marsh wetlands in the study area. The remaining marsh wetlands became fragmented from a landscape perspective. Second, reclamation weakened the ecological function of the remaining marsh wetlands as habitats. The remaining marsh wetlands lost their healthy habitats because environmental flow was cut or reduced as a result of agricultural irrigation systems that were strengthened continuously on the neighboring farms.

Well known as a natural "gene bank" of most wildlife on the Sanjiang Plain in China, the HNNR was established in 1984 and was upgraded to a national reserve in 1996. In 2002, it was listed in the Ramsar Convention as an international wetland reserve [42]. This reserve is a location of the original typical marsh wetland on the Sanjiang Plain.

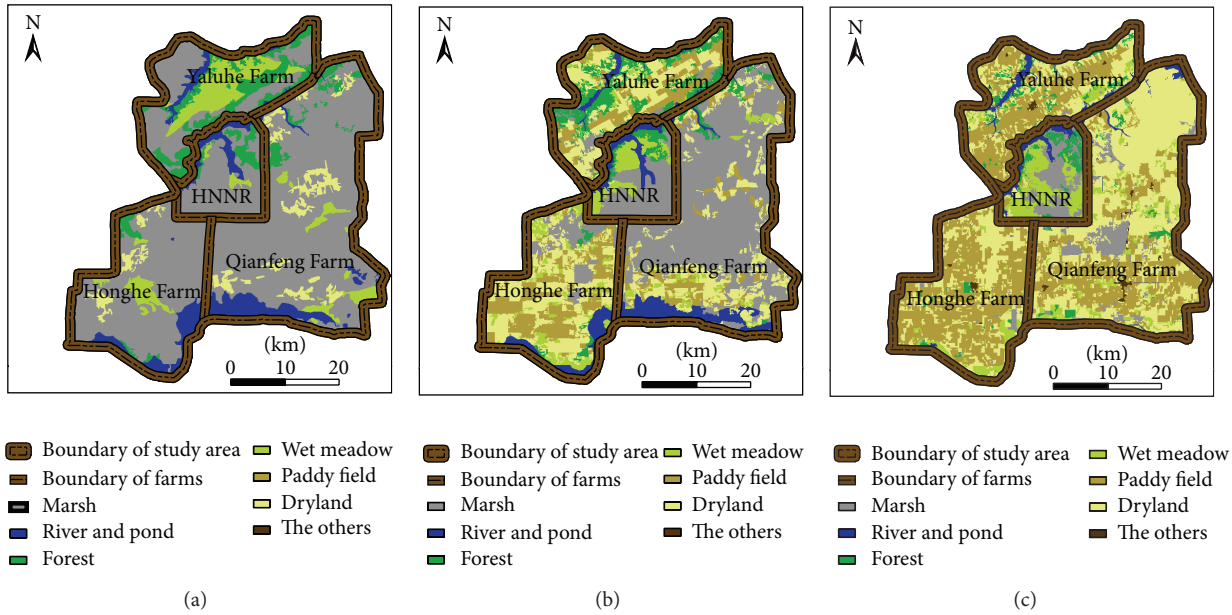

Figure 2: Changes of the wetland landscape within the past 30 years.

Table 1: Transformation matrix of landscape and land use within the study area during the period from 1975 to 1989 (unit: km^2).

1975	1989								
	Marsh	River and pool	Forest	Meadow	Paddy field	Dry farmland	Other types	Total	Proportion (%)
Marsh	759.87	16.18	72.20	137.31	252.9	383.30	0	1621.74	67.10
River and pool	9.20	206.93	7.39	19.29	5.61	7.75	0	256.17	10.60
Forest	24.79	5.45	86.11	14.14	44.27	62.22	0	236.96	9.81
Meadow	36.37	9.95	22.05	72.68	15.74	33.12	0	189.90	7.86
Paddy field	0	0	0	0	0	0	0	0	0
Dry farmland	22.70	0.04	1.85	3.41	32.65	51.38	0	112.03	4.64
Other types	0	0	0	0	0	0	0	0	0
Total	852.92	238.54	189.60	246.82	351.16	537.76	0	2416.80	
Proportion (%)	35.29	9.87	7.85	10.21	14.53	22.25	0	100	
Variation rate (%)	−47.41	−6.88	−19.99	+29.97	/	+380.01	0		

Table 2: Transformation matrix of landscape and land use within the study area during the period from 1989 to 2006 (unit: km^2).

1989	2006								
	Marsh	River and pool	Forest	Meadow	Paddy field	Dry farmland	Other types	Total	Proportion (%)
Marsh	118.68	12.61	23.89	99.06	143.93	453.49	1.26	852.92	35.29
River and pool	22.83	86.06	8.17	45.62	9.96	64.85	1.05	238.54	9.87
Forest	6.05	10.84	39.84	15.12	31.37	86.31	0.08	189.60	7.85
Meadow	13.34	6.74	11.04	61.37	61.14	93.02	0.17	246.82	10.21
Paddy field	0.87	1.06	6.37	14.98	233.82	93.13	0.93	351.16	14.53
Dry farmland	3.76	3.19	9.71	15.13	332.45	162.5	11.02	537.76	22.25
Other types	0	0	0	0	0	0	0	0	0
Total	165.53	120.50	99.02	251.28	812.67	953.30	14.50	2416.80	
Proportion (%)	6.85	4.99	4.10	10.40	33.63	39.44	0.60	100	
Variation rate (%)	−80.59	−49.48	−47.77	+18.07	+131.42	+77.27	/		

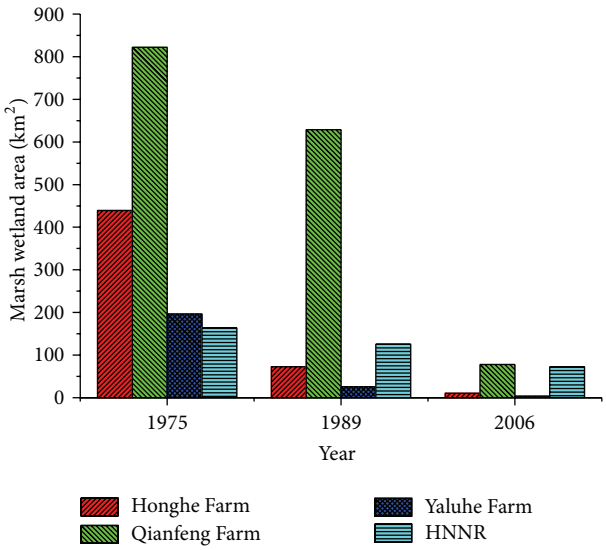

Figure 3: Loss of the marsh wetland of 4 units within the study area in the past 30 years.

Compared with the other three farms that have experienced extensive disturbances, the HNNR maintains a basic marsh landscape with less human disturbance. However, its marsh area decreased since the 1980s. Rapidly developing irrigation projects in the surrounding farms cut the water sources to the marsh ecosystem in the HNNR. Therefore, 30% of the marsh wetlands in the HNNR degraded into meadow wetlands [43]. From this research, we can conclude that the establishment of this natural reserve protected the remaining marsh wetlands with the intensified regional agricultural development. Because of the limited reserve area of the HNNR, however, our further analysis showed that marsh wetland ecosystems in the HNNR have been indirectly influenced by the agricultural activity of the surrounding farms that have changed the landscape pattern of the HNNR.

3.4. Damage to the Natural Wetland Ecosystem Caused by Local Agricultural Development. Wetlands are well-known habitats of water fowl. The HNNR is a transfer location in East Asia for rare water fowl, such as *Grus japonica* and *Ciconia boyciana*, which have first-order protection status in China [14]. Over 23 species of *Grus japonica* and 400 species of *Ciconia boyciana* were listed in the study area in the early 1970s [44], but only three species of *Grus japonica* and five species of *Ciconia boyciana* were recorded during an uninterrupted observation period in the study area between 2003 and 2004. This represents a nearly 90% loss since the 1970s [36]. For most water fowl, the increasing farmlands and paddy fields cannot replace their natural habitat. The shrinking natural marsh wetlands have an obvious negative impact on the existence of these water fowl [2, 45].

Damage to the wetland habitat for wildlife and plants has also resulted in the loss of rare plant species. Over 50 wetland plant species are listed as endangered at the national level in the region. Both *Dysophylla yatabeana* and *D. fauriei* are now extinct, although they were very common species 30 years ago. The damage to natural habitats harms wetland plants from both biological and ecological perspectives [43]. *Carex lasiocarpa* is a representative species of the local marsh wetland ecosystem. It was recorded in the 1970s as a robust and large plant with an average height of 73.7 cm. However, its height has decreased to an average of 40.5 cm, 33.2 cm shorter than 30 years earlier, according to a field survey conducted between 2003 and 2004 [36]. Its average biomass decreased from 653 g/m^2 to 403 g/m^2 (a 30% decrease) in the past 50 years [46]. With regard to plant composition, the richness of the species of wetland plants has also decreased because of decrease in the quality of the wetland habitats. Currently, there is an average of 6.7 species per square meter, a reduction of one species compared with 30 years ago [47]. The biodiversity of the natural ecosystems has definitely been damaged in the region because of the large amount of marsh wetland degradation into meadows. Our future research will precisely assess the weakness of ecological function due to the agriculture development.

3.5. Discussion. Marsh wetlands were widespread on the Sanjiang Plain before the 1980s. The growing season is very short (only 4 months) on the Sanjiang Plain. Most of the area in this region is flooded year round because of the extremely cold and moist climate. The rough natural conditions result in few permanent residents in this region. Therefore, the Sanjiang Plain is well known as "The Big Wild" because of its unique natural marsh landscape [9]. With the increasing Chinese population, the country is seriously challenged by the increasing demand for grain. Grain production is a priority for the Chinese government. Therefore, a series of agricultural policies were made to encourage marsh reclamation and the expansion of farmlands for the purpose of agricultural development [13]. The Sanjiang Plain became the highest priority for reclamation because of its abundance of wild land [10]. Within our study area, the Qianfeng Farm was established in 1969. The Yaluhe Farm was established in 1977, and the Honghe Farm was established in 1980. The purpose of these established farms was to reclaim marsh wetlands. However, encroachment on the marsh wetlands was not excessive because of the lower productivity during that time. Local farmers were not willing to produce more grain because of socialist equalitarianism [48–50], and people were busy engaging in various political movements throughout China during that time.

The initial stage of Chinese reform and open policy occurred from 1978 to 1983. During that time, China implemented successful reform of socialistic economic institutions throughout its widespread countryside. Under the reformation rubric, some local farms on the Sanjiang Plain were selected by the central government for pilot projects of modern agricultural farming. The farmers achieved efficient grain production while continuing to reclaim marsh wetlands under reclamation leadership in Jianshanjiang, although this did not reach a climax of regional marsh reclamation [40, 50]. The progression of encroachment on marsh wetlands accelerated on the Qianfeng Farm because of the widespread

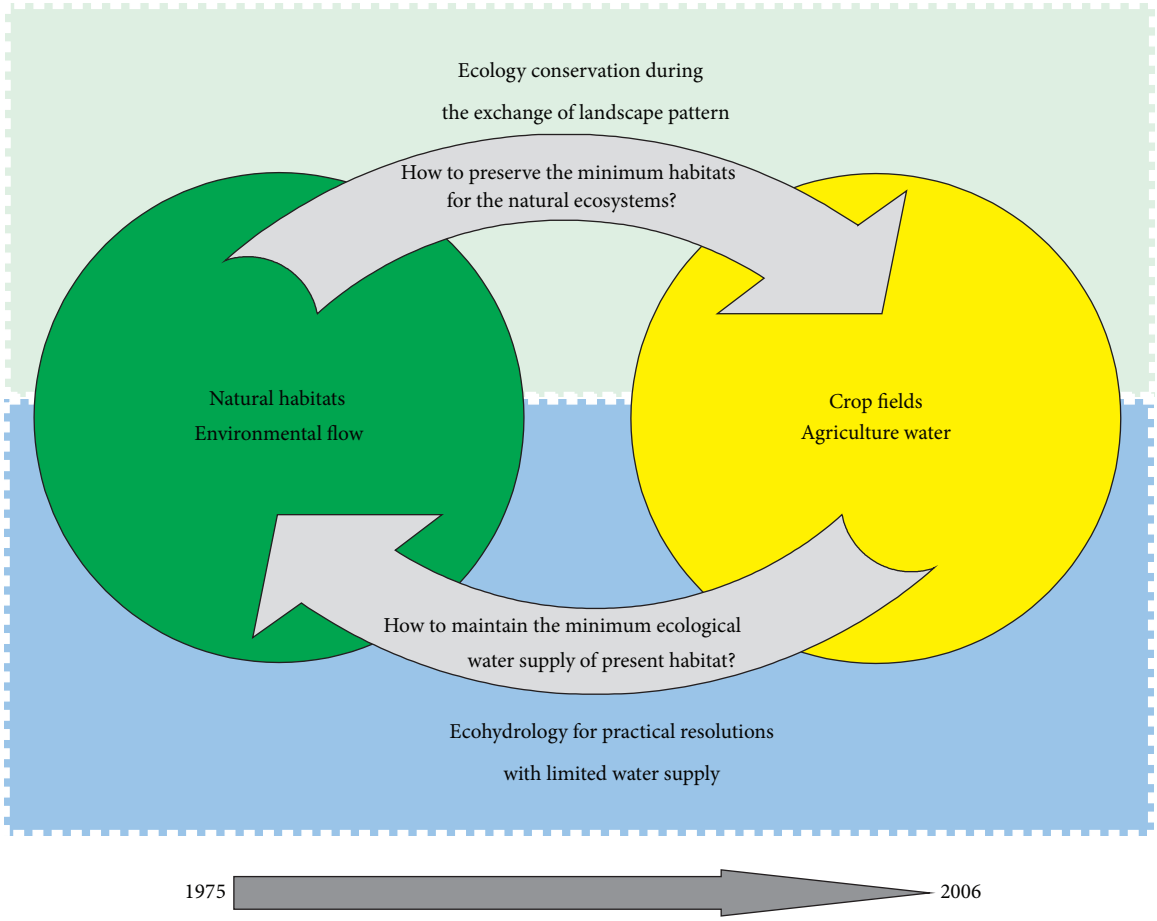

Figure 4: Two key issues in wetland eco-hydrology during the regional process of marsh wetland reclamation.

policy of organizing family farms encouraged by the parent body after 1985 [50]. Following the Qianfeng Farm, the Yaluhe Farm, which was previously a socialist institution, was divided into many small family farms in 1988, and this policy was followed by the Honghe Farm in 1993 [48]. With the new policy, farmers were actively involved in running their family farms. They made investments in various agricultural equipments to expand their own production capacities. Farming efficiency was improved so much that grain production increased during this period [13, 49] by somehow successfully reclaiming marsh wetlands to expand the farmland owned by the family. Encroachments on marsh wetlands most rapidly occurred on the Sanjiang Plain (Figure 3).

The Government of Heilongjiang province published the Regulation of Wetland Protection in Heilongjiang Province on June 20, 2003. It was the first regional regulation on wetland protection by a local government in China. The regulation declared the prohibition of all activities that encroach on wetlands [36]. However, number 1 document from the central government that encouraged an increase in the income of farmers at the national level was published in 2004. The document suggested subsidizing farmers by reducing their agricultural tax [9]. This policy stimulated the farmers' will to increase grain production. Local farmers attempted to reclaim the marsh wetlands to expand their farmland to maximize grain production, even through various illegal means that were against the Wetland Protection Regulation [16, 36]. The technical means of reclaiming marsh wetlands improved substantially during that period, and the modern agricultural facilities helped farmers reduce the cost of marsh reclamation [43]. Marsh reclamation also took disadvantage of both global warming and regional aridity [51]. The gradual illegal encroachment on the few remaining marsh wetlands has not been suspended in the study area, although the reclamation of marsh wetlands has been ceased on a large scale.

Marsh reclamation causes obvious negative impacts to wetland ecosystems. Wetlands, the natural habitats of most wildlife and plants, are well known as the "gene bank of wildlife." Wetlands have significant value for biodiversity in most ecosystems [52–54]. Extensive alterations of both regional hydrology and ecological patterns have occurred at a large scale on the Sanjiang Plain. Marsh reclamation has caused an irreversible and rapid change from a natural ecosystem to an agricultural ecosystem at the regional level. As a consequence of the change, irrigation water has replaced the previous natural environmental flow. However, little research has scientifically assessed the huge disturbance and

ecological impact [55, 56]. The challenges include resolving two key scientific issues at the regional level. The first issue is how to preserve a minimum of natural habitats during the rapid progression of ecosystem reductions. The second issue is how to maintain a minimum amount of environmental flow for the remaining natural ecosystems confronted by the increased demand of irrigation water (Figure 4).

4. Conclusions

(1) Intensified agriculture development has changed a unique natural marsh landscape into an agricultural landscape during the past 30 years in the study area. The reclamation process of marsh wetlands accelerated in response to various national policies that demanded grain production beginning in the 1980s. Ninety percent of the natural marsh wetland area was lost in the study area from 1975 to 2006 while most dry farmland has been replaced by paddy fields.

(2) Attempt of current Chinese institution for preserving the regional natural wetlands has achieved limited success. A few wetlands remain healthy because of the establishment of the HNNR, although their ecological quality has declined because of increased threats to the remaining wetland habitats. Irrigation expansion plays a key role in such threats.

(3) The sustainability of the natural wetland ecosystems is being threatened by continuous reduction in the wetland habitats number and decline in the ecological quality due to the intensified agriculture development. In the future, it is a big challenge to preserve a minimum of natural habitats during the rapid progression of natural ecosystem reductions while natural resource administrators attempt to maintain a reasonable amount of environmental flow for the remaining natural ecosystems confronted by the increased demand of irrigation water.

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Research Article

Polder Effects on Sediment-to-Soil Conversion: Water Table, Residual Available Water Capacity, and Salt Stress Interdependence

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The French Atlantic marshlands, reclaimed since the Middle Age, have been successively used for extensive grazing and more recently for cereal cultivation from 1970. The soils have acquired specific properties which have been induced by the successive reclaiming and drainage works and by the response of the clay dominant primary sediments, that is, structure, moisture, and salinity profiles. Based on the whole survey of the Marais Poitevin and Marais de Rochefort and in order to explain the mechanisms of marsh soil behavior, the work focuses on two typical spots: an undrained grassland since at least 1964 and a drained cereal cultivated field. The structure-hydromechanical profiles relationships have been established thanks to the clay matrix shrinkage curve. They are confronted to the hydraulic functioning including the fresh-to-salt water transfers and to the recording of tensiometer profiles. The $CE_{1/5}$ profiles supply the water geochemical and geophysical data by their better accuracy. Associated to the available water capacity calculation they allow the representation of the parallel evolution of the residual available water capacity profiles and salinity profiles according to the plant growing and rooting from the mesophile systems of grassland to the hygrophile systems of drained fields.

1. Introduction

The degradation and restoration of wetlands are consequences and/or objectives of the successive human activities which are, or which have been, governed by the society needs. The coastal marshes have often been the result of land reclaiming from the sea by successive polders. The initial attempt was to earn new areas, generally fit for farming from primary muddy sediments. The main consequences due to the polder mechanisms are to provoke the "primary" muddy sediment-to-soil evolution by desiccation, consolidation, and maturation of the clay-rich material. The final objective was the development of "dry" territories. The mechanism is generally associated to the descending of the water table levels. In case of too slow desiccation rates, the phenomenon has been emphasized by drainage. On the hydraulically point of view, the need of restoration is the consequence of the too large extension of the "dry marshes" in place of the "wet marshes" which constitute the residual and "primitive" domains of the flora and wildlife. In these conditions the control of the water table levels can be a cause of fighting due to the different objectives and economical or ecological interests.

One of the objectives prevailing during the years 1970–1980 was the conversion of the grazing system to the cereal cultivation system. Unfortunately the too high water table levels limit the rooting depth for cereals. Consequently, the drainage was systematically developed during these years [1]. Besides, a consequence of the cereal system development was the impact of the fertilizers and phytosanitaries on the coastal oyster and mussel cultures [2]. Thus the cultivation system approach has to consider the exploitation-environment relationship since the 1970s [3]. In fact, the functioning of the coastal marsh soils is mainly based on their salinity and

sodicity evolutions. These two factors depend on the water table levels which are governed by the successive polders since the Middle Ages and more recently by the drainage. However, the drainage efficiency may be limited because of the soil structural stability and low permeability in such clay dominant soils [1, 4–6]. Due to this problem, successive typologies of soil exploitation with and without drainage were proposed for these territories: that is, stable, intermediate, and unstable "dry marshes" [7]. Finally, Pons and Gerbaud propose in 2005 [6] a soil classification based on the salinity, sodicity, and a dispersion index measurement derived from the Emerson method [6, 8].

Eventually, the two main problems induced by the land reclamation on sea wetlands are the drastic structure evolution of the clay dominant sediments due to the desiccation and shrinkage phenomena and the induced soil salinity and sodicity evolutions. The consequences are the superimposition of the water and salt stresses due to the plant growing. The present paper is based on works made on the French Atlantic coast wetlands many years ago [9–15]. The successive works focused on the relationships which can exist between the microstructure evolution of clay matrices and the macroscopic properties of the clay dominant soils. The coastal territories were chosen thanks to their textural and mineralogical homogeneity of soils and sediments and thanks to the large water content domain available along each vertical profile. The primary investigations were driven in order to ensure the homogeneity of the shallow sedimentary formations throughout the marshlands of the Marais Poitevin and Marais de Rochefort. Secondly, the investigations were concentrated on the INRA experimental site of St Laurent de la Prée for all the calibrations of soil structure-hydromechanical properties relationships [13, 15] (Figures 1 and 2). Thirdly the goal was to understand the role of the soil structure evolution on the plant growing and crop yield. The work is based on the study of the soil profiles developed on drained cereal systems and on undrained grasslands. It focuses on the explanation of the soil behavior mechanisms due to the desiccation and flora-soil interaction phenomena and thanks to the fluviomarine origin of the sediments, on the water to salt stress behavior.

2. Material and Methods

2.1. Material and Geological Setting. The results presented in this work were obtained from the wetlands located along the Atlantic coast of France. Two areas were studied: the "Marais Poitevin" and the "Marais de Rochefort." In the two wetlands, the soils are formed on clay dominant sediments named "bri" which result from the silting of an erosion basin in Jurassic limestones during the Flandrian transgression. The ages of the sediments range from 8,000 years to nowadays. In the "Marais Poitevin," they were deposited directly on granitic rocks on the north part of the basin (Vendée department) and on Jurassic limestones in the south part of the basin (Deux Sèvres and Charente Maritime departments). In the "Marais de Rochefort" they were deposited on the Jurassic limestones. The sediments have fluviomarine origin and were deposited up to +3 m asl which was the maximum level reached by the sea during this period. The origin of the sediments allows the distinction between the marine bri, deposited by sea water, and the continental bri stemming from the peripheral hills. The continental sediments (continental bri) are enriched in organic matter and frequently include peat layers.

The marshes have emerged and dried since at least the Xth century following the sea regression. The polder works started during the XIth century and were the most effective during the XVIIth century. The polder reclaiming was led in order to isolate the marshlands from the sea water inlet along the Atlantic coast and from the continental fresh water inlet along the peripheral hills. Finally the basin was divided into three characteristic territories:

(i) the "mizottes" which are salt territories lining the Atlantic coast and which are recovered by the sea during high tide,

(ii) the "dry marsh" constituted by the oldest polders which are isolated from the "mizottes" and from the continental water inlet by series of embankments,

(iii) the "wet marsh" located along the periphery of the basin whose role is the storage of rain and fresh water flowing from the peripheral hills.

The climate is coastal-oceanic with a mean annual rainfall of 780 mm distributed throughout two marked seasons: 52% of the rainfall occurs between October and January. The mean June–August temperature and potential evapotranspiration (ETP) from 1971 to 2001 were 16.3°C and 321 mm, respectively [16].

The studies particularly focused on the soils formed on bri in the "dry" and "wet" marshes, excluding the continental organic bri and peat-rich formations. In these "dry" and "wet" marshes the bri shows quite homogeneous mineralogy and texture [9, 14]:

(i) clay to silty clay texture with 85 to 95% of particles lower than 20 μm and 40 to 60% of particles lower than 2 μm,

(ii) small organic matter content, 0.4 to 2.4 weight %,

(iii) dominant illite, plus kaolinite and illite/smectite mixed layers, and small amount of pure smectite assemblages,

(iv) the measured CEC accord to the illite domain (20–30 meq/100 g),

(v) the shrinkage, plasticity, and liquidity limits are 20%, 40%, and 70% in gravimetric water content, respectively.

In the 0-to-20 cm soil surface and in the recognized paleosoils the infra 2 μm fraction decreases from the 40–60% of the primary sediment to 5–23%. In parallel the illite/smectite mixed layer assemblages present a weak increase of the smectitic layers % in depth [9, 17]. This increase of smectitic layer % was identified by X-ray diffraction only on the 0.2 μm fraction. Similar mineralogical evolution was identified according to the ages of soils sampled in the successive

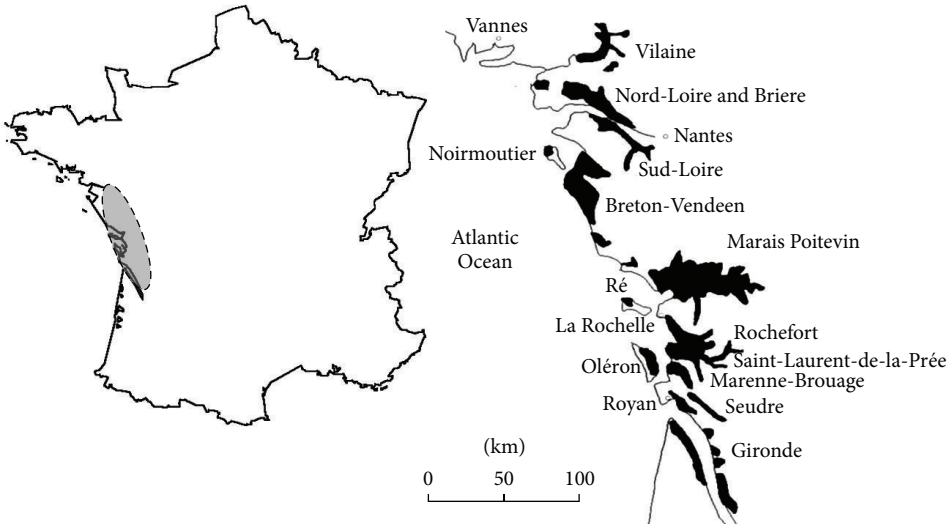

Figure 1: Location of the "Marais de Rochefort" and Marais Poitevin on the West Atlantic coast of France. The INRA experimental site of St Laurent de la Prée is located in the "Marais de Rochefort."

polders [18]. The phenomenon was attributed to a smectite-to-illite conversion associated to the time of soil maturation.

The general structure of the soils and/or sediment-to-soil evolution was shown by Bernard [9], Bernard et al. [10], and Dudoignon et al. [12]. The authors demonstrated the role of the shrinkage, plasticity, and liquidity limits (W_s, W_p, and W_l, resp.) on the soil structure behavior. In fact the gravimetric water content (W) profiles exhibit a progressive W increase from the surface to the depth. They can be divided in two superimposed "layers" (Figure 2):

(i) a W_s-W_p surface layer developed down to a level equivalent to W_p. The W profiles evolve according to the climatic changes and associated wetting-desiccation cycles of surface. The W_p level remains quite invariant through the seasons,

(ii) a W_p-W_l subjacent layer virtually insensitive to the seasonal effects and surface desiccation.

This vertical superimposition operates hydraulically as two superimposed media isolated by the W_p layer. The surface layer is a double hydraulic conductivity medium: that is, small permeability of the clay matrix constituting the inner part of prisms and peds (micro-to-centimeter scale) and large permeability of the shrinkage fracture network (macroscopic scale). The subjacent layer is characterized by only the small permeability of the clay matrix which is governed by the microstructure and the particle arrangements in the W_p-W_l saturated domain: about 10^{-6} m·s^{-1} in W_l state down to 10^{-10} m·s^{-1} in the W_p state. The W_p layer works as an impermeable line preventing all water transfers from the surface layer down to the subjacent one. The first consequent question is as follows: is the two-layer structure responsible of two superimposed water tables: a surface and free water table fed by the rainwater and a subjacent captive water table resulting from the initial capture of seawater during the sedimentation? In these conditions the water table level

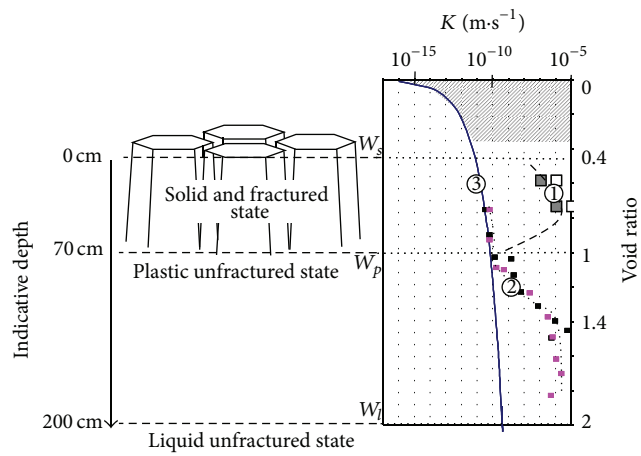

Figure 2: Parallel representation of the vertical evolution of the soil structure and hydraulic conductivity. In the W_s-W_p surface layer the clay dominant soil evolves in its solid state. In the W_p-W_l subjacent layer the clay matrix evolves in plastic-to-liquid states. The hydraulic conductivity has been "*in situ*" measured by infiltrometry in the fractured surface layer (large squares), via oedometer compressibility tests on saturated and consolidated clay material (small squares), and calculated taking into account the clay matrix microstructure parameters (continuous line; [18]). The dashed domain does not exist "*in situ*" (from Gallier et al. [15]).

descending should involve the descending of the "impermeable" W_p level and the thickness of the surface layer due to eventual leaching by the rain water. The second question is as follows: can there be a vertical fresh-to-sea water exchange from the surface to the subjacent layer? In such conditions the plant growing and the pedodiversity would be mainly governed by the water table level via the eventual addition of a salt stress on the water stress.

The geological structure of the Marais Poitevin basin was studied by geoelectrical prospecting. The resistivity sections

were recorded on twenty areas distributed over the whole basin, from the thin peripheral bri deposits of the "wet" marshes to the inner areas of the "dry" marshes where the bri deposits can reach 30 meter thick. The resistivity sections were obtained using a Syscal R1+ resistivimeter interfaced with a 48 electrodes switch. From the loading to the inverse calculation of the apparent resistivity sections the ELECTRE II, PROSYS, and RES2DINV software were used, respectively. The length of the device around 240 m allows investigations down to 30 meter depth. In these conditions the resistivity sections propose good images of the geological structure with (1) the bri/limestone interface and (2) the typical vertical structure of the bri which can be observed on the whole territory. From the surface down to the limestone contact the bri is characterized by a progressive evolution of its resistivity according to the W_p-to-W_l structure evolution and by the consolidation in depth (Figure 3). On the contrary, such long devices miss the subsurface layers (0-to-1 m depth). The resistivity sections show also a horizontally evolution of the bri resistivity from the limestone contact to the "dry" marsh. For equivalent structures the bri is characterized by higher values of resistivity along the limestone contact. It is characteristic of lower water salinities along the peripheral limestone which are indicative of the peripheral fresh water inlet through the bri in W_l "liquid" state.

The 0-to-2 m depth layer was prospected by coupling the resistivity measurement using a salinometer and auger drill holes. The method allowed the measurement of real resistivity associated to measured depth and the calibration of the Archie's law for the clay dominant material constituting the bri [11]:

$$\rho s = 1.01 \rho f \phi^{-2.73} \mathrm{Sat}^{-2}, \quad (1)$$

with ρs is the soil resistivity, 1.01 the formation factor α, ρf the water resistivity directly dependent of its salinity, ϕ the porosity, −2.73 the cementation factor m, Sat the saturation index, and −2 the n factor characteristic of the medium.

In order to take into account the clay mineral surface conductivity, the soil resistivity was calculated following the Waxmann and Smits equation [19]. The calculations made with an average 25 meq/100 g CEC and fluid conductivities measured on water sampled in piezometers (2 S·m^{-1} and 4 S·m^{-1} in corn field and grassland, resp.) give results equivalent to the resistivities calculated following the simple Archie's law [15]. In these salt media of coast marshes, the high fluid conductivities minimize the clay mineral surface effect.

After a large investigation through the Marais Poitevin territories the study focused on the INRA experimental site of the Saint Laurent-de-la-Prée in the Marais de Rochefort. The main objective is to quantify the role of the water table level variations and rain water-salt water transfer on the plant growing and yields. The studied plots have been distributed on undrained grasslands (since at least 1962) and drained cultivated parcels. The cultivated parcels are lean against the surrounding limestone hill which is the main source of fresh water inlet in the clay-dominant bri system (Figure 4).

The study of the soil structure-hydromechanical property relationships was performed by coupling the water profiles with cone resistance and shear stress profiles [14, 15]. Finally authors used the shrinkage curve of the clay dominant soil to represent the results in successive crossed diagrams [9–15].

2.2. Methods. The subsurface soil structure is governed by the desiccation/shrinkage and rehydration/swelling cycles which operate during the seasons. The shrinkage is a 3D mechanism which induces the clay matrix compaction and the opening of a fracture network according to the descending of the desiccation front (Figures 2 and 3). The role of the W_p level was demonstrated by Bernard [9] from the study of the soil structure profiles. The author observed the stop of the shrinkage fractures at the W_p depth and measured the very small hydraulic conductivity of the clay matrix by the oedometer compressibility test.

Two types of *in situ* investigations focused on the eventual water table superimposition and salt-to-fresh geochemical nature of the underground water. The first investigation method was the geoelectrical prospecting. Based on the Archie's law, the apparent soil resistivity gives indications on the salinity of water. The second investigation consists in drilling couple of piezometers in each spot. Each couple of piezometer includes one short piezometer drilled down to 1.50–2.00 meter depth and one long piezometer drilled down to 3 or 6 meter depths. They have been drilled in order to follow the levels of the eventual two water tables and in order to sample the associated groundwaters for chemical analyses. The chemical analyses have been performed on pumped water in each piezometer. They were made by ionic chromatography in liquid phases for Cl^- and SO_4^{2-} anions and by I C P for Ca^{++}, Na^+, K^+,and Mg^{++} cations (Table 2). Unfortunately, the chemical analyses give global and average chemical compositions of the pumped water which are too inaccurate due to an eventual vertical salinity gradient. In these conditions, the soil profiles were characterized by electrical conductivity ($CE_{1/5}$) measurements. The measurements were made on soils sampled from the surface to 2.00 m depth every 10 cm, according to the clay auger sampling for the water content measurements. The $CE_{1/5}$ was measured on the 1/5 extracts (10 g of dried soil in 50 g of distilled water) [6]. In fact, the $CE_{1/5}$ measurements are made from 10 g of dried soil; thus the $CE_{1/5}$ is independent from the soil water content. Nevertheless it is possible to calculate the fluid conductivity (CEf) for each soil sample following the Montoroi [20] formulae, and Montoroi's equation:

$$\mathrm{CEf} = \mathrm{CE}_{1/5}\,(5W)\,, \quad (2)$$

with CEf = the "*in situ*" fluid conductivity, $CE_{1/5}$ the soil conductivity measured in laboratory, and W the gravimetric water content.

The study of the soil structure-crop yield relationship was based on the comparison between the $CE_{1/5}$ profiles, the crop yield components, tensiometer profiles recording, and available water capacity (RU) calculations. The yield components included the foot plant number/m^2, ears/m^2, depth of rooting, weight of thousand grains, and final yield. Only the final yield in 10^2 kg/ha is used in the paper. The

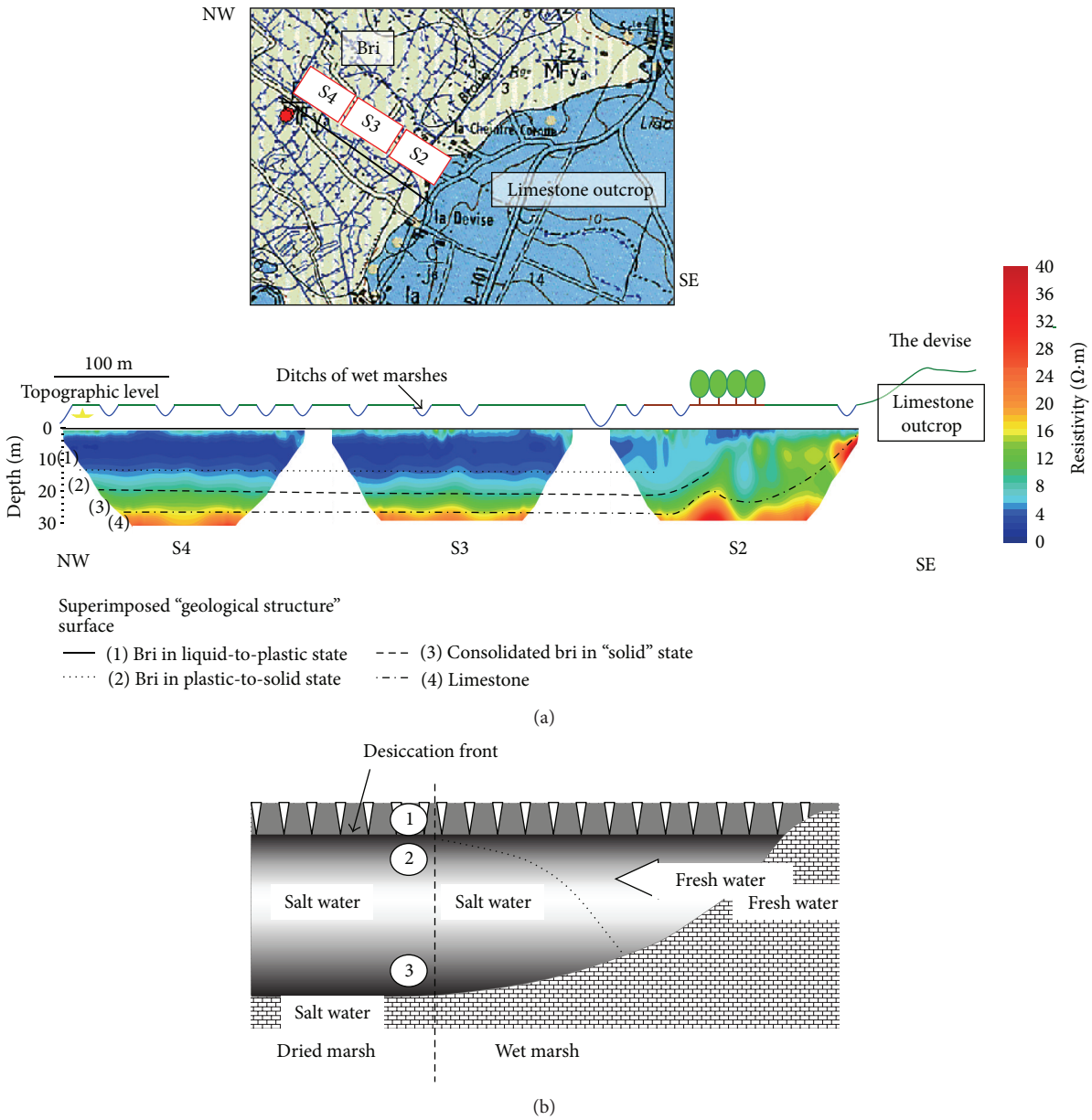

Figure 3: (a) Typical structure revealed by the long resistivity sections in geoelectrical prospecting, (b) schematic representation of the bri structure from the peripheral limestone contact to the inner "dry marsh." (1) Surface layer impacted by the desiccation-shrinkage effects, (2) W_p to W_l evolution due to the reclaiming and desiccation effect, and (3) W_l to W_p consolidation in depth.

different yield components were counted in the K1a-b, K2a-b, and K3a-b spots in 2006 and 2007. They were counted in K1A-B, K2A-B, K3A-B, and K3C-D spots from 2008 to 2011. Each spot is 4.5 m^2 area which corresponds, for example, to 2 contiguous 3 m length rows of corn. In addition the eight K-spots were equipped with tensiometers at the depths of 30, 60, and 90 cm in cultivated field and 40, 60, and 80 cm in the grassland. The suction pressures were measured in 2011 and 2012 in grassland and in a field alternatively farmed for wheat (2011) and corn (2012). The tensiometers are SMS 2500S type. They are limited to 850 mb pressure equivalent to 2.8 pF. Nevertheless they allow the monitoring of the desiccation profiles during the seasons and associated descending rooting.

3. Results

3.1. Water Table Levels and Water Chemistry. All the resistivity sections obtained by long devices across the marshes of the Marais Poitevin and Marais de Rochefort accord to the different drill holes made in these territories to identify the same vertical geological structure [9, 21]:

(i) bri in solid-to-plastic state in surface,

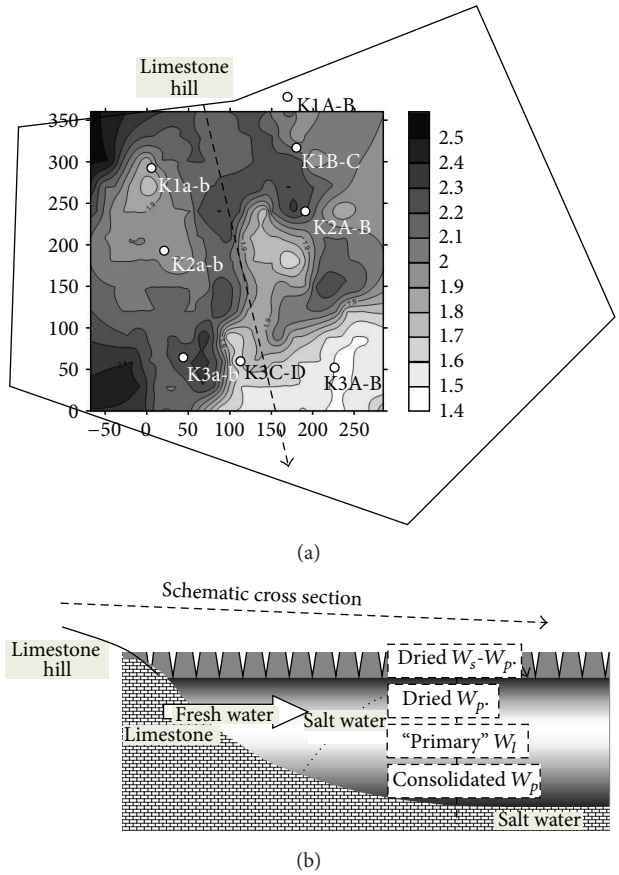

Figure 4: (a) Topographic map (2.5 m altitude near the limestone hill to 1.4 m altitude on K3A-B plot) of the cultivated field with location of the monitoring plots K1 to K3. (b) Schematic cross-section. W_s = shrinkage limit; W_p = plasticity limit; W_l = liquidity limit. Dried = W_l-to-W_s behavior due to the surface desiccation; "primary" = initial liquid state; consolidated = W_l-to-W_p behavior due to the earth weight.

(ii) bri in liquid state approximately between 2 and 5 meter depth,

(iii) consolidated bri in plastic-to-solid state in depth and often evolving to marls,

(iv) subjacent limestone.

Because of the consolidated bri or marl layer directly superimposed on the limestone no significant water transfer was measured between the two formations [21]. Nevertheless the pumped water near the interface presents chemical compositions characteristic of sea water plus fresh water mixture. In fact, from isotopic data, Anongba [21] dated the fresh plus sea water mixture as a phenomenon synchronous of the progressive marine transgressions. The mechanism of fresh water inlet still operates nowadays along the peripheral limestone-bri contact (Figure 3).

The vertical fresh water transfers from the fractured bri in solid state of surface, down to the subjacent plastic-to-liquid bri which is initially saturated by salt water, are confronted to the very low permeability of the intermediate W_p level (Figure 2). Nevertheless the monitoring of the water table levels in the couple of piezometers drilled in K1A-B, K2A-B, and K3A-B show only one continuous water table in the bri (Figure 5). A difference between the water table levels measured in a couple of piezometers was only observed in K1A-B during the corn growing. It can be effectively explained by a larger permeability in depth.

The waters pumped in the double piezometer have been analyzed in January and February 2009. The chemical compositions are characteristic of salt plus fresh water mixtures (Figure 6; Table 1). The low salinities have been measured in the short piezometers (2 m depth), and the high salinities have been measured in the long piezometers (3–6 m depth).

3.2. $CE_{1/5}$ Profiles. The calibration of the Archie's law in these sediments allows the calculation and imaging of the water salinity. Nevertheless the measured resistivity sections are always composed of apparent resistivity and apparent depth. The second "defect" specific to the long geo-electrical device is the lack of data in the 50-to-70 cm surface layer. If the method allows realistic basement recognition, it does not give sufficiently accurate data on the soil sodicity and nape salinity in the 0- to -2 m depth. This lack of data was supplied by the $CE_{1/5}$ profile measurements. This $CE_{1/5}$ profile evolution is well showed in the undrained grassland (Figure 7). For horizontal water tables the profiles are governed by the surface topography and by the channel proximity: from the surface down to 1.50 m depth the ranges of $CE_{1/5}$ evolution are 1–7, 8–20, and 10–30 mS·m^{-1} for the locations close by the channel (5 m), intermediate (10 m), and far-off (22 m), respectively. The three profiles correspond to mesophile, mesohygrophile, and hygrophile systems, respectively [5].

Similar evolutions have been measured in the cultivated and drained parcel (Figure 8). The $CE_{1/5}$ profiles show progressive increase of $CE_{1/5}$ with depth nearby the limestone hill. The profile patterns evolve with the distance from the limestone hill toward "two slope patterns": that is, a quite constant small conductivity from the surface to the high water table level and a sharp $CE_{1/5}$ increase in depth. The "two slope pattern" of the $CE_{1/5}$ profiles is enhanced by the gradually descending of the water table as the distance from the hill increases. The $CE_{1/5}$ profiles evolve according to the depth of the water table and consequently according to the thickness of soil surface open to the leaching by rain water (Figures 7 and 8). The $CE_{1/5}$ profiles are also governed by the eventual fresh water inlet from the peripheral limestone and/or from nearby water channels.

The $CE_{1/5}$ profiles are quite invariant through the seasons. They can be used as references for the calculation of the water salinity profiles. They are deduced from the fluid conductivity (CEf) calculated from the $CE_{1/5}$ and Wprofiles using the Montoroi [20] formulae (Figure 9).

3.3. Soil-Plant Interaction. The evolutions of the crop yields on the different parcels are impacted by the prevailing water conditions and the $CE_{1/5}$ profiles. The yields have been often estimated from the depth of the sodic zone [6]. This depth of the sodic zone can be easy to be determined from the

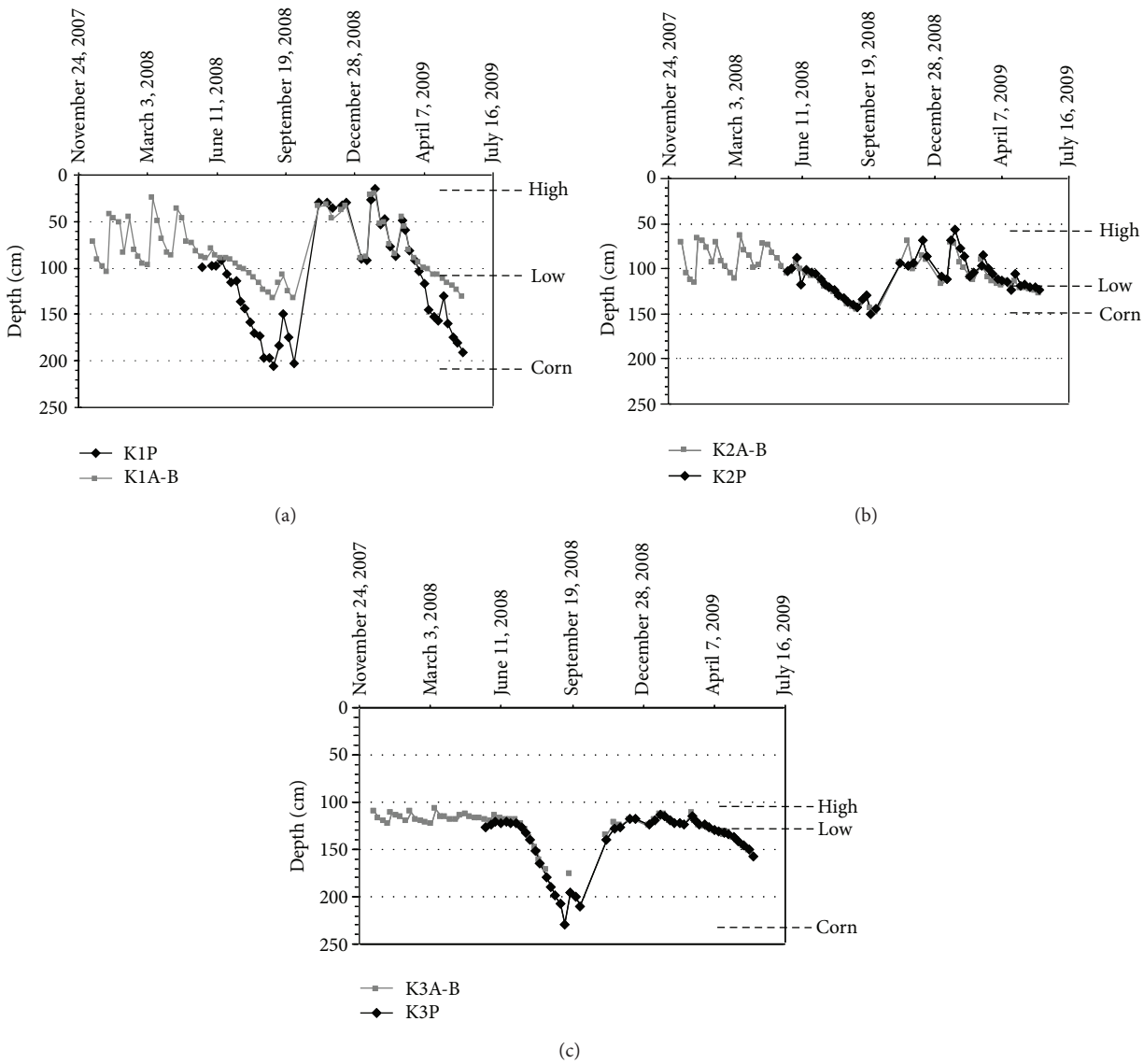

Figure 5: Monitoring of the water table levels in the short piezometers K1A-B (2 m), K2A-B (2 m), and K3A-B (2 m) and in the long ones K1P (3 m), K2P (6 m), and K3P (6 m) from 24 October 2007 to 16 July 2008. High and low = natural water table variation; corn = descending of the water table due to the corn water consumption.

"two slope" patterns $CE_{1/5}$ profiles similar to the K3 profiles (Figure 8). On the contrary, it becomes difficult to determine a depth of sodic zone in progressive $CE_{1/5}$ profiles similar to the K1 and K2 profiles (Figure 8). In fact the $CE_{1/5}$ profiles exhibit two superimposed patterns:

(i) from the surface to the top of the salt water table, the $CE_{1/5}$ values increase following an exponential way

(ii) and from the top of the salt nape to the depth, the $CE_{1/5}$ values show a low increase or remain almost constant.

Whatever are the $CE_{1/5}$ profiles, their upper patterns can be calculated following a simple logarithmic function (Figure 10):

$$D = a * \ln\left(CE_{1/5}\right) + b, \quad (3)$$

with D the depth in m, $CE_{1/5}$ the electrical conductivity in mS/cm^2, and a and b two parameters fit to obtain the good superimposition of the calculated $CE_{1/5}$ pattern on the measured $CE_{1/5}$ pattern [14]. Reciprocally the $CE_{1/5}$ profiles can be calculated as follows:

$$CE = \exp\left(\frac{D-b}{a}\right). \quad (4)$$

The shape of the $CE_{1/5}$ patterns is governed by the water table level fluctuations during the seasonal cycles. The deep descending of the water table during the dry season provokes the descending of the desiccation front. The consequences are the leaching of the unsaturated surface layer by the rainfall and a possible phenomenon of capillary ascent. Finally the

Table 1: Chemical compositions (in mg/L) of the waters pumped in January and February 2009 in the piezometers K1A-B (2.00 and 3.00 m), K2A-B (2.00 and 6.00 m), K3A-B (2.00 and 3.00 m), and K3C-D (2.00).

	Cl^-	Na	Ca	K	Mg	S	SO_4	Total salinity (mg/L)
				January piezometer				
K1A-B (3.0 m)	4080	2508	361.1	116.6	339.4	756	2269	9673.9
K2A-B (6.0 m)	8370	4438	247.5	208.4	355.5	506	1517	15136.5
K3A-B (3.0 m)	4960	3059	301.8	154.8	513.2	1063	3189	12178.2
K1A-B (2.0 m)	1640	1302	330	61.16	199.1	427	1282	4814.16
K2A-B (2.0 m)	430	390	190.5	24.02	66.09	191	573	1673.51
K3A-B (2.0 m)	1980	1760	580.8	97.51	253.5	790	2369	7040.71
K3C-D (2.0 m)	970	1098	594.4	61.2	144.1	527	1580	4447.8
				February piezometer				
K1A-B (3.0 m)	2207	1797	292.5	86.59	242.3	543.2	1629.6	6254.99
K2A-B (6.0 m)	7201	4203	230.8	193.7	467.7	455	1365	13661.2
K3A-B (3.0 m)	3578	2643	435.9	128.8	431.4	821	2463	9680.1
K1A-B (2.0 m)	825	683	263.1	38.84	131.3	303.7	911.1	2852.34
K2A-B (2.0 m)	631	514.6	218.7	23.48	76.93	197.5	592.5	2075.21
K3A-B (2.0 m)	2661	2057	448.1	107.7	356.8	837.3	2511.9	8142.5
K3C-D (2.0 m)	834	1068	265.3	56.48	139.2	530.2	1590.6	3953.58

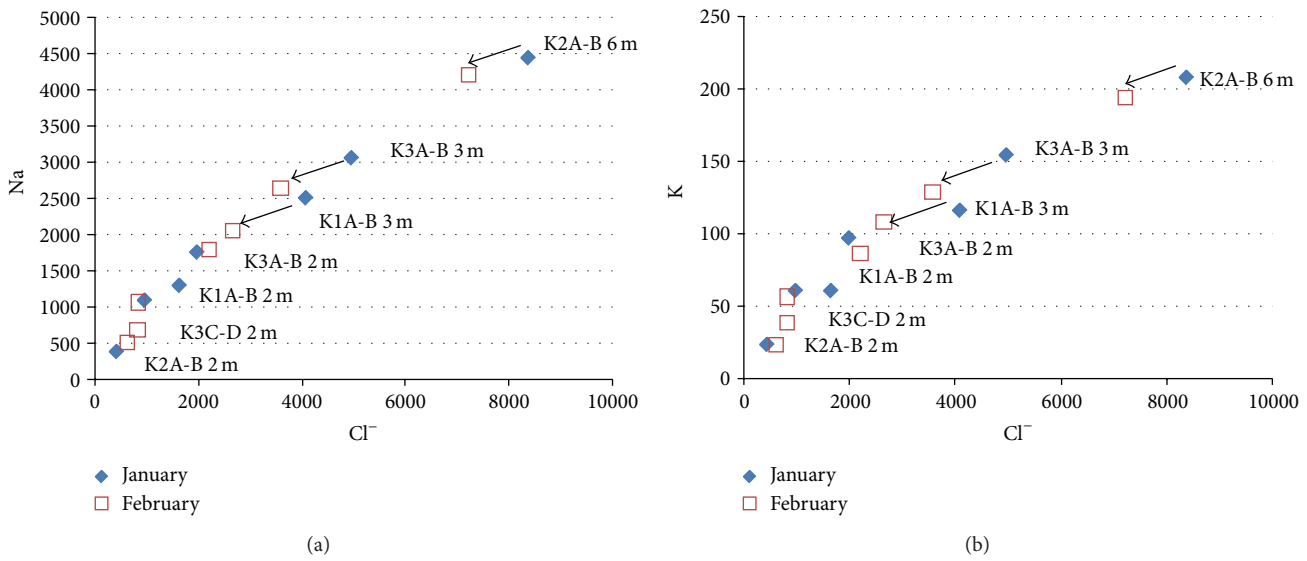

Figure 6: Na and K contents versus Cl^- contents (in mg/L) for the different chemical compositions of the waters pumped in January and February 2009 in the piezometers K1A-B (2.00 and 3.00 m), K2A-B (2.00 and 6.00 m), K3A-B (2.00 and 3.00 m), and K3C-D (2.00).

calculated $CE_{1/5}$ profiles have been related to the corn yields for 2006, 2008, and 2010 [14] (Table 2; Figure 10).

The descending of the desiccation front is caused by the summer climatic conditions enhanced by the water consumption by the plants. The interaction between the water profiles and the plant growing can be suggested by the evolution of the tensiometer profiles (Figures 11 and 12).

In the grassland the progression of the desiccation front is well shown by the "increase" of the measured suction pressures and by the time offset between the 40 cm and 60 cm measured pressures. The low suction pressures measured at the depth of 80 cm are due to the near water table level in this undrained field. Nevertheless, the d40 and d60 plug disconnections operate at different gravimetric water contents and thus at different suction pressures. In these conditions the plug disconnection cannot be explained by a simple difference between the gravimetric water contents.

In the drained wheat field a similar time offset is observed between the suction pressures measured at the depths of 60 and 90 cm. The d60 and d90 plug disconnections operate also at different water contents. The mechanism is similar to the mechanism observed in the grassland. Two main differences emerge: the deepening of the desiccation front in this drained field and the behavior of the tensiometer measurements near the surface (30 cm). The surface desiccation does not imply any increase of the measured suction pressures at the depth

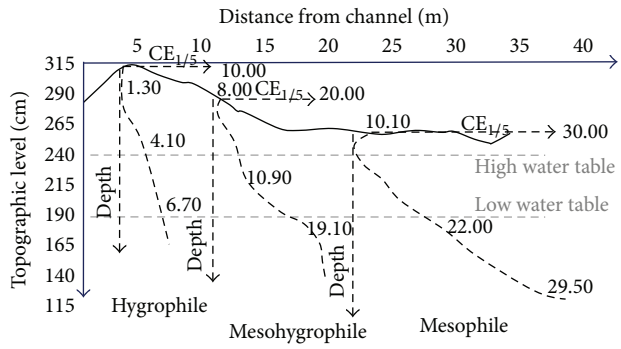

Figure 7: Evolution of the $CE_{1/5}$ profiles according to the topographic level and thickness of leached soil surface (undrained grassland). The mound is due to sediment and soil deposited during the successive channel cleaning works. Low and high water tables indicate the domain of water table variation during the seasons.

Table 2: Culture yields in 10^2 kg/ha measured in 2006, 2008, and 2010 for corn field and 2007, 2009, and 2011 for wheat field.

	Corn			Wheat		
	2006	2008	2010	2007	2009	2011
K1 a-b	104.4			56.1		
K2 a-b	109.4			67.2		
K3 a-b	143.3			67.1		
K1 A-B		126.9	105.1		92.1	78
K2 A-B		132.8	111		103.8	72
K3 A-B		157.5	117		99.6	97
K3 C-D		147.3	111.6		108.3	86

of 30 cm. In fact the 0–30 cm surface layer is affected by the tillage every year. The consequence is the formation of a "crumble" soil structure in surface characterized by a large unsaturated meso- to macroporosity. At variance the soil structure in depth, unaffected by the tillage, evolves according only to the clay matrix microstructure along its shrinkage curve.

Using the $CE_{1/5}$ profile of each spot as reference, the water salinity may be calculated for the successive depth using the couple water content-$CE_{1/5}$ following the Monteroi equation. For each depth the suction pressure accords to the water content. In these conditions the clay matrix shrinkage curve can be used as a tool to represent the parallel evolution of the water content, tensiometer pressure, and water salinity (Figure 13).

3.4. Available Water Capacity Behavior. The two main parameters which act on the plant growing and flora diversification are the water and salt stresses. The high water salinities limit the plant growing to the mesophile systems: the moderate water consumption limits the salt "storage" in the plant organism. On the contrary the poor water salinity allows the growing of hygrophile plants: despite the high water consumption the salt "storage" remains limited in the plant organism. In these coastal marshlands the two water and salt stresses are governed by the mechanisms of "wet land" to "dried land" conversion:

(i) the water table descending caused by the hydraulic works and associated surface desiccation,

(ii) the soil structure behavior due to the shrinkage properties of the clay dominant material,

(iii) the eventual fresh water inlet from channel of peripheral limestone hill,

(iv) and, consequently, the progressive evolution from the mesophile systems to mesohygrophile and hygrophile ones.

Considering the water table level, the plant growing and the root deepening are confronted firstly to the available water capacity (RU) of the unsaturated soil and secondly to the level of anaerobic in the subjacent saturated medium. The high water table level prevailing during the wet winter season stops the plant rooting. It is the main problem for the yields of the winter cultivation as wheat. The quite linear 1/1 water table depth/root depth ratio was demonstrated in these marshes by Pons et al. (2000) [1]. The problem of rooting depth has been in part resolved in the 70–80 years by the extensive politic of drainage. On the salinity point of view, the $CE_{1/5}$ profiles indicate low salinities near the surface and high salinities in depth. The problem becomes evident for the yields of the summer cultivation as corn. Due to the drastic decrease of the available water capacity of soil in surface the plant rooting has to progress in depth and reaches the salt levels.

The available water capacity is usually calculated according to the soil texture and/or according to the rainfall-evapotranspiration balances [22]. The available water capacity (RU) can be calculated as follows:

$$\mathrm{RU} = h \times d\left(W_{\mathrm{fc}} - W_{\mathrm{wp}}\right) \times 10, \tag{5}$$

with h the thickness in m, d the apparent density, W_{fc} the water content at field capacity, and W_{wp} the water content at wilting point.

The W_{wp} is usually assessed as the equivalent 4.2 pF (15,000 hPa tensiometer pressure), thus around the W_s for the clay dominant soils. The difficulty is the choice of the W_{fc} equivalent to the 2.5 pF (330 hPa tensiometer pressure) which is "located" between the W_s and W_p of the clay dominant soils. The W_{fc} is mainly governed by the soil texture: that is, from 29% to 32% for clay-dominant soil to silty clay soils [23]. The W_{fc} and W_{wp} may be calculated taking into account the sand %, clay %, and organic matter % (OM) as follows [24]:

$$\begin{aligned} W_{\mathrm{fc}} &= 257.6 - (2 * \mathrm{sand\%}) \\ &\quad + (3.6 * \mathrm{clay\%}) + (29.6 * \mathrm{OM\%}), \\ W_{\mathrm{wp}} &= 26 + (5 * \mathrm{clay\%}) \\ &\quad + (15.8 * \mathrm{OM\%}). \end{aligned} \tag{6}$$

Thus the RU may be calculated for each soil layer as follows:

$$\mathrm{RU} = \left(W_{\mathrm{fc}} - W_{\mathrm{wp}}\right) * h, \tag{7}$$

with RU in mm and h the layer thickness in m.

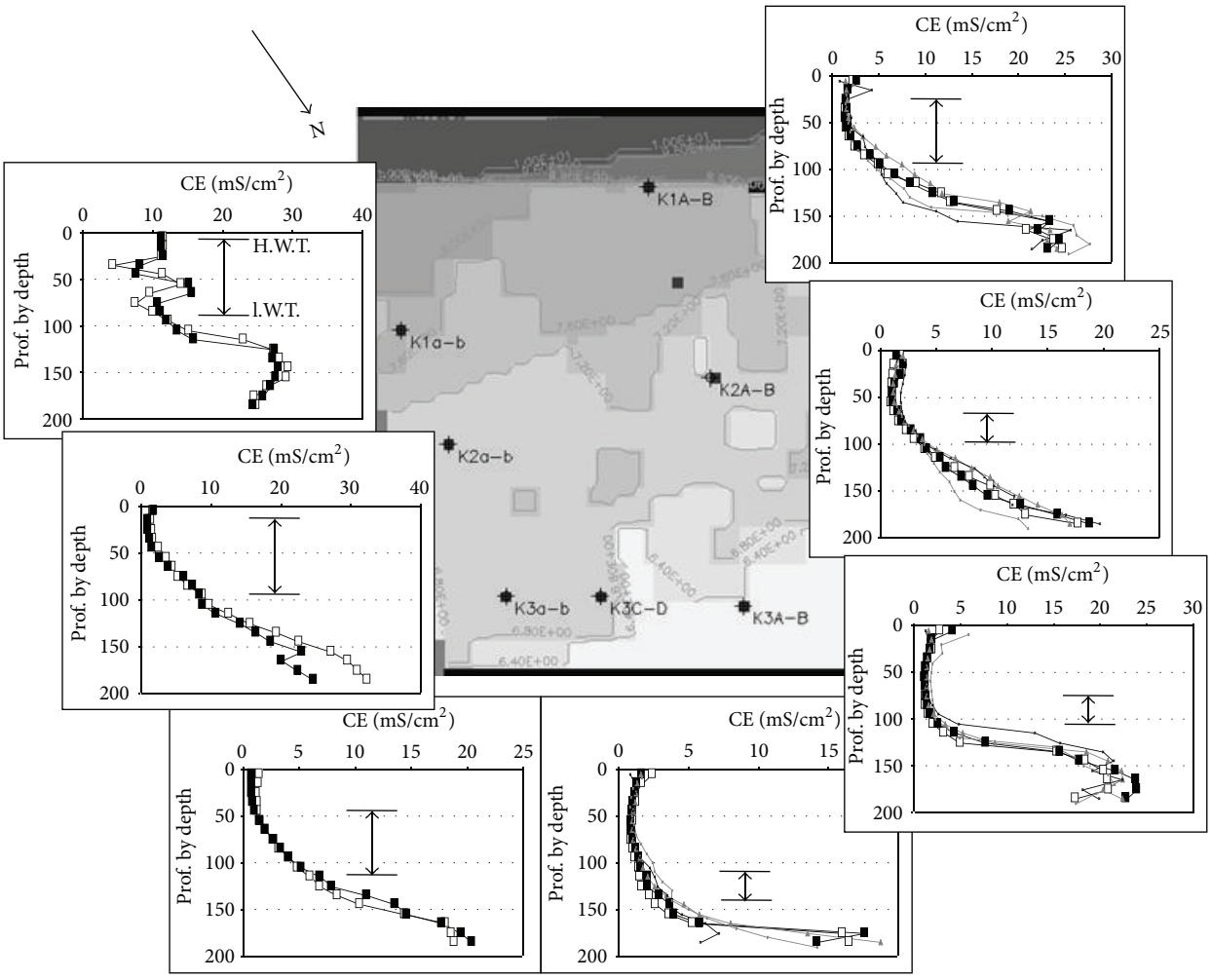

Figure 8: Representation of the $CE_{1/5}$ profiles measured in K1a-b, K1A-B, K2a-b, K2A-B, K3a-b, K3A-B, and K3C-D. The dates of measurements are September 2006 and April 2007 in K1a-b, K2a-b, and K3a-b. They are September 2006, April 2007, December 2007, and June 2008 in K1A-B, K2A-B, K3A-B and, K3C-D. The double arrows indicate the water table variation between the high levels (H.W.T.) and the low levels (l.W.T.).

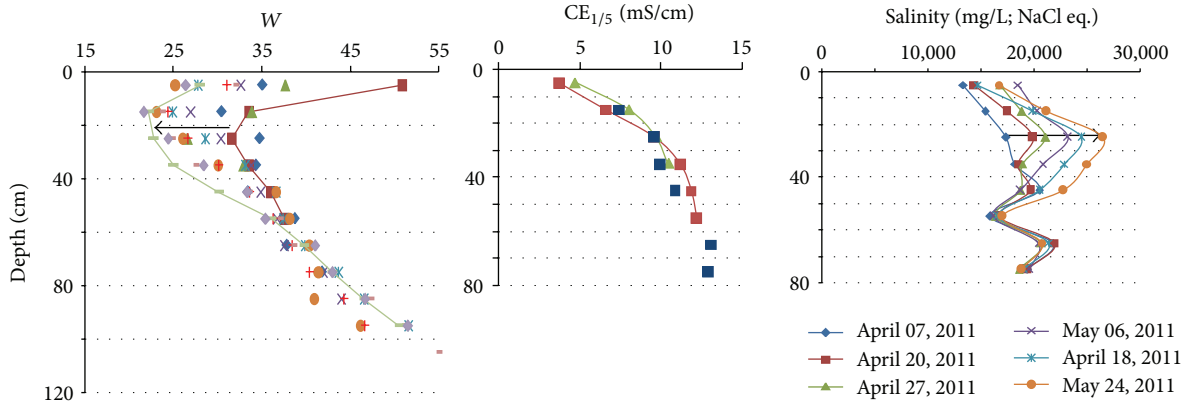

Figure 9: Example of the evolution of the salinity profiles associated with the measured W profiles (in grassland). The salinity profiles are calculated in NaCl equivalent, from 07 April to 24 May 2011.

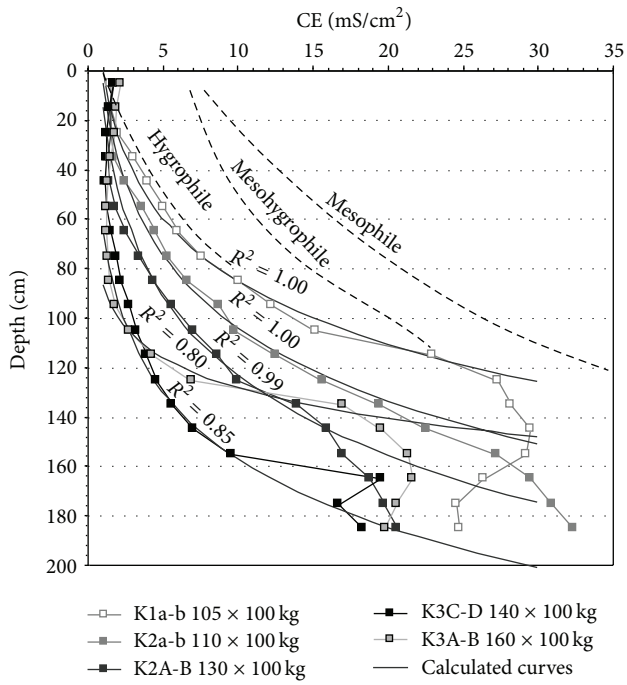

Figure 10: Fits of the calculated $CE_{1/5}$ patterns on the measured profiles from the surface down to the top of salt water table and associated yields in ×100 kg (2006 corn yields for K1a-b and K2a-b; 2008 corn yields for K2A-B, K3A-B, and K3C-D). The corn yields increase progressively from K1a-b to K3A-B following the increasing "slope" of the $CE_{1/5}$ profiles. Dashed line = domains of mesophile to mesohygrophile and hygrophile $CE_{1/5}$ profiles developed under the grassland.

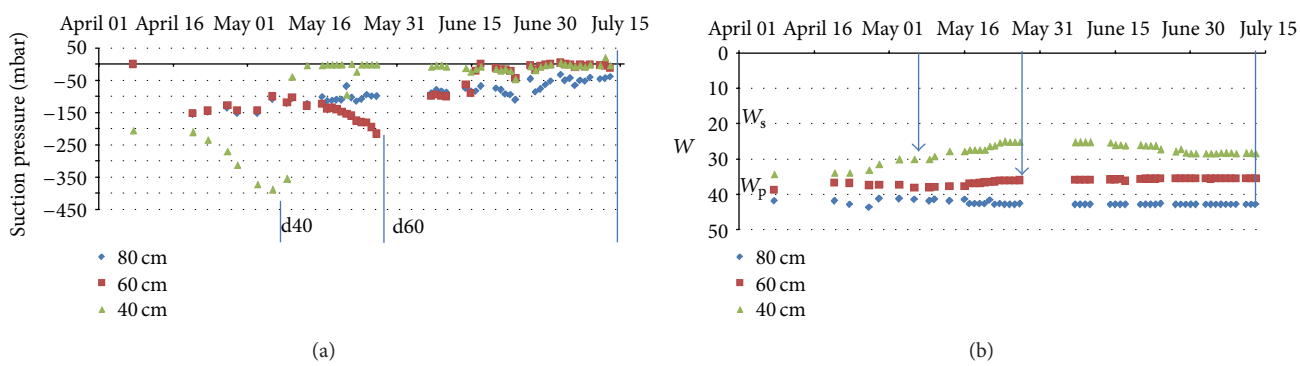

Figure 11: Evolution of the tensiometer pressures and gravimetric water contents at 40, 60, and 80 cm depth from 01 April 2011 to 15 July 2011 in the grassland. The d40 and d60 indicate the disconnection of the porous plugs at the depths of 40 and 60 cm, respectively.

According to their textures the soils developed from the bri exhibit two groups of RU values (Figure 14):

(i) the sediment-to-soil transition characterized by non-significant textural change. The material is composed of clay dominant material. The RU remains around 1.80 mm,

(ii) the soil surface and the paleosol characterized by silty clay soil texture. The RU remains around 1.80–2.00 mm.

In fact, if the soil may be overall characterized by its RU in mm/m, for a vertical profile, the RU is generally calculated as a cumulative RU which is obtained by the RU (mm/m) × depth for each level. The progressive soil desiccation which progresses from the surface down to the depth may be used to calculate the "real or residual" available water capacity (RUr) available for plants through the season. The real or residual soil RUr profiles may be calculated using the difference between the water profile at the considered date and the reference W_{fc} (Figures 14 and 15):

$$\mathrm{RUr} = h \times d\left(W_{7/04} - W_{wp}\right) \times 10, \tag{8}$$

with h the thickness in m, d the apparent density, $W_{d/m}$ the day/month measured water content at each depth, and W_{wp} the water content at wilting point.

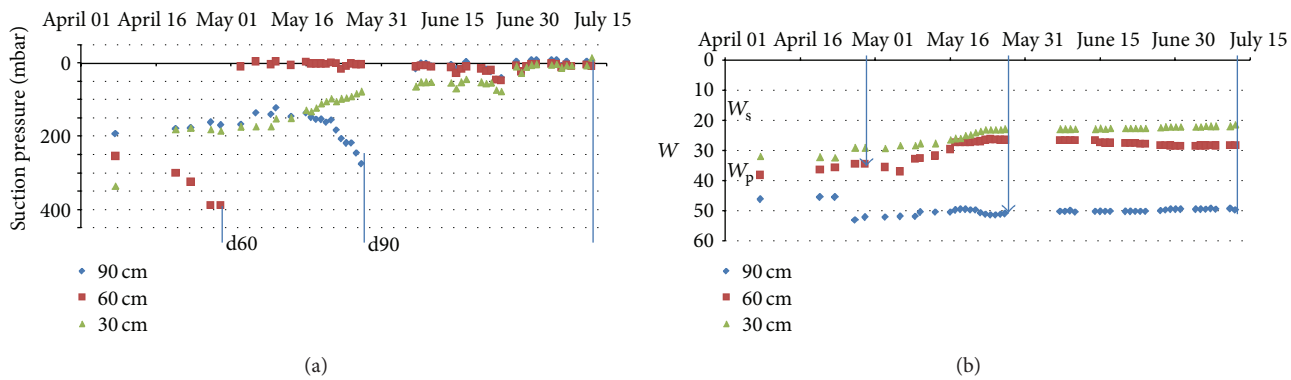

Figure 12: Evolution of the tensiometer pressures and gravimetric water contents at 30, 60, and 90 cm depth from 01 April 2011 to 15 July 2011 (K2A-B; wheat field). The d60 and d90 indicate the disconnection of the porous plug at the depths of 60 and 90 cm, respectively.

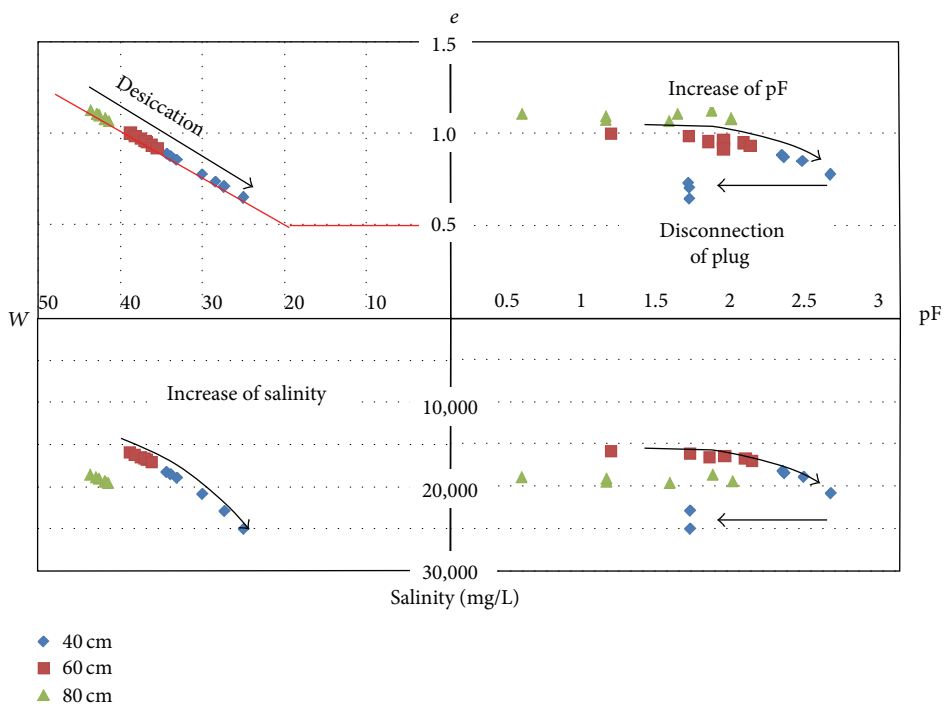

Figure 13: Example of W-e-pF-Salinity relationship (grassland). The shrinkage effect (W-e) causes the increase of the tensiometer pressure and associated pF (e-pF). The shrinkage fracture opening causes the disconnection of the porous plug. The salinity behavior can be represented due to the water content and tensiometer values.

This evolution of the real RUr profiles with time can inform on the soil-root interaction: that is, potential water consumption, wilting approach, and rooting in depth.

In the corn field and grassland examples, in April, the RUr profiles calculated by differences between the W profiles and the W_{wp} are quite superimposed on the theoretical RU = W_{fc}-W_{wp}. In May the RUr profiles calculated by differences between the W profiles and the W_{wp} are largely shifted to the 0 vertical axis (Figures 15, 16, and 17). The shift results from the water evaporation plus plant consumption: that is, synchronous descending of the desiccation front and rooting. On the other hand, in such brackish to salt media and due to the quite stable $CE_{1/5}$ profiles, the soil desiccation implies the increase of the water salinity for each level (Figure 18).

4. Discussion

4.1. Discrete Mineralogical Evolution. Whatever the authors, the impacts on the mineralogy of the clay dominant sediments have been interpreted as smectite-to-illite conversion associated to the soil maturation in oxidative conditions. The clay mineral evolution was demonstrated by Righi et al. [18] on soil surfaces dated from the ages of the successive polders in the west part of the Marais Poitevin. It was also demonstrated through vertical profiles sampled throughout the West French marshlands by the vertical decrease of the smectite contents from the sediment in depth to the soil surface. It was also verified in paleosols identified between 1 and 2 meters depths [9, 17]. In fact the smectite-to-illite

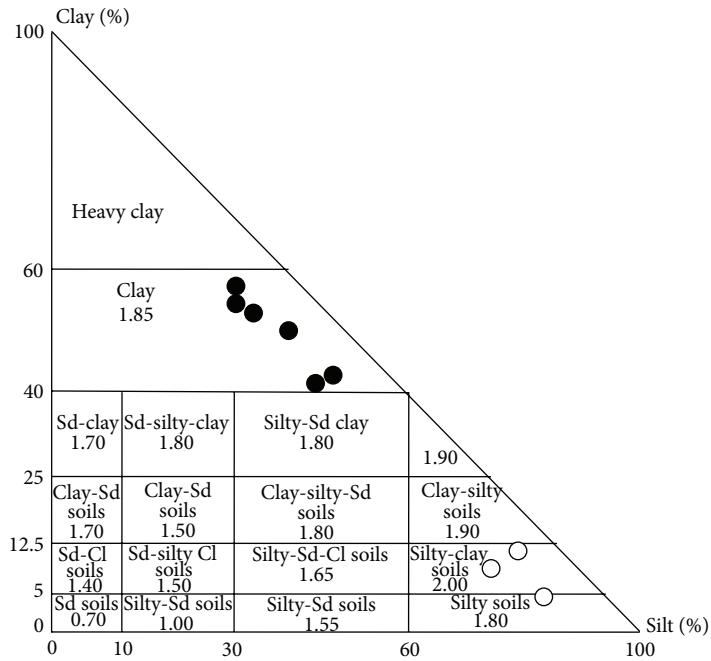

Figure 14: Location of the soil texture and associated available water capacity RU. Black circles = clay dominant sediment to soil; white circles = silty soil surface and paleosol identified on the INRA St Laurent experimental site.

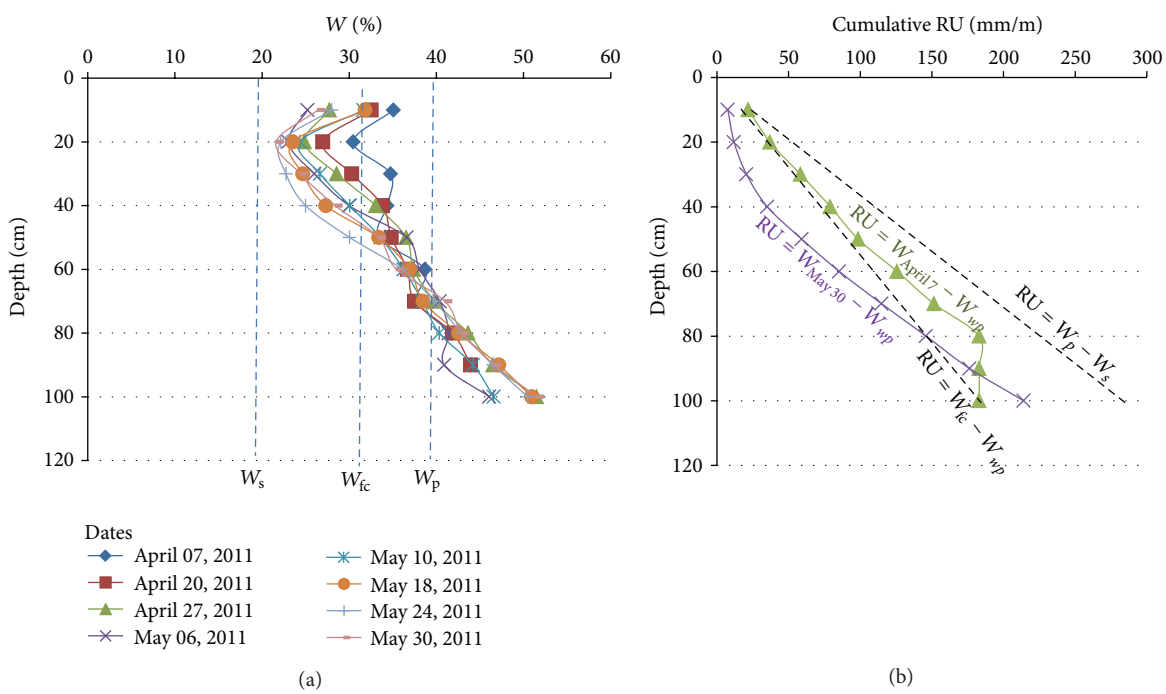

Figure 15: Evolution of the W profiles (a), RU profiles, and RUr profiles (b) from 07 April to 30 May in the grassland. W_p = plasticity limit; W_s = shrinkage limit; W_{wp} = wilting point ($W_{wp} = W_s$); $W_{30/05}$ = water profile at 30 May; $W_{7/04}$ = water profile at 07 April.

conversion has been observed via XRD patterns performed on the infra 0.02 μm fraction and only thanks to mathematical decomposition of the XRD patterns. The impact of the phenomenon is very weak due to the whole clay dominant soil behavior. This weak vertical decrease of the smectite content in the clay dominant matrix was also explained by a vertical leaching of the very thin particles by rain water. This second argument is based on the associated textural evolution from the surface to the depth. The analyzed soil surface (0–30 cm depth) and paleosol exhibits textures of more silty soils

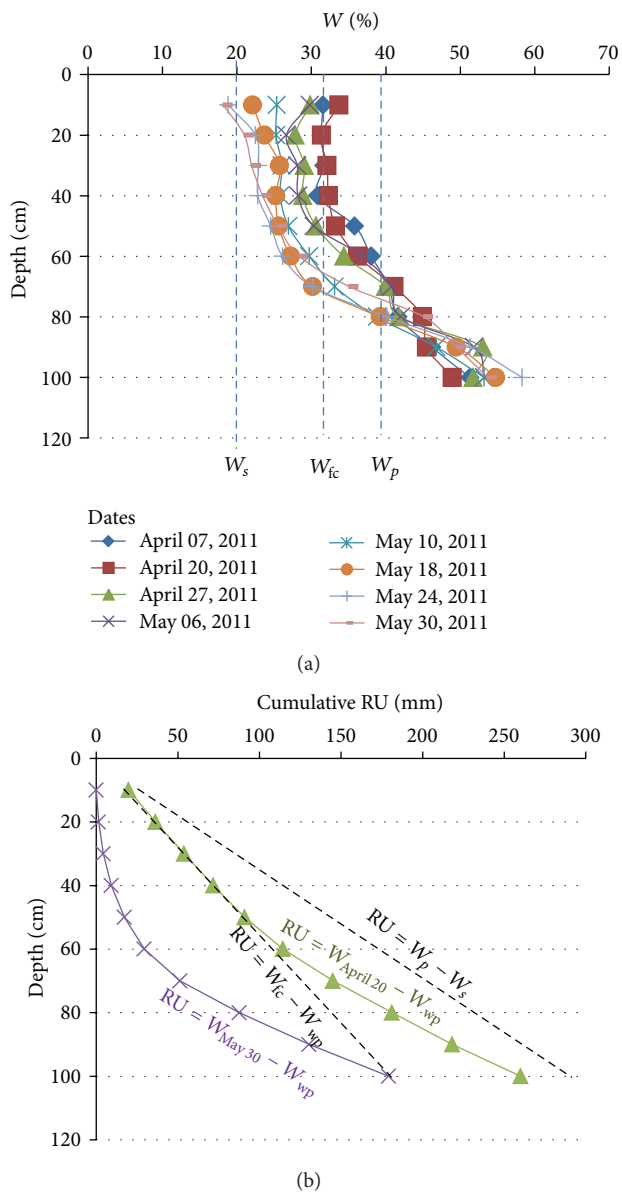

Figure 16: Evolution of the W profile (a), RU profiles, and RUr profiles (b) from 20 April to 30 May in the K1A-B under corn farming. W_p = plasticity limit; W_s= shrinkage limit; W_{wp} = wilting point ($W_{wp} = W_s$); $W_{30/05}$ = water profile at 30 May; $W_{20/04}$ = water profile at 20 April.

whereas the whole material exhibit clay dominant texture [9, 10]. In fact these very weak mineralogical and textural evolutions are common throughout the territories and do not imply significant differences on the marsh management.

4.2. Dominant Role of the Soil Structure Behavior. Since the Xth century the soil formation in the coastal marshlands is governed by the structural evolution of the primary sediment in surface caused by the desiccation mechanism. In depth, the primary clay dominant sediment, saturated by sea water, is consolidated because of the earth-or-sediment weight pressure. The two mechanisms induce similar structural evolutions of the clay matrix which are provoked by the clay particle rearrangement. In a void ratio (e) versus water content (W) the desiccation way and the consolidation way are superimposed (Figure 19) [13, 25, 26]. They provoke the outlet of interparticle water and the decrease of the microporosity by rearrangement of the clay particles [18].

Finally, the initial muddy sediment has been affected by two opposite mechanisms:

(i) the progression of the desiccation-shrinkage front from the surface to the depth, from W_s-W_p in surface down to the W_l state,

(ii) the consolidation in depth due to the earth pressure, from the W_l state to the W_p state.

The two mechanisms act to the vertical evolution of the bri in the three-layer structure [10, 12, 13, 15] (Figure 19):

(i) solid-to-plastic state near the surface characterized by low clay matrix hydraulic conductivity,

(ii) intermediate plastic-to-liquid state, medium hydraulic conductivity,

(iii) plastic-to-solid state in depth, low hydraulic conductivity.

The hydraulic functioning of the marshes is governed by this natural vertical structure caused by the hydraulic developments which have been successively built in order to accelerate the reclaiming and in particular to isolate the wet marshes from the dried marshes. The vertical structure and the very low Wp hydraulic conductivity limit the vertical inlet of rain water. On the contrary it allows the fresh water inlet from the peripheral limestone through the intermediate bri in liquid state.

The vertical evolution has been recorded on many sites by cone penetrometer, shear stress, bulk density, and water content profiles. Thanks to the textural and mineralogy homogeneity of the clay dominant material, the clay matrix shrinkage curve has been used as tool for the representation of the numerical relationships which can exist between the hydromechanical properties and the microstructure of the clay material [10, 12, 13, 15] (Figure 20). The microstructure-cone resistance (Qd) and microstructure-cohesion (C) relationships have been written as Perdok's equation or power law [10, 14, 15]. The same authors have presented the clay matrix microstructure-hydraulic conductivity relationship from simple calculations and oedometer measurements.

4.3. Desiccation Front Water and Salt Stress Relationship. The desiccation, locally accelerated by the drainage, has induced the descending of the water table and the increase of the solid-to-plastic layer thickness from the surface. Because of the high shrinkage properties of the clay material, this drying upper layer was crosscut by the shrinkage fracture network. In these conditions, the mechanism has allowed the surface desalination by the rain water leaching through the unsaturated plastic-to-solid clay material via the fracture network. These drying and desalination mechanisms have been the two main objectives of the reclaiming. Nevertheless the obvious questions are as follows.

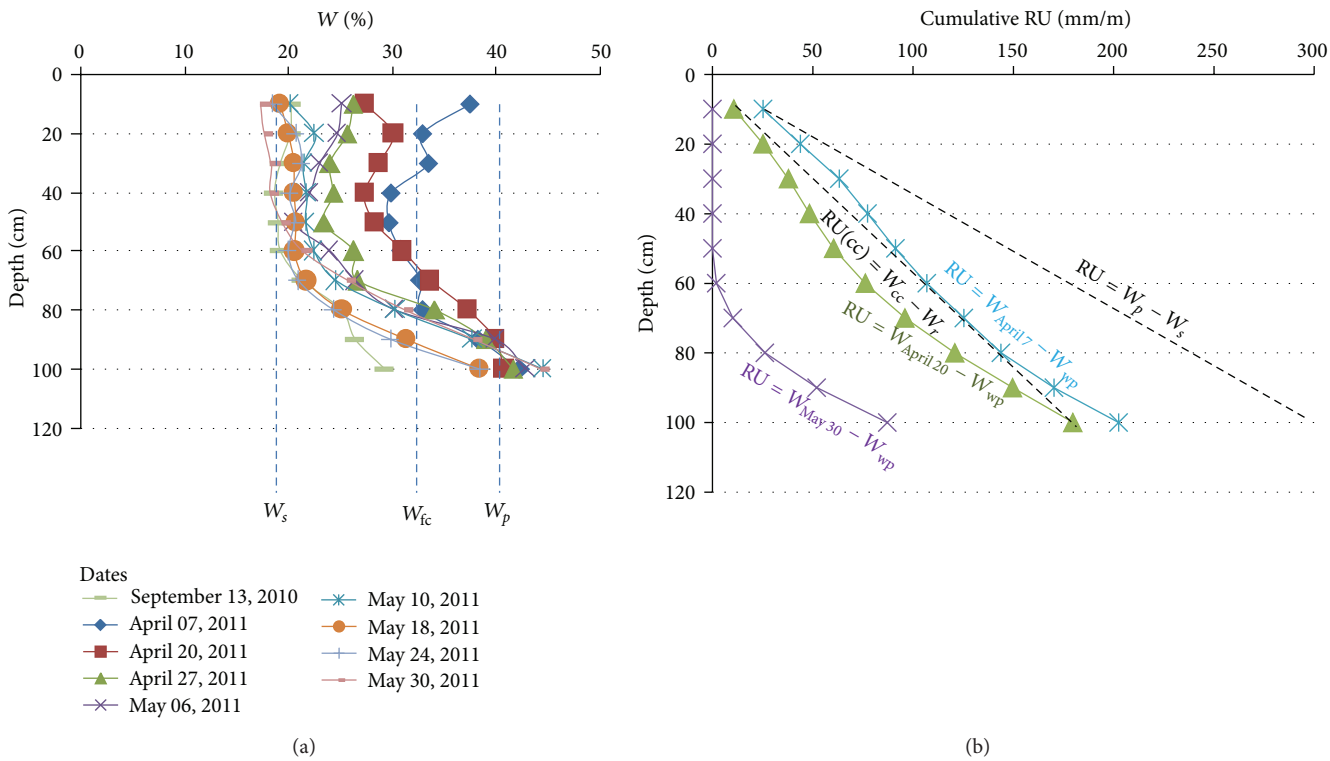

Figure 17: Evolution of the W profile (a), RU profiles, and RUr profiles (b) from 07 April to 30 May in the K2A-B spot. W_p = plasticity limit; W_s = shrinkage limit; W_{wp}= wilting point ($W_{wp} = W_s$); $W_{30/05}$ = water profile at 30 May; $W_{20/04}$ = water profile at 20 April; $W_{07/04}$ = water profile at 07 April.

(i) How to characterize the eventual fresh-to-salt water exchange and the soil salinity profiles?

(ii) What are the consequences on the soil-plant relationships?

(iii) Eventually, what are the most suitable tools or methods for mapping the fresh to salt area?

The tensiometer pressure measurements give information about the water content-microstructure couple during the desiccation phenomenon. The measured tensiometer pressures are governed by the reduction of micropores due to the desiccation-shrinkage coupling. The tensiometer pressure-equivalent pore diameter can be calculated via the Jurin law. The disconnection of the porous plug observed at the depths of 40, 60, and 90 cm operates following a time offset which may be interpreted as the descending of the desiccation front (Figures 11 and 12). The clay matrix desiccation induces the clay matrix shrinkage and provokes the additional opening of the mesopores and/or mesofractures (Figure 20(b)). The geometry of the shrinkage fracture network depends on the initial W-C couple of the clay matrix and shift of the clay matrix from the ductile-to-brittle behavior: that is wide and spaced fractures around the W_l domain and thin and close to each other in the W_p-to-W_s domain (Gallier, 2011 [14]; Gallier et al., 2012 [15]; Figure 20). In these marshlands the W profiles indicate the W increase with depth. Thus, for the successive depths, the mesopore-to-fracture opening operates at different initial hydric states of the clay matrix and different W-C couples. The differences between the tensiometer pressures associated with the plug disconnection at the different depths can be explained by the vertical evolution of the W profiles from W_s in surface to W_l: that is, dominant mesoporosity and close mesofractures near the surface and larger and spaced fractures in depth which can open for higher water contents (Figure 20).

The geo-electrical prospection is able to give vertical "images" of the bri structure down to the limestone basement. The Archie's law was calibrated taking into account the porosity and the water resistivity (equivalent salinity) for our clay dominant material [11, 13, 15]. Nevertheless, the surface recording of resistivity sections is limited in accuracy because of the measurement of apparent resistivity for apparent depths. Yet the resistivity-porosity relationship given by the Archie's law allows the transformation of the resistivity section into a permeability section which can be indicative of the local hydraulic running [13]. The most realistic vertical recording of resistivity profile can be obtained by coupling salinometer and water content profiles [11]. In fact the characterization of the profiles is easy by auger sampling which allows the density, water content, and $CE_{1/5}$ measurements for each depth of sampling. The water salinity is easily calculated from the couple W-$CE_{1/5}$.

The $CE_{1/5}$ profiles have progressively matured due to the descending water table. They are quite insensitive to the season, at least over a few years. Nowadays they represent the result of the effects of desiccation and desalination induced

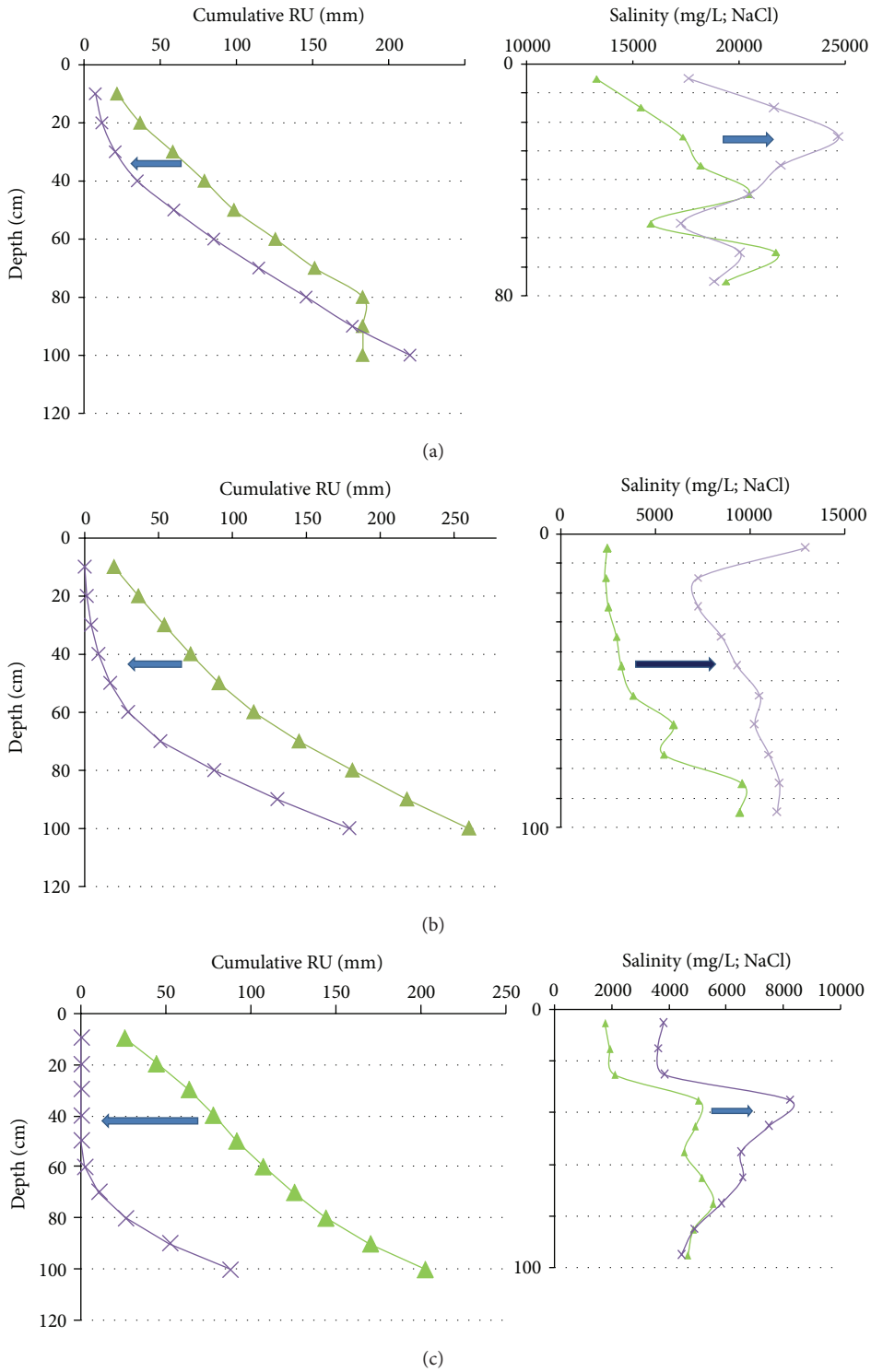

Figure 18: Simultaneous effects of the desiccation on the RUr profiles and on the water salinity. (a) In the grassland the high initial salinity drastically increases during the surface desiccation. (b) In the K1A-B the increases of salinity are moderated by the fresh water inlet from the limestone hill. (c) In the K2A-B the increases of salinity are moderated because of small residual salinity of the leached soil. Triangle = 07 April in K2A-B and grassland profiles and 20 April in K1A-B profiles; crosses = 30 May profiles.

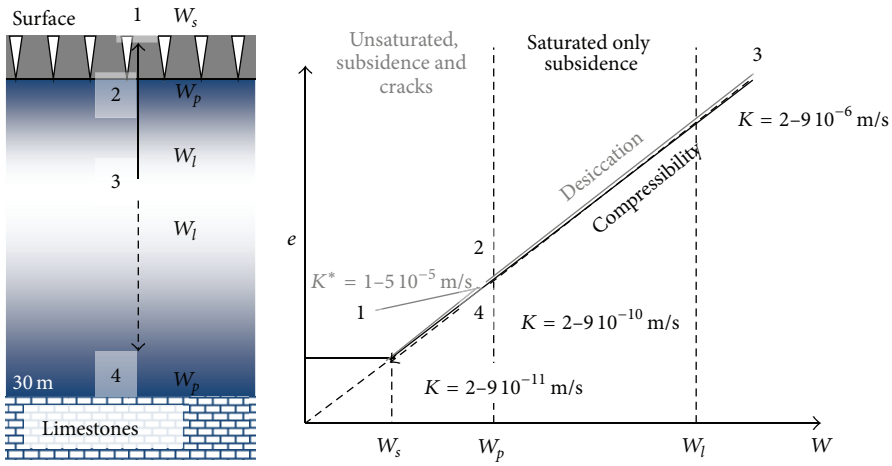

Figure 19: Schematic representation of the relations prevailing between the microstructure behavior of the clay matrix and the macroscopic behavior at the field scale during the surface desiccation phenomenon (3 to 2 to 1) and during the consolidation in depth (3 to 4). K = hydraulic conductivity measured by oedometer compressibility test on unfractured and saturated clay matrix K^* = surface hydraulic conductivity measured by Porcher test on the fractured soil (from Dudoignon et al. [13]).

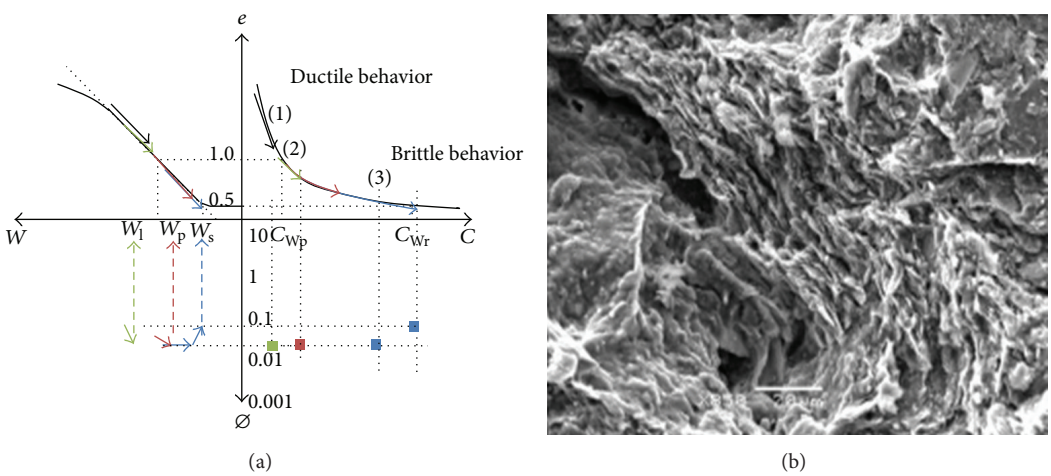

Figure 20: Schematic representation of the mesopore and fracture opening of the clay matrix for the different water contents associated with the increasing depths. (a) Ductile-to-brittle behavior of the clay material taking into account the W-C couple evolution and the successive steps of fracturing: (1) ductile behavior in the plastic-to-liquid state domain, (2) plasticity limit (W_p), (3) brittle comportment in the solid state [14]. W = gravimetric water content; e = void ratio; C = shear stress; Ø = equivalent pore diameter. (b) SEM microphotography of a mesopore adjacent to the W_s clay matrix.

by the successive reclaiming works since the Middle Ages. Two factors act on the $CE_{1/5}$ profile evolution: the depth of the water table quite equivalent to a thickness of soil leaching by the rain water and the eventual fresh water inlets. The mechanism is condensed by the observations made on the undrained grassland: the differences in leached layers are generated by the surface topography considering the water table horizontal, and the water inlet can be observed nearby the channel boundary (Figure 7). The thickness of leaching added to the peripheral fresh water inlet induces the progressive shift of the $CE_{1/5}$ profiles from a "mesophile profile," characterized by a very high $CE_{1/5}$ values (10–30 mS/cm) and a very weak slope of the profile, to a "hygrophile profile" characterized by low $CE_{1/5}$ (1–7 mS/cm) values and a hard slope of profile near the surface. In fact the comparison between the undrained grassland and the drained cultivated field allows to observe the continuity of $CE_{1/5}$ profile evolution from the undrained to drained domains (Figures 10 and 21(a)). The water table depth impacts directly on the shape of the $CE_{1/5}$ curves: slope evolution from the surface and thickness of leaching. The evolution from the mesophile systems to the hygrophile ones clearly appears in the RUr and associated salinity profiles (Figure 21(b)). For equivalent initial cumulative RU profiles, due to the homogeneous texture, the initial salinity of the hygrophile systems (3 and 2) is restrained to the 2–5 mS/cm domain (Figure 21(a)). For the mesophile system the salinity is between 10 and 20 mS/cm. Finally three types of couple salinity-RUr profiles can be

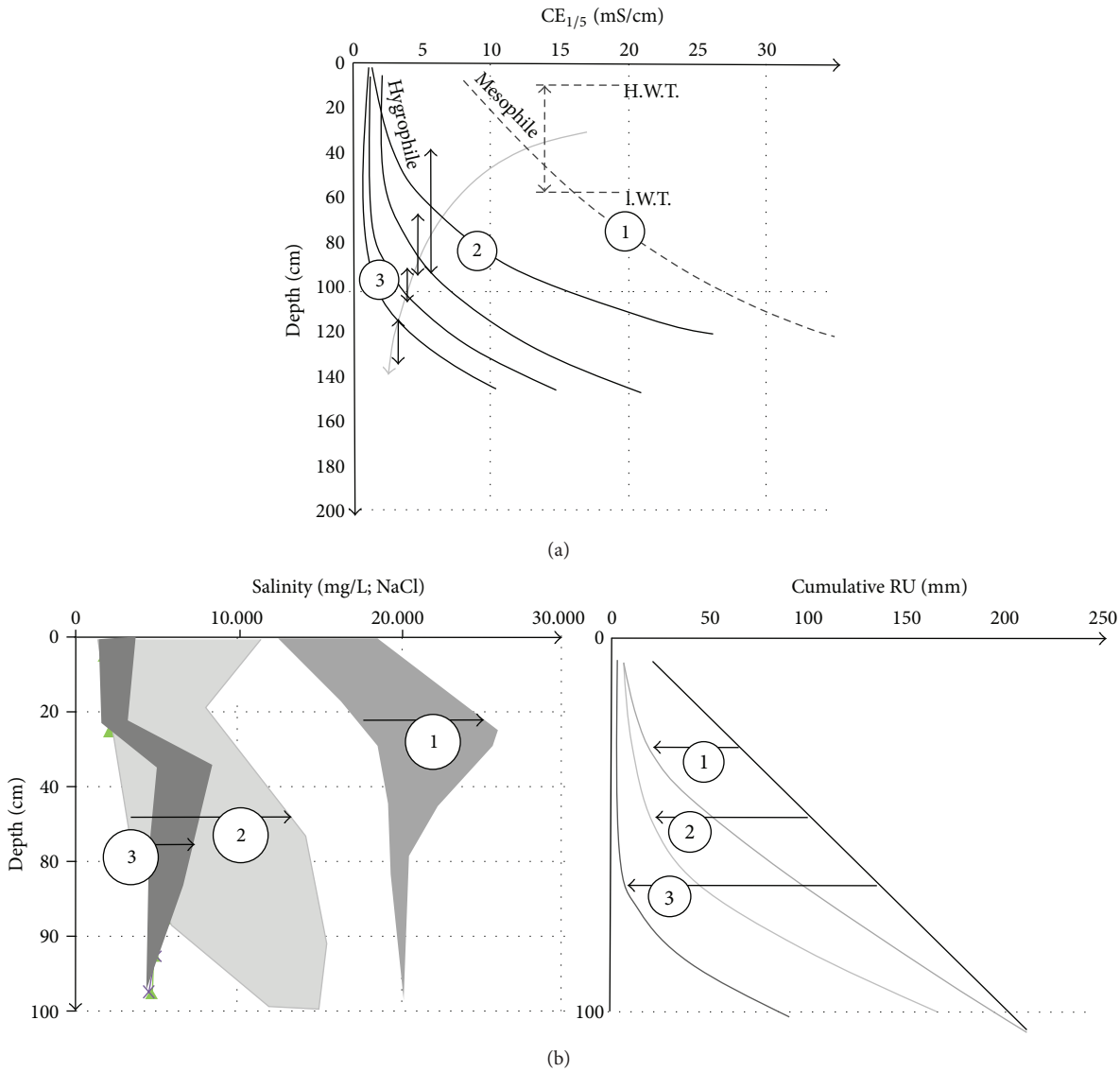

Figure 21: (a) Schematic evolution of the $CE_{1/5}$ profiles induced by the water table descending. Grey arrow = water table trend; double vertical black arrows = water table fluctuation between the high water table level (H.W.T.) and the low water table (l.W.T.). 1 to 2 indicate the transition from the mesophile to hygrophile systems. (b) Parallel evolutions of the RU profiles and salinity profiles characteristic of the three studied systems. Straight black line = initial (textural) cumulative RU profiles. 1, 2, and 3 residual cumulative RUr profiles calculated after the plant growing under the grassland and under the cultivated field (example of wheat). Trend 1 = grassland; trend 2 = cultivated area nearby the limestone hill, thus impacted by the fresh water inlet; trend 3 = cultivated area far from the limestone hill.

distinguished for the different spots. They are governed by presence or not of drainage, the proximity of the fresh water source, and the difference in plant farming. They describe the soil functioning from the mesophile to hygrophile systems.

The mesophile profile type is characteristic of the undrained territories which are traditionally occupied by cattle, thus let in grassland. The water consumption by the grassland is temperate. Thus the residual RUr profile is weakly shifted from the initial RU line (Figure 21, trend 1). Nevertheless the increase of soil salinity is very high because of the residual salinity due to the lack of leaching.

The hygrophile profile types exhibit two trends: trend 2 corresponds to the nearby limestone profile with fresh water inlet and trend 3 far away the hill. In trend 2, the fresh water inlet allows the high water consumption by the plant. The shift of the residual RUr profile from the initial RU is moderated. The evolution of the salinity between the initial and residual RU profiles is significant. It results from the fresh water plus salt water mixture. In trend 3, the lack of water inlet due to the water consumption by plants induces the high shift of the RU profile from the initial one. The general low water table level is associated with the high thickness of leached

layer; thus the initial low salinity limits the increase of salinity during the desiccation period.

5. Conclusion

The successive works made on the French Atlantic marshlands in recent years have shown the role of the clay matrix microstructure on the clay dominant soils behavior. The results have been obtained thanks to the mineralogical and textural homogeneity of the initial sediments and thanks to the very low impact of the reclaiming, dated from the Middle Ages, on these two parameters. The sediment-to-soil evolution has mainly been governed by the desiccation-shrinkage mechanisms of the clay matrix. Thus, the soil microstructure-hydromechanical property relationships have been represented in successive clay matrix shrinkage curve-cone resistance or cohesion-hydraulic conductivity crossed diagrams [9–15]. The clay matrix behavior allows the explanation of the hydrologic functioning of the marshlands taking into account the desiccation effect of surface on the clay dominant soils and the compressibility effect on the clay dominant sediment in depth.

The two main objectives of the reclaiming have been the descending of the water table levels and the desalination of soil. Sufficient for extensive grazing grounds, the reclaiming has been completed by the drainage for the cereal cultivation from 1970. The interaction between the water table level and the soil salinity is easy to follow using the $CE_{1/5}$ profiles. These ones are characteristics of the hydraulic history of the territories. Quite invariant during the seasons, they can be explained as the result of the water table management many years ago. Moreover, they can be used to predict the crop yields.

The shift of the study from the grassland to the drained cultivated field allows the presentation of a quantitative evolution of the $CE_{1/5}$ soil profiles from the undrained to the drained lands. Thanks to the Montoroi equation [14, 20] and coupled with the moisture profiles they allow the calculation of the water salinity profiles and their evolutions during the seasons. Thus, the *in situ* tensiometer recordings and calculated water salinity can be represented taking into account the clay matrix microstructure in a W-e-pF-water salinity crossed diagram.

Finally, the soil-plant interaction can be "modeled" for the grassland and cultivated fields taking into account the location of the studied spots relative to the fresh water inlet from the peripheral limestones. The "model" is based on the shift of the "initial" available water capacity profiles of soil to the "residual" available water capacity profiles resulting from the plant growing and associated shift of the water salinity profiles. The results show the evolutions of the different associated profiles in the mesophile, mesohygrophile, and hygrophile systems from undrained to drained territories.

The method can be used to characterize the available diversity of the plants due to the different domains of water and salt stress on these territories reclaimed on fluviomarine deposits. It can be used as diagnostic tool and/or projection tool for the crop yields but also for the flora evolution due to extension of the "dry" territories against the residual "wet" territories which can be induced by different choices of hydraulic managements or by the climatic changes. In the opposite direction it can be used as projection tools for the evaluation of the eventual "dry" marsh rewetting.

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Research Article

Phytoplankton and Eutrophication Degree Assessment of Baiyangdian Lake Wetland, China

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Eight typical sampling sites were chosen to investigate the phytoplankton community structure and to assess the eutrophication degree of Baiyangdian Lake in 2009. Our results showed that among the total 133 species identified, Cyanophyta, Chlorophyta, and Bacillariophyta dominated the phytoplankton community. In spring, Chlorophyta and Bacillariophyta were the dominant phyla, and the dominant species included *Chlorella* sp., *Chroomonas acuta* Uterm., and *Microcystis incerta* Lemm.; the density of the phytoplankton ranged from 496×10^4 to 6256×10^4 cells/L with an average of 2384×10^4 cells/L. However, Chlorophyta and Cyanophyta became the dominant phyla in summer, and the dominant species were *Chlorella* sp., *Leptolyngbya valderiana* Anagn., and *Nephrocytium agardhianum* Nageli.; the density of the phytoplankton varied from 318×10^4 to 4630×10^4 cells/L with an average of 1785×10^4 cells/L. The density of the phytoplankton has increased significantly compared to the previous investigations in 2005. The index of Carlson nutritional status (TSIM) and the dominant genus assessment indicated that the majority of Baiyangdian Lake was in eutrophic state.

1. Introduction

Eutrophication is a phenomenon, in which the excess trophic substances (i.e., nitrogen and phosphorus) in lakes, reservoirs, estuaries, rivers, and certain coastal waters cause a great increase in algae and a decrease in dissolved oxygen, thus, leading to serious death of a lot of fishes and other hydrophytes. A lot of freshwater water bodies have occurred eutrophication in the 1990s [1], and it was the first time for eutrophication began to become the major pollution problem of lakes and reservoirs in most countries such as Europe and North America. An investigation showed that eutrophication took place in 54% of Asian lakes, 53% of European lakes, 48% of lakes in North America, 41% of lakes in South America, and even 28% of lakes in Africa [2]. For example, Japan's second largest lake (Lake Kasumigaura) once played a very important role in irrigation, human life, industry, inland fishery, recreation, and so on; however, serious eutrophication has been observed in this lake since the early 1900, and this once well-known tourist attraction was forced to shut down due to the deteriorative water quality [3].

There are in total 2,759 lakes with the area of more than $1\,km^2$ in China, which covers an area of $91019\,km^2$ and makes up 0.95% of land area in China. About one-third of these lakes is fresh water lake, and they are mainly located in eastern coastal area and the lower-middle reaches of Yangtze River. In recent years, the rapid development of economy, the inappropriate development and utilization of water resources, and the worsening agriculture nonpoint pollution have resulted in serious eutrophication: most of these lakes have been plagued by eutrophication or are in the course of eutrophication development [4]; the structure and function of the lake ecosystems have degenerated with the frequent occurrence of blue-green algae blooms and pollution-induced water shortage. These water problems seriously affect the production activities and life of people in the lake region, limit the sustainable development of regional

social economy, and cause great financial losses and social problems [5].

Phytoplankton including many species is widely distributed in the aquatic ecosystem, which maintains the structural functions of ecosystem and plays an important and irreplaceable role of indicator and purifier on lake pollution, through participating in material cycle and energy flow in lakes [6, 7]. In recent years, the species and community structure characteristics of phytoplankton combined with the chemical detection of water quality have been generally accepted as environmental assessment indicators [8]. Among them, the index of Carlson nutritional status (TSIM) and the dominant genus assessment have been widely applied for the assessment of eutrophication in lakes and reservoirs [9, 10].

Baiyangdian Lake, located in the eastern part of North China Plain, is the largest fresh water lake in Haihe River Basin. It is also the largest inland shallow lake in North China and is historically given the name of "the Pearl of North China" because of its excellent water quality and great biodiversity. However, in the highly populated Baiyangdian Lake area, industrial effluents and domestic sewage flow directly into the lake through surface runoff [11], upstream water supply declines, and water level drops, which results in low self-purifying capacity of this lake. As a result, the water quality of this lake is increasingly more worse [12] with serious eutrophication [13].

The objectives of this paper were to identify the species composition and spatial distribution of the phytoplankton and to evaluate the eutrophication degree in Baiyangdian Lake.

2. Materials and Methods

2.1. Study Area. Baiyangdian Lake, with an area of about 366 km^2, is a typical plant-dominated shallow freshwater lake consisting of about 143 lake parks, with a maximum depth of 4.0 m. Reeds and cattails are the dominant macrophyte, covering about 22% of the lake area. Raised fields with reeds and cattails coverings are divided into multiple blocks by more than 3,700 artificial ditches. These ditches are regarded as multiple connected corridors in the reed wetland landscape. It has the semiarid monsoon climate with the average annual precipitation of 570.2 mm, and the average annual evaporation in this region is 1369 mm, which is far higher than the average annual precipitation [14]. Meanwhile, the reduction of lake surface area and depth worsens the eutrophic situation.

2.2. Sample Collection and Analysis. Eight sampling sites were chosen in the Baiyangdian Lake (115°56′ to 116°06′E, 38°49′ to 38°57′N; Figure 1), and phytoplankton and relevant environment factors were investigated in the early April (spring) and in the middle of June (summer) of 2009. Quantitative sample collection of phytoplankton is as follows: organic glass hydrophore was used to collect water samples; the volume of water sample was 1000 mL; for water which was no more than 3 meters deep and the water masses of which were well mixed, one sample of 1000 mL needed to be collected from the surface (0.5 meter), while for water which was 3 to 10 meters deep, one sample of 500 mL from the surface and one sample of 500 mL from the bottom needed to be collected and mixed. Lugol's solution was used to fix samples, and 1% (vol) formalin solution was used to preserve samples. The treatment, analysis method, and water quality analysis method of the phytoplankton samples were carried out according to the standard methods from *Lake Ecosystem Observation Method* [15].

2.3. Data Processing. Data were proceeded using SPSS and PRIMER V6 software packages [16] to get ecological indicators such as species number, density, Shannon-Weiner index, and richness index of phytoplankton.

Shannon-Weiner index is calculated using formula (1):

$$H' = -\sum_{i=1}^{s} P_i \ln P_i, \tag{1}$$

where P_i represents the percentage of species i in the samples; for example, if the total number of species is N and the number of species i is n_i, then $P_i = n_i/N$.

Richness index is calculated using formula (2):

$$d = \frac{(S-1)}{\log_2 M}, \tag{2}$$

where S is the number of species in the samples collected from certain sampling sites and M is the number of individuals of all the species in this sampling site.

Uniformity is calculated using formula (3):

$$J' = \frac{H'}{\ln S}, \tag{3}$$

where H' is the Shannon-Weiner index and S is the number of species in the samples collected from certain sampling sites.

3. Results and Discussion

3.1. Community Composition and Biodiversity. All the phytoplankton collected in both dates could be categorized into 8 phyla and 133 species (genus). Amongst them, 8 phyla and 78 species (genus) were observed in spring, and Chlorophyta, Cyanophyta, and Bacillariophyta were the dominate phytoplankton community with the greatest number of Chlorophyta including 33 species (genus), which accounted for 42.3% of the total number of algae. Fifteen species (genus) of Bacillariophyta and 11 species (genus) of Cyanophyta were observed, accounting for 19.2% and 14.1% of the total number of algae, respectively. There were 7 species (genus) for Euglenophyta and 5 species (genus) for Cryptophyta, accounting for 9.0% and 6.4% of the total number of algae, respectively. However, there were only 2 species (genus) for Pyrrophyta, 2 species (genus) for Xanthophyta, and 3 species (genus) for Chrysophyta, accounting for 2.6%, 2.6%, and 3.8% of the total number of algae, respectively, (Figure 2(a)).

Eight phyla and 133 species (genus) were observed in summer, and Chlorophyta, Cyanophyta, and Euglenophyta

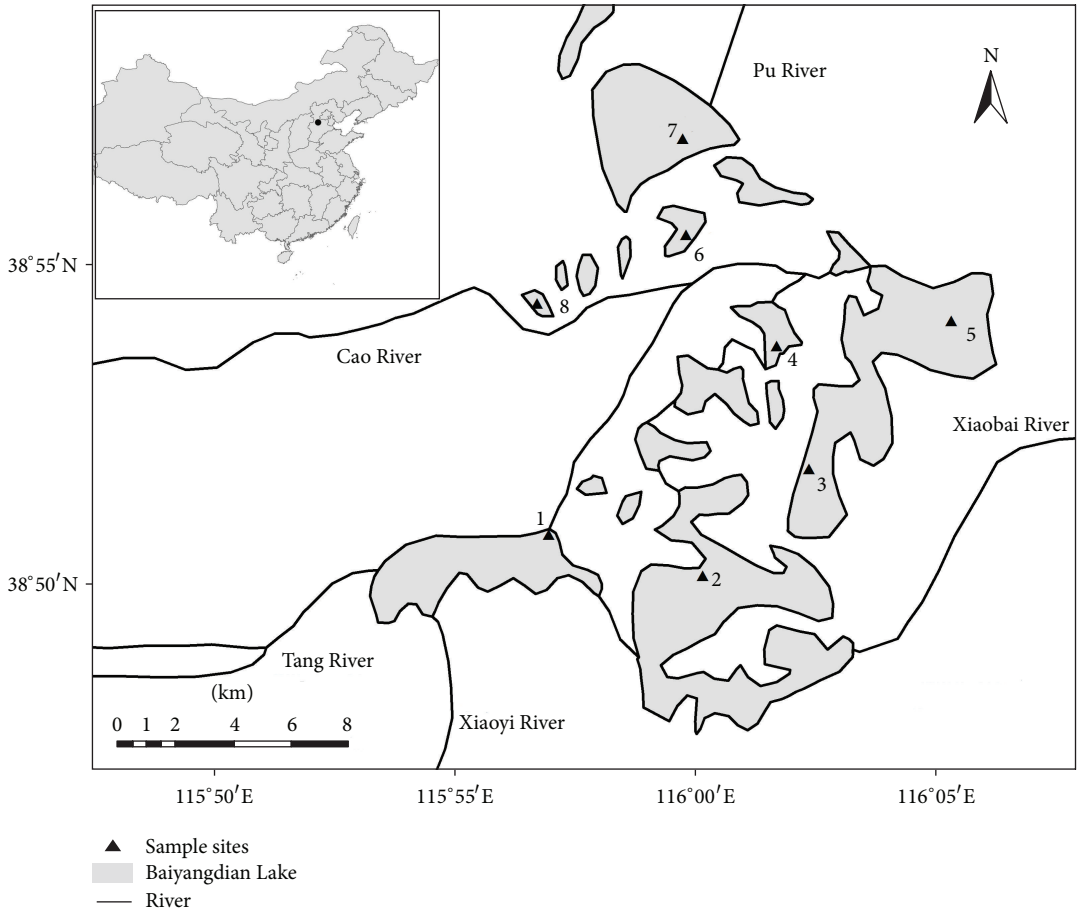

Figure 1: Sampling sites of the Baiyangdian Lake in Hebei Province, China.

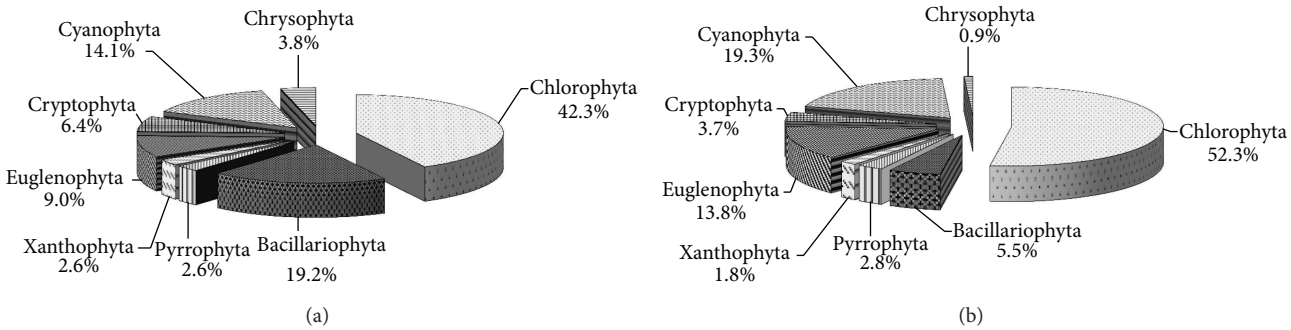

Figure 2: Composition percentage of the main phytoplankton in the study area in spring (a) and summer (b).

were the dominate phytoplankton community with the greatest number of Chlorophyta including 57 species (genus), which accounted for 52.3% of the total number of algae. Twenty-one species (genus) of Cyanophyta and 15 species (genus) of Euglenophyta were observed, accounting for 19.3% and 13.8% of the total number of algae, respectively. There were 6 species (genus) for Bacillariophyta and 3 species (genus) for Pyrrophyta, accounting for 5.5% and 2.8% of the total number of algae, respectively. However, there were only 2 species (genus) for Xanthophyta, 4 species (genus) for Cryptophyta, and 1 species (genus) for Chrysophyta, accounting for 1.8%, 3.7%, and 0.9% of the total number of algae, respectively, (Figure 2(b)). In spring, the first dominant species in Baiyangdian Lake is *Chlorella sp.* which belongs to the Chlorophyta phylum with the occurrence frequency of 100%. The second dominant species are *Chroomonas acuta* Uterm. and *Microcystis incerta* Lemm., which belong to the Cryptophyta and Cyanophyta phyla with the occurrence frequency of 87.5% and 100%, respectively. In summer, the first dominant species in Baiyangdian Lake is *Chlorella sp.* which

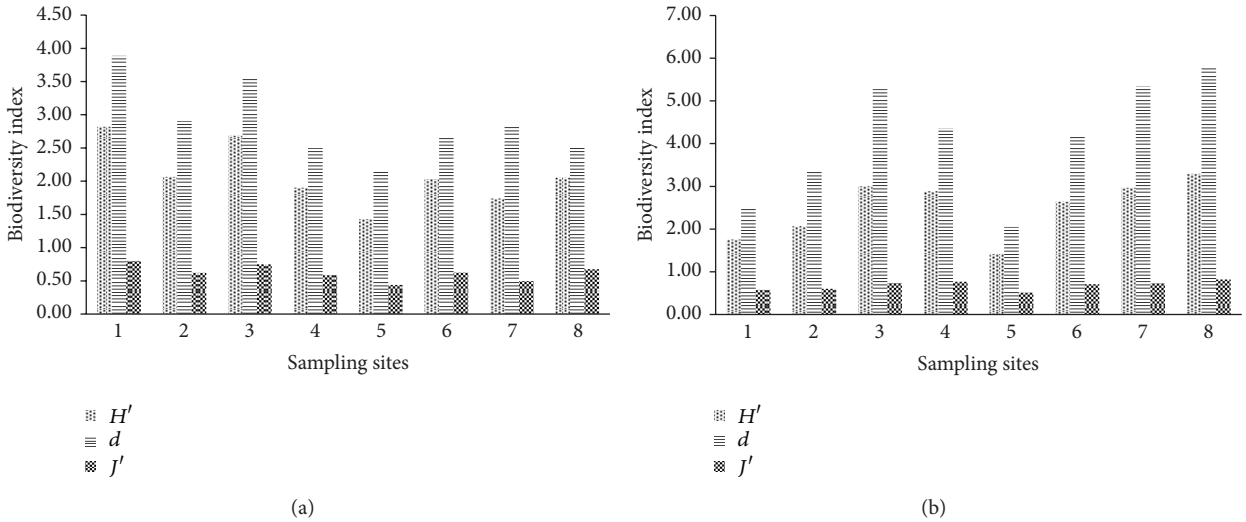

Figure 3: Biodiversity index of the phytoplankton in different sampling sites in Baiyangdian Lake in spring (a) and summer (b).

belongs to the Chlorophyta phylum with the occurrence frequency of 100%. The second dominant species are *Leptolyngbya valderiana* Anagn. and *Nephrocytium agardhianum* Nageli. which belongs to the Cyanophyta and Chlorophyta phyla with occurrence frequency of 70% and 88%, respectively.

Figure 3 shows the biodiversity index of the phytoplankton in Baiyangdian Lake sampling sites in spring (a) and summer (b). As shown in Figure 3(a), the Shannon-Weiner index of phytoplankton was 1.43~2.82 with an average of 2.09, and the species richness index of phytoplankton was 2.15~3.89 with an average of 2.88. The Shannon-Weiner index for Site 5 was the lowest (1.43), and the richness index is 2.15; the Shannon-Weiner index for Site 1 was the highest (2.82), and the richness index is 3.89. It is obvious that the space distribution tendency of Shannon-Weiner index and species richness index was consistent. The uniformity index was 0.44~0.80 with an average of 0.62. In summer, the Shannon-Weiner index of phytoplankton was 1.42~3.29 with an average of 2.50, and the species richness index of phytoplankton was 2.05~5.78 with an average of 4.11 (Figure 3(b)). The Shannon-Weiner index for Site 5 was the lowest (1.42), and the richness index is 2.05; the Shannon-Weiner index for Site 8 was the highest (3.29), and the richness index is 5.78. The space distribution tendency of Shannon-Weiner index and species richness index was consistent. The uniformity index of phytoplankton was 0.51~0.82 with an average of 0.68. According to this, in Baiyangdian Lake, there were more species of phytoplankton in summer than in spring but there was no dominant group; Chlorophyta and Cyanophyta dominated the community in both spring and summer. In sampling sites close to residential area, fish culturing cages, livestock, and poultry farms which were under great influence of human activities, there were more varieties of phytoplankton; while in sampling sites, which were situated in lake outlets and in standing water, there were fewer varieties of phytoplankton. This is consistent with the results in previous studies of phytoplankton in Baiyangdian Lake [17].

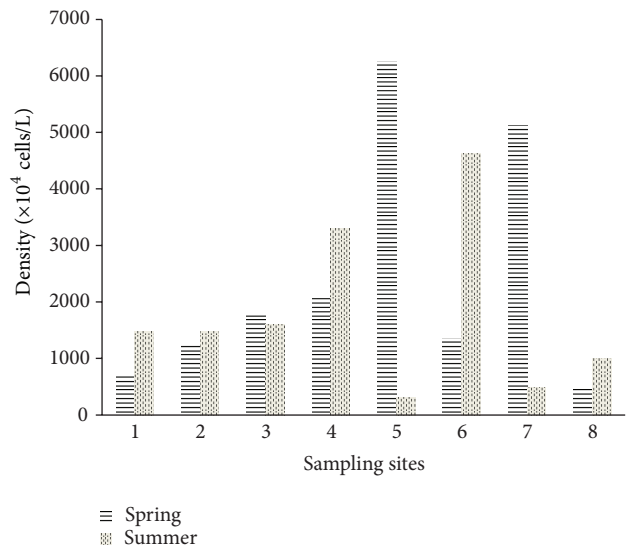

Figure 4: Comparison of phytoplankton density in different sampling sites in Baiyangdian Lake in spring and summer.

3.2. Phytoplankton Density. The number of phytoplankton in Baiyangdian Lake varied greatly between spring and summer. In spring, the density of phytoplankton varied from 496×10^4 to 6256×10^4 cells/L with an average of 2384×10^4 cells/L; the density varied greatly between different sampling sites: the density of phytoplankton cells in Sites 5 and 7 was high, while the density of Sites 1 and 8 was low. In summer, the density of phytoplankton varied from 318×10^4 to 4630×10^4 cells/L with an average of 1785×10^4 cells/L (Figure 4); the density varied greatly between different sampling sites: the density of phytoplankton cells in Sites 4 and 6 was high, while the

Table 1: Variations of phytoplankton density and dominant species.

Year	Number (species)	Density ($\times 10^4$ cells/L)	Dominant species	Number (species)	Percentage %	Dominant species	Number (species)	Percentage %	Data resource
2005	152	664.4	Chlorophyta	80	52.6	Cyanophyta	28	18.4	[20]
2006	155	518.2	Chlorophyta	81	52.3	Cyanophyta	29	18.7	[20]
2009	133	2084.6	Chlorophyta	65	48.9	Cyanophyta	22	16.5	This study

Table 2: Correlation analysis between phytoplankton density and environmental factors of Baiyangdian Lake in spring.

	T	DO	COD	COD_{Mn}	pH	TN	TP	Chla	SD	Density
T	1									
DO	0.024	1								
COD	0.760*	−0.454	1							
COD_{Mn}	0.867**	−0.285	0.948**	1						
pH	0.458	0.753*	0.022	0.108	1					
TN	0.274	−0.476	0.566	0.602	−0.249	1				
TP	0.286	−0.428	0.550	0.600	−0.207	0.998**	1			
Chla	0.627	0.513	0.442	0.492	0.851**	0.057	0.089	1		
SD	−0.069	−0.212	−0.264	−0.220	−0.439	−0.382	−0.399	−0.687	1	
Density	0.644	−0.132	0.651	0.543	0.308	−0.223	−0.241	0.512	−0.067	1

*Significant correlation at the level of $P < 0.05$; **significant correlation at the level of $P < 0.01$.

density of Sites 5 and 7 was low. Generally speaking, the density of phytoplankton and dominant species can indicate the eutrophication degree of certain water. As species of high pollution tolerant, if Cyanophyta increases sharply and finally becomes the dominant species, it indicates that the water is eutrophic, that is to say, the higher the density of Cyanophyta cells is, the more serious the eutrophication is [18]. According to this survey, cyanophyta and Chlorophyta became dominant in the phytoplankton community, that may be caused by the increased organic matters after organic matters in industrial wastewater and domestic sewage came into Baiyangdian Lake [19] and resulted in the increase of varieties and number of phytoplankton, especially these species with high pollution tolerance.

To get a general understanding of the change of phytoplankton community in Baiyangdian Lake, data of the three times surveys of phytoplankton since 2005 were compared (Table 1), and dynamic variations of phytoplankton in Baiyangdian Lake in recent years were analyzed from the aspects of species composition, density, and dominant species. The data of the years 2005 and 2006 were averages from the 8 sampling sites in Baiyangdian Lake in spring and summer, and the data of the year 2009 were averages from the current survey in 8 sampling sites in Baiyangdian Lake in spring and summer. The survey methods were the same.

Table 1 shows the variations of phytoplankton density and dominant species in Baiyangdian Lake in recent years. Taking the year 2005 as the reference point, the average phytoplankton density decreased 0.22 times in 2006, whereas the average phytoplankton density increased 2.41 times in 2009. In the three surveys, Chlorophyta and Cyanophyta were the mainly dominant species. The annual variations tendency of different algae phylums were different; the phytoplankton variety number tended to decrease, while the phytoplankton density tended to increase. In the current survey, the phytoplankton density increased observably, because the nutrient load in Baiyangdian Lake changed; when sampling, algae were growing and blooming, and as the temperature became higher and the light became stronger, the water environment became more favorable for the growth of phytoplankton which abundance increased as a result. Generally speaking, in mesotrophic lakes, Pyrrophyta, Cryptophyta, and Bacillariophyta are the dominant species, and in eutrophic lakes, Chlorophyta and Cyanophyta are the dominant species [21]. From this, a conclusion can be drawn that *Chlorella sp.* and *Microcystis incerta* Lemm. which belong to the Chlorophyta and Cyanophyta phyla, respectively, are becoming the dominant species, indicating that Baiyangdian Lake had become eutrophic and organic matters tended to increase year by year.

3.3. Relationship between Phytoplankton and Environmental Factors. Nine environment factors, chemical oxygen demand (COD), potassium permanganate index (COD_{Mn}), total nitrogen (TN), total phosphorus (TP), chlorophyll a (Chla), pH, dissolved oxygen (DO), secchi depth (SD), and temperature, were selected in the eight sampling sites. The Pearson correlation analysis showed that no significant correlation was observed between phytoplankton cell density and environmental factors in spring (Table 2). In summer (Table 3), phytoplankton cell density was greatly influenced by DO ($r = 0.813$, $P = 0.014$), Chla ($r = 0.818$, $P = 0.024$), and TP ($r = 0.833$, $P = 0.010$), and there was a positive correlation between them. According to this, the best environment factor combination could not be identified which influenced the space distribution of phytoplankton in Baiyangdian Lake in summer, and DO, Chla, and TP

Table 3: Correlation analysis between phytoplankton density and environmental variables of Baiyangdian Lake in summer.

	T	DO	COD	COD_{Mn}	pH	TN	TP	Chla	SD	Density
T	1									
DO	−0.321	1								
COD	−0.617	0.295	1							
COD_{Mn}	0.087	0.518	0.100	1						
pH	0.241	0.329	−0.070	0.484	1					
TN	−0.814*	0.472	0.555	0.455	0.107	1				
TP	−0.522	0.930**	0.424	0.423	0.267	0.645	1			
Chla	−0.454	0.930**	0.337	0.669	0.313	0.687	0.982**	1		
SD	0.339	−0.089	−0.342	−0.556	−0.364	−0.724*	−0.318	−0.622	1	
Density	−0.245	0.813*	−0.011	0.498	0.630	0.505	0.833*	0.818*	−0.357	1

*Significant correlation at the level of $P < 0.05$; **significant correlation at the level of $P < 0.01$.

were probably the important factors which influenced the space distribution of phytoplankton in Baiyangdian Lake in summer. The main reason for this is that summer is usually the time for algal bloom occurrence in eutrophic lakes. In the sampling Sites 3 and 7, because breeding industry and crop farming were flourishing, the concentration of nutrients in water was high, while algae bloom, DO, and the phytoplankton cell density were low. In the sampling Sites 5 and 1, because the water was open and far from pollution source, the water quality was good and the phytoplankton cell density was low.

Phytoplankton cell density varied greatly from season to season mainly due to the water temperature. Environment factors have different influences on the community structure and species number of phytoplankton in different lakes: SONG Xiao-lan's study on the community structure of phytoplankton in Taihu Lake and Wulihe River found that wind wave and eutrophication degree were the restrictive conditions for growth of phytoplankton species [22]. Jeppesen pointed out that in lakes the density of filtering-feeding fish was a factor closely correlated with the variety and number of phytoplankton [23]. However, other unknown physical, chemical, and biological factors can probably directly or indirectly influence the number of phytoplankton in a certain way, because the growth of phytoplankton is also related to many other factors (e.g., water stability, climate, lake area, lake depth, spatial distribution of organic matter and heavy metals in wetland soils, the community structure, and density of hydrophyte) [24–35]. Therefore, it is difficult to do "dose-effect analysis" of the interaction between the number of phytoplankton and the abovementioned factors. It is suggested that in the future research, more efforts should be made to study lake type, climate characteristics, and surrounding environment, and simulated experiments should be also included to the impact factors on variation of phytoplankton number.

3.4. Eutrophication Degree Assessment. Generally, the excessive growth of phytoplankton is the characterization of eutrophication. Chla, SD, and dominant species are usually regarded as the most important indicators for the assessment of eutrophication degree. In this paper, the index of TSIM and the dominant genus assessment were used to assess the trophic status of Baiyangdian Lake [36]. The index of Carlson nutritional status (TSIM) can elaborately describe the change of water trophic status and can also improve water quality monitoring and assessment. The method is to grade the lake trophic status with numbers from 0 to 100 according to the relation between SD, Chla, and TP. Index under 30 indicates oligotrophic water, index from 30 to 50 indicates mesotrophic water, and index from 50 to 100 indicates eutrophic water. Under the same trophic status, the higher the index is, the more serious the eutrophication is. According to this assessment result (Table 4), water in Baiyangdian Lake is mesotrophic and eutrophicconsider the following:

Table 4: Assessment results of trophic status index of Baiyangdian Lake in 2009.

Item	TSIM	Trophic status
TP	77.00	Eutrophic
Chla	47.6	Mesotrophic
SD	69.24	Eutrophic

$$\begin{aligned} \text{TSIM (Chl } a) &= 10 \times \left(2.46 + \frac{\ln \text{Chl } a}{\ln 2.5}\right), \\ \text{TSIM (TP)} &= 10 \times \left[2.46 + \frac{(6.71 + 1.15 \times \ln \text{TP})}{\ln 2.5}\right], \\ \text{TSIM (SD)} &= 10 \times \left[2.46 + \frac{(3.69 - 1.53 \times \ln \text{SD})}{\ln 2.5}\right]. \end{aligned} \quad (4)$$

In this formula, Chl a is the content of chlorophyll a; SD is secchi depth; TP is total Phosphorus.

According to the dominant genus assessment, dominant phytoplankton genus observed in this survey included not only indicator species for eutrophication, such as *Chlorella* sp., but also indicator species for serious eutrophication, such as *Microcystis incerta* Lemm. and *Chroomonas acuta* Uterm. with indicator species for eutrophication as the dominant species. According to these two methods, Baiyangdian Lake was at eutrophic status, which was consistent with SHEN Hui-tao's [19] research conclusion in the year 2006, in which

the majority of Baiyangdian Lake was at eutrophic status and the eutrophication tended to be aggravated.

4. Conclusions

Eutrophication degree assessing methods can be broadly divided into two types: biological monitoring method and comprehensive indicators method. This study was carried out by combing the dominant genus assessment and the index of TSIM to assess comprehensively the eutrophication degree of Baiyangdian Lake. Generally, the species composition of phytoplankton in Baiyangdian Lake changed significantly, and the density tended to increase compared to the results of comprehensive ecological surveys in recent years. Baiyangdian Lake tended to become seriously eutrophic now. It is necessary to use different approaches to assess eutrophication due to the limitation of these approaches. Moreover, both methods for eutrophication degree assessment in this study are convenient and applicable, and it can be used to assess the impacts of organic matter pollution in other similar cases across China. The finding of this study provides necessary theoretical and data support for the control of eutrophication in Baiyangdian Lake. However, further studies are still needed on the species composition, quantity characteristics, and distribution characteristics of the phytoplankton in Baiyangdian Lake for eutrophication prevention and control and for the conservation of biodiversity.

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Research Article

Evaluation of the Impacts of Land Use on Water Quality: A Case Study in The Chaohu Lake Basin

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It has been widely accepted that there is a close relationship between the land use type and water quality. There have been some researches on this relationship from the perspective of the spatial configuration of land use in recent years. This study aims to analyze the influence of various land use types on the water quality within the Chaohu Lake Basin based on the water quality monitoring data and RS data from 2000 to 2008, with the small watershed as the basic unit of analysis. The results indicated that there was significant negative correlation between forest land and grassland and the water pollution, and the built-up area had negative impacts on the water quality, while the influence of the cultivated land on the water quality was very complex. Besides, the impacts of the landscape diversity on the indicators of water quality within the watershed were also analyzed, the result of which indicated there was a significant negative relationship between them. The results can provide important scientific reference for the local land use optimization and water pollution control and guidance for the formulation of policies to coordinate the exploitation and protection of the water resource.

1. Introduction

The land use within the watershed has great impacts on the water quality of rivers. The water quality of rivers may degrade due to the changes in the land cover patterns within the watershed as human activities increase [1, 2]. Changes in the land cover and land management practices have been regarded as the key influencing factors behind the alteration of the hydrological system, which lead to the change in runoff as well as the water quality [3, 4].

There have been three waves of the research that tried to reveal the effects of the land use and land cover change on the quality of surface water [5, 6]. The researchers have started to study the linkage between land cover and the river water quality in order to investigate the effects of morphological features of watersheds on the turbidity, dissolved oxygen and temperature of the river water since the early 1960s [7]. The second wave of researches on this topic emerged in the 1970s, focusing on the analysis at the watershed scale [8]. The third wave of these studies have started to take advantage of the remote sensing, GIS, and multivariate analysis to explore the influence of the land cover on the suspended sediment, nutrients and ecological integrity of the stream [9–14].

Related research in China started from the 1980s and mainly focused on the role of the macroscopic characteristics of nonpoint source pollution and urban runoff pollution and the quantitative calculation of pollution loading [15–17]. For example, the research carried out by Guo indicated that it is necessary to take into account both the land use and the land cover pattern simultaneously in the study of the impacts of land use on the water quality [18]. Besides, the export coefficient model and RS and GIS techniques have been applied in the study of the effects of the land use change on the nonpoint source pollution load in the upper reach of Yangtze River [19], and the result indicated that the grassland played a dominant role in influencing the TN and TP in Jinsha River, while cultivated land played a key role in other parts of the study area. There was also research about the relationship between land use types and water quality in Xin' anjiang River based on ArcGIS [20]. The results showed that cultivated

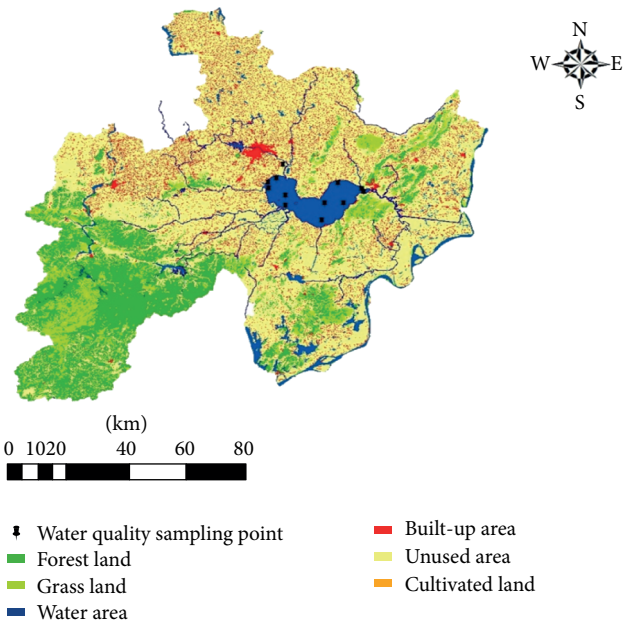

Figure 1: The Chaohu Lake Basin. Water quality points are shown in the figure. Upstream catchment of each water quality sampling point and land use types were delineated.

land, grassland, and forest land had the most significantly important impacts on TN, TP, and fecal coliform bacteria. On the whole, the previous studies in China have focused on only several lakes such as Tai Lake [21] and Dianchi Lake [22]. Besides, these researches have only taken into account the impacts of the composition of land use types within these basins on the water quality. Only few studies have considered the effects of spatial patterns of land use on the water quality, which could provide evidence on landscape planning and land use management. Therefore, The primary objectives of this paper were (1) to describe the water quality change in the Chaohu Lake Basin from 2000 to 2008; (2) to investigate the land use change in the study area; and (3) to identify the relationship between land use change and water quality.

2. Materials and Methods

2.1. Study Area. The Chaohu Lake Basin (Figure 1) is located in the central part of Anhui Province and between 117°16′46′16″–117°51′54′ 51″E, 30°43′28′ 43″–31°25′28′ 25″N. The Chaohu Lake belongs to the drainage system in the lower reaches of the Yangtze River, and it is the fifth largest freshwater lake in China, with a total watershed area of 13350 km^2. The total annual inflow from 33 rivers is 4.8×10^9 m^3 $year^{-1}$ and the total outflow is 3.4×10^9 m^3 $year^{-1}$. A large portion of the inflow is from Nanfei River, Hangbu River, and Yuxi River. The average annual temperature in the Chaohu Lake Basin ranges between 15°C and 16°C, with a mean annual rainfall of 1100 mm. The Chaohu Lake Basin is one of the most densely populated regions in Anhui Province [23], with a population of more than 9.65 million. The Chaohu Lake Basin also plays an important role in the local economic development, accounting for 24.65% of GDP in Anhui Province.

2.2. Water Quality Situation and Data Resources. The Chaohu Lake Basin was known as the land of fish and rice in the early stage when the local ecological environment was very good and the water quality of Chaohu Lake used to be very fine. However, the hydrological conditions and downstream ecosystems of this lake have been altered since the establishment of Chaohu Dam on Yuxi River [24, 25], with the amount of annual water exchange volume decreasing from 13.6×10^8 m^3 to 1.6×10^8 m^3 [26] and the average annual water level falling from 4.3 m to 2.9 m. Besides, with the rapid development of local economy and social activity, the wetlands in the Chaohu Lake Basin were reclaimed or occupied [27]. And water quality of Chaohu Lake has continuously deteriorated due to the large amount of pollution discharge from the local industry, agriculture, and daily life since the late 1970s [28]. Along with that is the outbreak of water bloom, which has aroused great concern of the government. The Chaohu Lake has reached the eutrophic state, with the high concentration of nutrient salts and rapid growth of algae.

The water quality of Chaohu Lake has consciously improved to some degree. In 1997, the Ministry of Environmental Protection of China promulgated the water quality of the five biggest fresh water lakes, among which the situation of Chaohu Lake was the worst. During 1996–1999, the data from water quality monitoring points around Chaohu Lake showed that the percentage of water exceeding the level V had decreased from 80% to 60%. Since 2000, water quality of the main tributaries of Chaohu Lake, including Nanfei River, Shiwuli River, Pai River, and Shuangqiao River, was always exceeding the level V, with ammonia nitrogen as the key pollutant. But the water quality of other tributaries was better, generally fluctuating between level III and level IV (Table 1).

In 2003, Chaohu Lake reached the meso-eutrophication the whole, with the total phosphorus (TP) and total nitrogen (TN) as the main pollutants. The water quality of Chaohu Lake improved slightly in 2005, reaching the meso-eutrophic state on the whole. The mean value of CODmn and TN succeeded in achieving the goal of the tenth Five-year Plan in Chaohu Lake, but the mean value of TP failed. For the moment, Chaohu Lake is known as one of the three most polluted freshwater lakes in China [29]. Since Chaohu Lake serves as the primary drinking water source of Hefei City, it has been ranked as the key lake to be managed.

In this study, the Chaohu Lake Basin was divided into nine watersheds according to the local river systems and the monitoring points were set up over there (Figure 1). There are many variables of water quality available from these monitoring points, including pellucidity, TP, TN, DO, BOD5, and CODcr. But we only selected TP, TN, DO, NH3-N, and CODmn measured in every month from 2000 to 2008. Other water quality variables were also important; we chose these five not because they were more important than others in Chaohu Lake, but because many researches had chosen them, and these water quality variables had the complete data. The average annual values of these variables were also used in view

Table 1: Water quality of Chaohu Lake between 2000 and 2007.

River	2000	2001	2002	2003	2004	2005	2006	2007
Nanfei River	Bad V	Bad V	Bad V	Bad V	Bad V	V	V	Bad V
Shiwuli River	Bad V	Bad V	Bad V	Bad V	Bad V	Bad V	Bad V	Bad V
Pai River	Bad V	Bad V	Bad V	Bad V	Bad V	Bad V	Bad V	Bad V
Hangbu River	IV	II	III	III	IV	IV	III	IV
Baishitian River	IV	III	IV	III	IV	III	IV	IV
Zhao River	III	III	III	III	IV	IV	IV	IV
Tuogao River	III	III	IV	III	III	IV	III	III
Yuxi River	IV	III	III	IV	II	IV	III	IV
Shuangqiao River	Bad V	Bad V	Bad V	Bad V	Bad V	Bad V	Bad V	Bad V

Table 2: Descriptive statistics for the study area, including water quality and land use characteristics.

Categories	Variable	Obs	Mean	Min	Max	S.D.
Water quality	CODmn (mg/L)	81	5.18	2.7	7.8	1.09
	NH3–N (mg/L)	81	0.54	0.00	1.61	0.39
	TP (mg/L)	81	0.19	0.07	0.57	0.19
	TN (mg/L)	81	2.23	1.04	6.48	1.11
	DO (mg/L)	81	8.22	6.69	9.65	0.58
Land use	Cultivated land (%)	81	0.47	0.32	0.80	0.12
	Forest land (%)	81	0.02	0.00	0.06	0.02
	Grassland (%)	81	0.02	0.00	0.67	0.02
	Water area (%)	81	0.36	0.00	0.58	0.16
	Built-up area (%)	81	0.11	0.03	0.33	0.08
Landscape metrics	Shannon	81	2.83	1.76	3.85	0.50

S.D.: standard deviation.

of the seasonal variations of algal species and water quality in Chaohu Lake [30].

2.3. Land Use Data. The land use data, which was extracted from the Lantsat TM images (from 2000 to 2008), was provided by the Data Center of the Chinese Academy of Sciences. There are six kinds of land use types, that is, the cultivated land, forest land, grassland, water area, built-up area and unused land. The Landsat ETM images in 2000 and 2005 were interpreted at a scale of 1 : 100,000 and the overall interpretation accuracy of the land use categories reached 92.7% according to the field survey and random sampling check conducted by Data Center of the Chinese Acadamy of Sciences (CAS) [31, 32]. The watershed boundaries were delimitated based on the DEM data with the "automatic delineation utility" in BASINS. The Chaohu Lake Basin was divided into nine small watersheds; then we used GIS tools to calculate the area of each land use type within each subwatershed. Based on that we got the proportion of each land use area within each sub-watershed.

2.4. Spatial Patterns of Land Use. Landscape pattern change is mainly caused by the change in land cover and land use change [33]. The landscape ecologists and other researchers have developed numerous metrics to investigate the effects of the landscape pattern on the ecological processes [34]. In view of the multicollinearity among metrics and the erratic behaviors of some metrics across scales, we selected Shannon's diversity index (SHDI) as the indicator of landscape metric use in this study. SHDI indicates the patch diversity in a landscape based on the information theory, and it is calculated with the following form:

$$\mathrm{SHDI} = -\sum_{i=1}^{m} (\mathrm{pi} \ln \mathrm{pi}), \quad (1)$$

where pi is the proportion of the landscape occupied by land use type i and m is the number of land use type present in the landscape.

The SHDI is a sensitive indicator to analyze the diversity and heterogeneity of the same landscape in different times. The big value of SHDI means that the land use pattern is various and the degree of fragmentation is high. We calculated the percentages of the five land use types and the SHDI in these nine sub-watersheds and then analyzed the relationship between the SHDI and the indicators of water quality.

3. Results and Discussions

3.1. Descriptive Statistics of Measures. As showed in Table 2, the average CODmn concentration in 2000 and 2008 was 5.18 mg/L, with the concentration of NH3-N, TP, TN, and

Table 3: LUCC of the Chaohu Lake Basin between 1995 and 2005.

Year	Statistics variable	Cultivated land	Forest land	Grassland	Water area	Built-up land	Nonuse land
1995	Area (km^2)	17153.11	5781.29	1817.64	2009.12	2165.91	1.32
	Proportion (%)	59.30%	19.98%	6.28%	6.95%	7.49%	0.00%
2000	Area (km^2)	16960.22	5770.15	1817.34	2015.09	2364.22	1.32
	Proportion (%)	58.63%	19.95%	6.28%	6.97%	8.17%	0.00%
2005	Area (km^2)	16850.80	5764.96	1816.65	2019.72	2474.93	1.32
	Proportion (%)	58.25%	19.93%	6.28%	6.98%	8.56%	0.00%

DO being 0.54 mg/L, 0.19 mg/L, 2.23 mg/L, and 8.22 mg/L, respectively. The standard deviation of indicators of water quality was generally very small.

The most important indicators of the water quality are the CODmn and TN, the average concentration of which reached 5.18 mg/L and 2.23 mg/L, respectively. The two indicators also vary greatly among the sub-watersheds, with their standard deviations being 1.09 and 1.11, respectively. There are main cultivated land and water areas in the Chaohu Lake Basin, accounting for 47% and 36% of the total area, respectively. Besides, the cultivated land and water area also vary most greatly among small watersheds, with their standard deviations reaching 0.12 and 0.16, respectively.

3.2. Water Quality Change in the Study Area. According to the data from water quality monitoring points, water quality change in the study area was shown in Figure 2. In general, the water quality has improved from 2000 to 2008. Change trend of NH3-N and TP was small. And the change trend of DO was upward. The rest has changed a lot among this period; however, the beginning value was close to the finishing value.

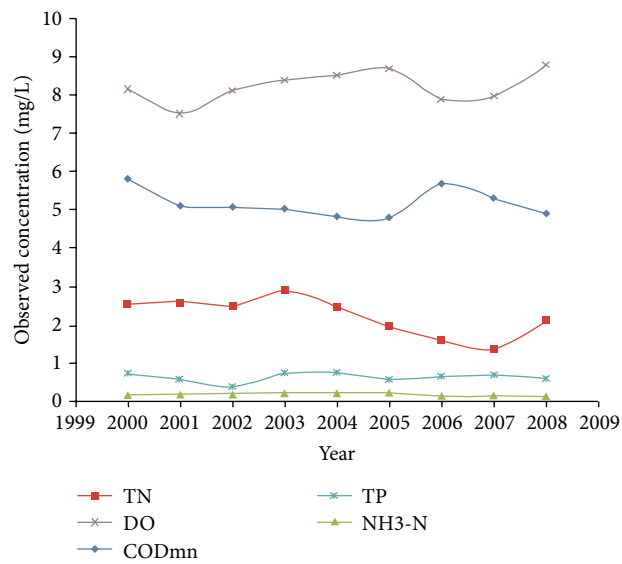

Figure 2: Water quality change from 2000 to 2008 in the study area.

3.3. Land Use Change. The overall state of land use in Chaohu Lake Basin was extracted based on the land use images (Table 3). The Chaohu Lake Basin is dominated by agriculture, and the cultivated land accounted for almost 60 percentage of the total area from 1995 to 2005. The total area of cultivated area has changed from 17153 km^2 in 1995 to 16850 km^2 in 2005, decreasing by about 1.7 percentage, while the area of built-up land increased by about 14 percentage. The urban expansion is the main driving factor of the decrease of cultivated land, and this kind of conversion would change the wetland soil which served as natural sinks and filtration system [35, 36]. The forest land accounted for approximately 20% of the total land area in the Chaohu Lake Basin, which was very stable in these years. The proportion of the grasslandand and water area in the Chaohu lake Basin was about 6.28% and less than 7%, respectively, both of which were very stable during 1995 and 2005. In this study, The unused land was not included in this study since it accounted for almost 0% of the total area.

3.4. Relationship between Land Use and Water Quality. The land use data and water quality monitoring data were analyzed by Stata. The model we used is an econometric model based on the research of four lakes watersheds in Hanyang district [18]:

$$\mathrm{NPS} = \exp\left(\beta 1 \times \mathrm{land}1 + \beta 2 \times \mathrm{land}2 + \cdots \beta i \times \mathrm{land}i\right), \quad (2)$$

where NPS means the water quality variables in the study area, α is a constant, and β means the correlation between land use area (%) and water quality variables. When $\beta i > 0$, it means that land use type i has a positive effect on the indicators of water quality. If $\beta i < 0$, it means that land use type i has a negative effect on the indicators of water quality.

Since we got the data, the panel data has been used so as to comprehensively and completely reflect the relationship between the water quality and land use types in the Chaohu Lake Basin. Our panel data analysis was about 9 small watersheds in 2000–2008. Given the robustness and accuracy of the econometric analysis, we used both the fixed effect model and the random effect model so as to make a comparison, and finally the fixed effect model was selected according to the result of the Hausman test. Then we got the forms to

Table 4: The correlation coefficients between different land uses and water quality variables at the scale of the whole watershed.

Variables (%)	ln (TN) coef.	ln (TP) coef.	ln (CODmn) coef.	ln (NH3–N) coef.	ln (DO) coef.
Cultivated land	−0.46	−0.42	−0.08	0.31	0.06
Forest land	-4.11^{**}	-9.59^{***}	-5.47^{***}	-9.26^{***}	0.31
Grassland	-6.52^{***}	-7.87^{***}	-2.93^{***}	-8.83^{***}	0.54
Water area	−0.30	0.26	0.29^{*}	1.22^{**}	0.0029
Built-up area	0.86	1.54^{**}	0.70^{**}	2.60^{***}	-0.27^{*}
Constant	1.20^{***}	-1.44^{***}	1.69^{***}	-1.04^{***}	2.08^{***}
R-squared	0.49	0.75	0.80	0.69	0.36

$^{***}P < 0.01$, $^{**}P < 0.05$, $^{*}P < 0.1$.

Table 5: The correlation coefficients between landscape matrix and water quality variables at the scale of the whole watershed.

Variables (%)	ln (TN) coef.	ln (TP) coef.	ln (CODmn) coef.	ln (NH3–N) coef.	ln (DO) coef.
Shannon	-0.410^{***}	-0.633^{***}	-0.296^{***}	-0.650^{***}	0.0435^{***}
Constant	1.866^{***}	−0.00560	2.463^{***}	1.232^{***}	1.981^{***}

$^{***}P < 0.01$, $^{**}P < 0.05$, $^{*}P < 0.1$.

describe the relationship between land use types and water quality according to the analysis results of Stata (Table 4):

$$\begin{aligned}
\ln(\mathrm{TN}) &= -0.47C - 4.11F - 6.52G - 0.3W \\
&\quad + 0.86B + 1.2, \\
\ln(\mathrm{TP}) &= -0.42C - 9.59F - 7.87G - 0.26W \\
&\quad + 1.54B - 1.44, \\
\ln(\mathrm{CODmn}) &= -0.08C - 5.47F - 2.93G + 0.29W \\
&\quad + 0.7B + 1.69, \\
\ln(\mathrm{NH3-N}) &= 0.31C - 9.26F - 8.83G + 1.22W \\
&\quad + 2.6B - 1.04, \\
\ln(\mathrm{DO}) &= 0.06C + 0.31F + 0.54G + 0.0029W \\
&\quad - 0.27B + 2.08,
\end{aligned} \tag{3}$$

where C is cultivated land area (%), F is forest area (%), G is grassland area (%), W is water area (%), B is built-up area (%).

There was a positive relationship between the cultivated land area (%) and the concentration of NH3-N and DO. This is mainly due to the developed agriculture in the Chaohu Lake Basin and the emission of NH3-N from the exposure of soil surface resulting from the agricultural practices and the application of chemical fertilizers [37]. Besides, the concentrations of TP and TN are negatively related with the cultivated land area (%). On the one hand, the fertilizers used in the cultivated land will get into the runoff and flow into the river and ultimately pollute the river water. On the other hand, the vegetation in the surface soil of the cultivated land can absorb, retain the pollutants. As a result, the cultivated land plays a complicated role in influencing the water quality in the Chaohu Lake Basin.

The forest land and grassland both have significant positive influence on the water quality. The areas of the forest land and grassland were negatively related to TP, TN, NH3-N, and the CODmn and was positively related to DO. Many researches had shown similar results [15, 37].The significant negative relationship between the forest land and grassland area and TP, TN, CODmn, and NH3-N indicates that the the forest land and grassland played a key role in reducing the nitrogen pollutants and phosphorus pollutants and played a controlling role in regrating the water quality. The vegetation and soil in the forest land and grassland can effectively reduce the nutrient salts brought into the river by the surface runoff since they play an important role in reducing the surface runoff, conserving the water and soil, and absorbing the pollutants. Therefore, the increase of the forest land and grassland area will reduce the concentration of TP, TN, and oxygen-consuming substances, increase the concentration of dissolved oxygen, and consequently improve the water quality.

The built-up area played a negative role in influencing the water quality on the whole. The built-up area was positively related to TP, TN, NH3-N, and CODmn and was negatively related to DO, indicating that the increase of the built-up area tends to degrade the water quality. Mouri et al. found similar relationship between concentration of TN and the area of built-up area [38], which is in agreement with that of Amiri and Nakane, who analyzed the relationship between TN, DO, NH3-N, and built-up area [37]. This could be the result of the increase of the nutrient concentration. The dense population density and economic activities both concentrated in the built-up area, which leads to very serious pollution. Besides, there is a lot of impermeable surface in the built-up area, which will contribute to the increase of surface runoff and may increase the concentration of nutrient salts in the river and consequently degrade the water quality within the watershed. There was a positive relationship between

the built-up area (%) and TP and CODmn. The increase of built-up area was the result of transformation from the land with natural vegetation that could prevent the soil erosion. Since the vegetation can protect the soil from raindrops and tends to slow down the movement of runoff and allows the excessive surface water to infiltrate into soil, the conversion of the land with vegetation into built-up area will aggravate the soil erosion and consequently increase the amount of TP into the runoff.

3.5. Relationship between Spatial Patterns of Land Use and Water Quality. The relationship between SHDI and water quality was revealed in Table 5. SHDI had a negative relationship with NH3-N, TP, CODmn, and TN. According to the results of Stata analysis, we can know that the relationship between SHDI to CODmn, TP, TN, and NH3-N was very significant. The significant negative relationship mean that the landscape diversity in the Chaohu Lake Basin is closely related to the water quality. The higher the SHDI is, the greater the diversity landscape is and the slighter the deterioration of water quality is. As the landscape diversity increases, the landscape heterogeneity increases and consequently makes the patches of each landscape type more evenly distributed.

4. Conclusion

Studying the relationship between the proportion of land use types and water quality in the Chaohu Lake Basin in this study indicated that built-up land was generally positively related to the indicators of water quality, and the forest land and grass land and water area were negatively related with the water quality variables, while the influence of the cultivated land on the water quality was very complex. Additionally, the built-up land, grassland, and forest land had significant influence on some indicators of water quality. The regression result of the landscape indicators and the indicators of water quality suggested that SHDI was negatively related to most of the water quality variables, indicating that the increase of landscape diversity can contribute to the improvement of water quality.

According the result mentioned previously and the current conditions of the local water quality in the Chaohu Lake Basin, it is necessary to increase the area of forest land, grassland, and water area in the local land use planning. Since the forest land is more closely related to the local water quality, it is specially important to increase the area of forest land. Besides, the growth rate of the urban land should be slowed down under the condition of guaranteeing the minimum land area needed by the city development within the watershed. In addition, it is necessary to increase the landscape diversity because the greater the landscape diversity is, the more evenly patches of each kind are distributed and the more the water pollution will be alleviated.

The results of these studies can provide scientific reference for the local land use optimization and water pollution control and assist the formulation of policies for coordinating the water resource exploitation and protection. In addition, the previous researches have indicated that their landscape diversity has impacts on the water quality within the watershed, but it is still necessary to add some other ecological indicators and analyze their influence on the water quality. In particular, we should expand the method of analysing the relationship between land use and water quality but not the simple regression. Our study focuses on the effect of land use types and landscape patterns on water quality in the study area. However, there are many factors related to water quality, such as the climate, precipitation, and density of population. In the future work, we will refine the method and indicators to deeply reveal the reasons causing water quality change within a watershed.

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Research Article

Installation of an Artificial Vegetating Island in Oligomesotrophic Lake Paro, Korea

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After cut off of inflowing water, Lake Paro, an oligomesotrophic lake lost littoral zone, an important region for the aquatic ecosystem. For the first step of restoration, the artificial vegetation island was installed. The concentration of nutrients in lake water was not sufficient for the growth of macrophyte as total phosphate was ranged from 58 to 83 μg L^{-1}. In order to overcome this problem, the hydrophobic substratum for bacterial attachment was selected as buoyant mat material of the artificial vegetation island. In this medium, total phosphate and total nitrogen were ranged from 190 to 1,060 μg L^{-1} and from 4.9 to 9.1 mg L^{-1}, respectively. These concentrations were high enough for macrophytes growth. After launching 1,800 m^2 of AVI in Lake Paro, the macrophytes, *Iris pseudoacorus* and *Iris ensata*, grew well after five years of launching without the addition of fertilizer. Furthermore, fishes were plentiful under the artificial vegetation island, and ducks were observed on the artificial vegetation island. Bacteria using sunlight as energy source and self-designed ecotechnology can be used as an alternative method for the restoration of disturbed littoral zone in oligo-mesotrophic lakes.

1. Introduction

Following construction of the Keumgangsan Dam in Democratic People's Republic of Korea in 2001, about 60% of main inflow into Lake Paro was cut off. To protect from possible breakage of the Keumgangsan Dam, a defensive dam, named Peace Dam, was constructed in 2005 near the upper part of Lake Paro. The water level in Lake Paro was lowered from 181 m to 150 m following the construction of the dam (Figure 1). After the decline in water level, this oligomesotrophic Lake Paro experienced problems related to fish habitat such as degradation of the littoral zone and lose of spawning and refuge areas. These problems resulted in a reduction in fish populations such as the common carp (*Cyprinus carpio*) and mandarin fish (*Siniperca scherzeri*) [1].

Restoration of littoral habitat in Lake Paro is vital to promoting the sustainability of the Lake Paro ecosystem. However, the littoral zone at Lake Paro cannot be naturally revegetated easily because of these steep slopes and frequent fluctuations in water level. The emergent macrophytes in the littoral zone are affected by local conditions such as water logged soils, partial submergence and nutrient availability [2]. In some cases, natural vegetation islands comprised of submerged, and emergent macrophytes can deteriorate the fishery and lake water quality [3].

Artificial vegetation islands (AVIs) can be ameliorate some of the problems associated with degraded littoral zones where is seriously degraded such as in Lake Paro [4, 5]. The broad concept behind AVIs is to mimic natural floating islands which are common in some lakes across the world. AVIs often consist of macrophytes and buoyant material. AVIs should include adequate biomass in the rhizosphere zone because of the important role of providing habitat for fish spawning [6] and refuge areas for zooplankton [7, 8] and invertebrates [9]. Specially, the effective design of the AVIs in oligo-mesotrophic lakes must include the suitable materials to allow for the accumulation of nutrient supply for macrophytes growth in the AVI, and the biotic components of the AVI are essential [10]. Besides, the materials of the AVI have to attach and support bacteria.

Bacteria play an important role in the accumulation of nutrients [11], and aggregated or attached bacteria have been

Figure 1: Degraded littoral zone in Lake Paro, Korea. Water level was lowered for lake construction (August, 2002).

shown to have an especially effective role in biofilm formation [12]. The bacteria in biofilms are embedded in exopolysaccharides [13], and form microcolonies on hydrophobic substances [14].

Thus, the material used in AVIs in oilgomesotrophic lakes should has a role not only sustaining macrophytes but also supporting an active biofilm community.

As an alternative littoral zone, we installed the AVI in Lake Paro and observed the growth of macrophytes and ecological changes.

2. Materials and Methods

2.1. Buoyant Mat Material. According to previous research [15], rubberized coconut fiber was selected for buoyant mat material. After 1 month of submerging the mat made of rubberized coconut fiber into distilled water, we did not find evidence of significant release of nutrients from the mat (data not shown). Before installing the AVI, in order to confirm the accumulation of nitrogen and phosphorus in the mat, three pieces of rubberized coconut fiber (1 m by 1 m at 0.1 m depth) were submerged in the center of Lake Paro for 1 week. After retrieving the mats, the interstitial water was sampled by gravity and analyzed for total nitrogen (TN) and total phosphorus (TP) concentrations using standard methods [16].

2.2. Installation of the Artificial Vegetation Island. After the effectiveness of buoyant mat material as a nutrient concentrator was confirmed, 1,800 m^2 of AVI was installed in Lake Paro (38°06′51″N, 127°49′25″E) (Figure 2). Two species of macrophytes, *Iris pseudoacorus* and *Iris ensata*, were planted at a density of 9 shoots m^{-2} in the AVI during August 2003. Ten shoots of each macrophytes species were randomly collected and the total length and root length were measured in the laboratory during May 2004.

2.3. Samples Preparation and Water Chemistry. Interstitial water samples from AVI were extracted using sterilized syringes, and lake water samples were collected aseptically with 1L plastic bottle as a control. These samples were transported in cold condition (4°C) and immediately analyzed.

Table 1: TP and TN concentrations in interstitial water of buoyant mat material after 7 days of submergence in lake water (May, 2003).

	T-P (mg P L^{-1})	T-N (mg N L^{-1})
Lake water	0.004	1.9
Interstitial water by gravity force	0.4	38.5
Interstitial water by squeezing	5.1	489.0

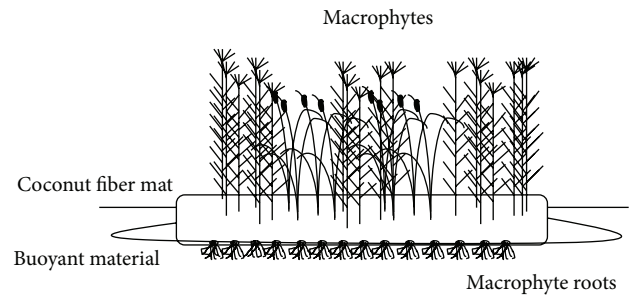

Figure 2: Diagram of AVI launched at Lake Paro. 1,800 m^2 of AVI was installed on August 2003.

Total nitrogen (TN) and total phosphate (TP) were analyzed by using standards methods [16]. Measurements were conducted in triplicate.

2.4. Bacterial Abundance. For the direct counting of bacterial abundance, samples were filtered through black polycarbonate membrane filter (Nucleopore, pore size 0.2 μm, dia 25 mm) and stained with acridine orange (100 μL of 1 g/L aqueous stock solution of acridine orange) [17]. Bacterial abundance was counted with an epifluorescent microscope (Olympus BX60, Japan).

3. Results

3.1. Nutrient Accumulation in the Buoyant Mat Material. After one week of submergence in Lake Paro, the TN and TP concentrations in the interstitial water were much higher than initial concentrations. Interstitial water samples collected by gravity force and squeezing had TP concentrations ranging from 0.4 mg L^{-1} to 5.1 mg L^{-1} and TN concentrations from 38.5 mg L^{-1} to 489.0 mg L^{-1}. These concentrations of collected samples by gravity force were about 100 times and 20 times higher than the concentrations in the lake water, respectively. Water samples collected by squeezing water had approximately 1,200 times the lake water column concentrations of TP and 50 times higher than lake water column TN concentrations (Table 1).

3.2. TP and TN in the AVI and Lake Water. Variations in interstitial TP concentrations in the AVI and lake water are shown in Figure 3(a). During the macrophytes growing period, TP concentration in lake water and interstitial water of AVI ranged from 53 to 83 μg L^{-1} and from 490 to 1,060 μg L^{-1}, respectively. TP concentrations in the AVI were 17 times higher than those in lake water. TN concentrations in the AVI interstitial water ranged from 4.9 to 9.1 mg L^{-1},

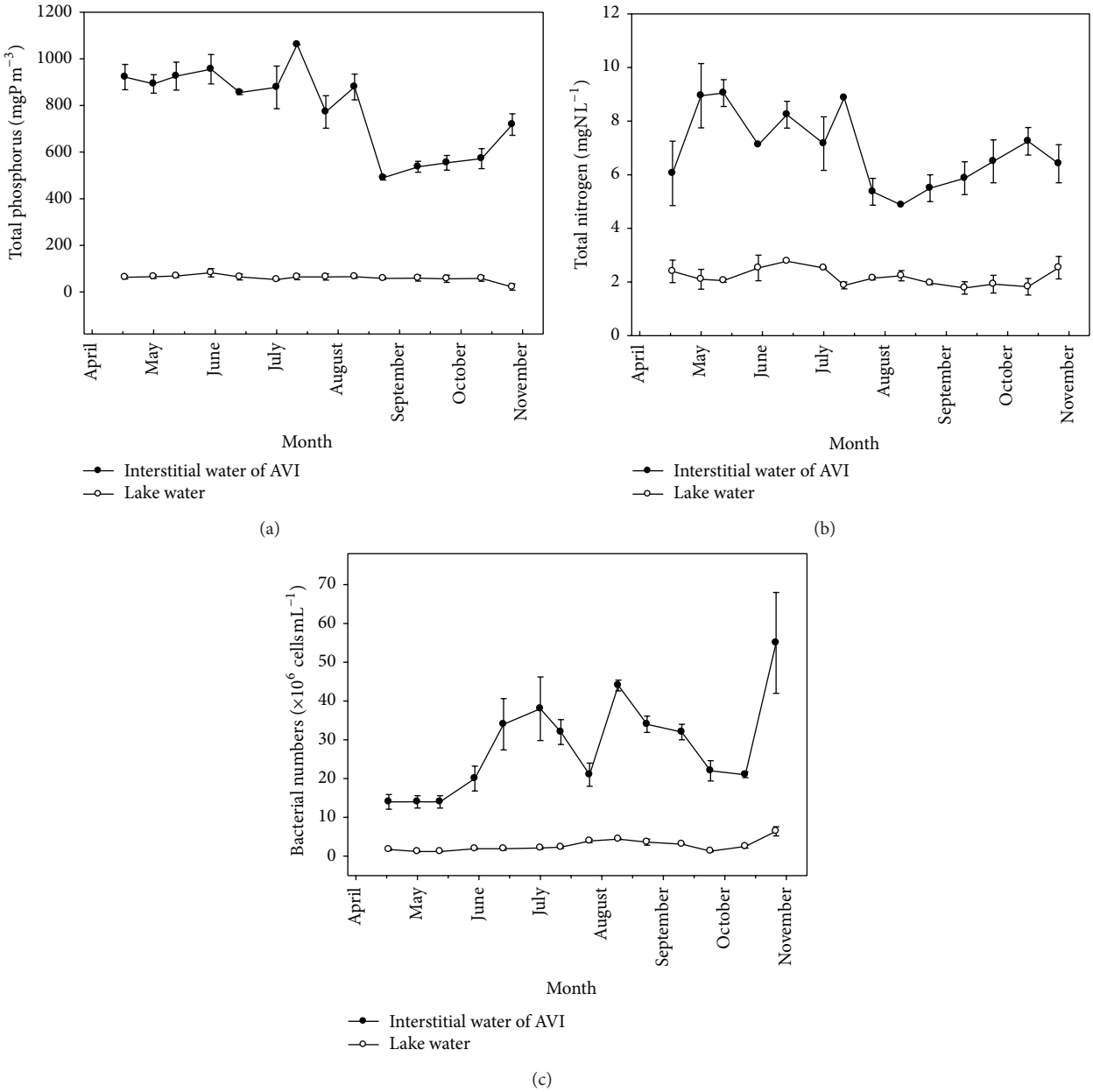

Figure 3: Variations in TP (a), TN (b), and bacterial numbers (c) in AVI interstitial water and ambient lake water in Lake Paro in 2004.

approximately 3.5 times higher than those of lake water (Figure 3(b)).

3.3. Bacterial Abundance and Macrophytes Growth. Variations in total bacterial numbers in the interstitial water of AVI and the lake water are shown in Figure 3(c). In interstitial water of AVI and the lake water, the bacteria abundance ranged from 1.4×10^7 to 5.6×10^7 cells mL^{-1} and from 1.2×10^6 to 6.5×10^6 cells mL^{-1}, respectively. Bacterial abundance in the interstitial water was always higher than the bacterial abundance in lake water, about 5 to 18 times higher.

Immediately following the launching of the AVI in Lake Paro, macrophyte growth was prolific. During late summer and autumn the macrophytes grew well and in the spring of 2004, healthy new shoots were produced by the 1-year-old macrophytes in the AVI (Figure 4). The length of roots of both macrophytes ranged from 50 to 100 cm, and the lengths of leaves ranged from 60–70 cm.

4. Discussion

The artificial vegetation island (AVI) in Lake Paro can be regarded as unique wetland ecosystem installed within an oligomesotrophic lake. AVIs can be effective for wetland restoration in places where the littoral zone has been severely degraded. Similar to natural floating islands, nutrient concentrations within the AVI are often higher than the ambient lake water (control) [18].

Figure 4: Iris flowering on AVI in Lake Paro (*Iris pseudoacorus* (left), *Iris ensata* (right), and May, 2004 (a), June, 2005 (b), May, 2008 (c)).

In Lake Paro, the AVI installation was selected with a focus on the macrophyte community as a key functional role [19]. The Macrophytes absorb inorganic and organic nutrients, provide habitat for beneficial microorganisms, and promote the breakdown of organic matter by oxidation in the root system [20].

In the AVI, the central component of the macrophytes community is the submerged root system. Some macrophyte roots spread through the mat material (coconut rubberized fiber) to obtain nutrients, and other roots hang directly in the water column. These roots showed the proliferation of bacteria. In our study, we did not analyze the physiological functions of the macrophyte roots, but we assume that their physiological and ecological functions are similar to the functions of submerged macrophytes, which include providing refuge areas for pelagic zooplankton [21], for eggs of fishes [6] and for invertebrates [8].

The AVI is composed of artificial buoyant material. In eutrophic and shallow lakes, the mat material may not be important for macrophytes growth, because the nutrients for macrophytes growth are supplied from the lake water or sediment. However, in oligotrophic lakes, the mat material might play a role of nutrient supply. We carried out the preliminary experiment for evaluating the different mat materials, with hydrophobic rubberized coconut fiber selected as most suitable [15].

The surface of rubberized coconut fiber is hydrophobic, so this substratum of the AVI can be used for bacterial adhesion. Desorption was considerably greater on the hydrophilic substratum, whereas desorption from the hydrophobic media was almost negligible [22]. Moreover, bacterial adhesion to the hydrophobic surfaces appears to be rapid, and binding may be stronger than those on hydrophilic surfaces [23]. Therefore, interstitial water of rubberized coconut fiber mat contains high concentration of nutrients, even though the nutrient concentrations in the lake water are low.

We did not measure the biomass of macrophytes, *Iris pseudoacorus* and *Iris ensata*. It is likely a large portion of nitrogen, and phosphorus accumulated by bacteria could be transported into the macrophytes biomass. The phosphorus

Figure 5: Fish eggs attached to macrophyte roots under the AVI (on June, 2007).

and nitrogen contents of *Iris*, grown at nutrients rich pond, have been measured to be 0.2% and 2.4%, respectively. Assuming a productivity of 5 g dry weight m^{-2} day^{-1}, a daily uptake of 0.01 g P m^{-2} and 0.12 g N m^{-2} can be estimated [24]. Direct uptake of nutrients by macrophytes from bulk lake water might be limited, because of the low nutrient concentrations. Therefore, the nutrients for macrophytes growth are from the mat. This hypothesis is supported by the healthy appearance of macrophytes leaves and active flowering, suggesting that the nutrient supply from the mat was sufficient [25].

The bacterial abundances at AVI water were about 50 times higher, and the bacteria cell size was larger than those found in ambient lake water. This indicates that the AVI supports a newly created bacterial ecosystem which is different from the ambient lake water ecosystem.

Another ecological function of the AVI is supporting a new biotope. We observed birds and fishes thriving after the installation of the AVI, and one study showed that an AVI consisting of *Phragmites japonica* and other macrophytes supported 43 species of insects after 2 years of installation [5].

In this study, we did not measure the diffusion of nutrients from AVI to lake water. The higher nutrients concentrations in the AVI interstitial water result from dilution, and a detailed nutrient budget would be needed to quantify all the processes involved in nutrient release and retention in the AVI. This could be the focus of future research.

5. Conclusion

This study focused on restoring the functions of the littoral zone in an oligomesotrophic lake by installing an AVI. Lake Paro is an oligomesotrophic lake with insufficient nutrient concentrations to support vigorous growth of macrophytes. However, any addition of fertilizer to the AVI in Lake Paro is not desired and is prohibited. We found that following the AVI installation, the bacteria community in the lake adhere to the hydrophobic substratum. Biofilms bacteria accumulated TP and TN from the lake water. The roots of macrophytes beneath the AVI provided refuge areas for eggs of fishes (Figure 5), zooplankton, and invertebrates and consequently attracted planktivorous and piscivorous fishes. Birds were also attracted to the AVI. The AVI macrophytes community has continued to grow without any special treatment; therefore, this system may permanently function only if there is no mechanical damage.

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Research Article

Estimation Model of Soil Freeze-Thaw Erosion in Silingco Watershed Wetland of Northern Tibet

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The freeze-thaw (FT) erosion is a type of soil erosion like water erosion and wind erosion. Limited by many factors, the grading evaluation of soil FT erosion quantities is not well studied. Based on the comprehensive analysis of the evaluation indices of soil FT erosion, we for the first time utilized the sensitivity of microwave remote sensing technology to soil moisture for identification of FT state. We established an estimation model suitable to evaluate the soil FT erosion quantity in Silingco watershed wetland of Northern Tibet using weighted summation method of six impact factors including the annual FT cycle days, average diurnal FT phase-changed water content, average annual precipitation, slope, aspect, and vegetation coverage. Finally, with the support of GIS, we classified soil FT erosion quantity in Silingco watershed wetland. The results showed that soil FT erosion are distributed in broad areas of Silingco watershed wetland. Different soil FT erosions with different intensities have evidently different spatial and geographical distributions.

1. Introduction

The annual freeze-thaw cycle of soils in cold regions is an important aspect of agricultural and ecological environments because this cycle may significantly impact soil physical properties [1, 2]. On the hydrological side, the precipitation occurring in early spring often results in high rates of runoff and erosion on frozen soil due to its poor infiltration capacity. Information about the depth of frost penetration in the subsurface is essential to avoid damage to roadbeds, structural foundations, sewers, and pipelines [3]. Soil freeze-thaw (FT) erosion is the gravity caused removing, migrating, and accumulating process of soil or rocks destructed mechanically by the differential expansion and contraction of different minerals induced by volume alteration of water at different phase due to temperature changes [4]. The main reason is that the moisture transfers to the freezing front during the freezing period, which results in water content increasing in places of shallow slope, and then frost heaving occurs under subzero temperature. The association and arrangement among soil particles was changed by frost heaving, and then the mechanical properties of the soil changed [5]. In the melting period of spring, the frozen layers in shallow slope thawed influenced by kinds of factors such as precipitation and increasing temperature. The melting water was hampered by unfrozen layer under them during their infiltration downward, results in the water content increasing rapidly in the layer between melt layer and frozen layer, and reaching saturation or super saturation state, then the effective stress within the slope reduced, the partial or whole of the shallow slope slide down along the water saturation layer under gravity [6]. It occurs mostly in the cold regions with high latitude and high altitude [7]. Recently with the onset of global warming there has been an increasing concern of greenhouse gas release from permafrost, and therefore interest in monitoring freeze-thaw erosion dynamics is increasing [8].

Soil freezing creates a situation where water and ice coexist in a near thermodynamic equilibrium. When soil temperature gradually drops below the freezing point, a portion of water in the soil turns into ice. The remainder of the water exists as adsorbed films around soil particles,

in pores with sufficiently small diameters and in crevices between soil particles [9]. As the temperature continues to fall, more water becomes frozen, leaving residual liquid water in progressively thinner adsorbed films, smaller pores, and crevices. Consequently, a small amount of soil water may remain in a liquid state at temperatures well below the freezing point of water [10, 11]. In addition, an upward migration of soil water takes place as the frost penetrates the ground surface [12], a consequence of the thermal gradient causing capillary flow from higher (deeper soil) to lower temperatures.

Present studies mainly focus on the impact of freeze-thaw erosion on the soil bulk density, permeability, moisture content, and stability [13–16], but about, estimation model of freeze-thaw erosion in large scale are still insufficient. The FT model includes six factors, annual FT cycle days, average diurnal phase-changed water content, annual average precipitation, slope and vegetation coverage has interest characteristics under the complex effects of freeze-thaw cycles, that factors take into account the environmental impact of soil properties, rainfall, topography, and ecology. Annual FT cycle days referred to freeze-thaw frequency; the freeze-thaw frequency at a particular station is the annual number of times the recorded temperature falls below the point of effective freeze following a period when the temperature was at or above the point of effective thaw. Phase change of soil water is an important sign of the development of soil freeze-thaw process causing the changing of liquid water content of soil. Wegmüller measured the brightness temperature change during several freeze/thaw circles using ground-based microwave radiometer and established a semiempirical model of frozen soil microwave emission [17]. Zuerndorfer developed a freeze-thaw state classification algorithm based on radiobrightness data from the Nimbus-7 Scanning Multichannel Microwave Radiometer (SMMR) [18, 19]. Other three factors were measured by the FT standard, respectively, from climate, topography, and vegetation growth status. Because there are few applications of FT estimate in the high altitude frozen region of China, not much work is done on the impact of freeze-thaw cycle on alpine areas.

Freeze-thaw area of more than 1,269,800 km^2 accounted for about 13.4% of the entire China according to second national soil erosion data by remote sensing survey. The majority of the freeze-thaw erosion area is mainly distributed in Northeast, Northwest of high mountainous, and Tibetan Plateau. According to a survey, Tibetan Plateau, the main source of the Yangtze River and Yellow River, has FT erosion area of 1.04×10^6 km^2, which has seriously affected the production activity and life of local people and development of regional economy. Meanwhile [20, 21], the products of FT erosion have become the main source of sediment in the Yangtze River and Yellow River. Previous studies have also shown that FT erosion can increase soil erodibility and slop soil instability, thereby increasing the amount of soil loss by water [22–24]. In some FT erosion areas, the snowmelt runoff erosion at spring accounts for most of annual soil erosion. Therefore, accelerating researches on FT erosion and finding the effective methods and means to prevent FT erosion are extremely urgent and necessary. As water conservation functions of Silingco wetland has highly soil moisture and large temperature difference between day and night, it is located in the high-altitude of the Tibetan Plateau, its freeze-thaw erosion occurred significantly. In this paper, we established an estimation model of FT erosion and applied it for analysis of FT erosion in Silingco watershed wetland as an example.

Figure 1: Spatial location of Silingco watershed wetland.

2. Materials and Methods

2.1. Study Area. Silingco watershed wetland locates in Tibet Autonomous Region, bordering the northeastern Tibetan Plateau, the southern Xigaze, Lhasa and Linzhi, eastern Ali, western Ali, and the northern Xinjiang and Qinghai. It has total area of 39.46 km^2, longitude of E 83°52′20″ ~95°01′00″, and latitude of N 29°56′20″ ~36°41′00″ (Figure 1). It has nine counties including Mani, Bange, Shenzha, Nagqu, Anduo, Suoxian, Baqing, Biru, and Jiali. The area has high altitude and low temperature [25]. The soil erosion in some regions is mainly FT erosion, which has appeared to have more and more evident influences on people's life and local economic development.

2.2. Index Calculation. Estimation of the quantities of FT erosion should be based on the amount of the soil loss per unit area per unit time in the FT region. Currently, methods used to measure the quantities of FT erosion have not been reported in China and abroad; thus it is very difficult to quantitatively estimate the quantity of FT erosion. In fact, FT erosion intensity or the quantity of FT erosion differs in different FT erosion zones due to the different impacts of FT process, migration conditions of FT products, and the climatic factors. Therefore, we chose six most influential factors on FT occurrence and development including annual FT cycle days, average diurnal FT phase-changed water content, average annual precipitation, slope, aspect, and vegetation coverage to comprehensively evaluate the quantity of FT erosion.

2.2.1. Annual FT Cycle Days. The annual FT cycle day is the number of days that FT cycles occur in a year. We for the first time applied the microwave remote sensing technology for judging FT state. The land surface brightness temperatures at day and night were calculated using the AMSR-E (*The Advanced Microwave Scanning Radiometer—EOS*) data from satellite-borne passive microwave radiometer and used to calculate the criterion factors for measuring the changes in land surface temperature and emissivity. AMSR-E is modified from the Advanced Earth Observing Satellite-II in NASA. It observes atmospheric parameters, including precipitation, snow water equivalent, surface wetness, atmospheric cloud water, and water vapor. It seems worthwhile to examine the importance of freezing-thaw by investigating the frequency of freeze-thaw cycles by the use of AMSR-E. The discriminant analysis algorithm was used to classify the FT state at the land surface, label the regions with FT cycles, and statistically calculate the annual FT cycle day [26, 27]. The images were then used to generate a 30 m × 30 m grid map and the classification map of annual FT cycle day in the erosion areas. The FT criterion indexes are calculated as follows:

$$F = 1.47\mathrm{Tb}_{36.5V} + \frac{91.69\mathrm{Tb}_{18.7H}}{\mathrm{Tb}_{36.5V}} - 226.77,$$
$$T = 1.55\mathrm{Tb}_{36.5V} + \frac{86.339\mathrm{Tb}_{18.7H}}{\mathrm{Tb}_{36.5V}} - 242.41, \quad (1)$$

where Tb is the polarized brightness temperature, $36.5V$ is the V-polarized brightness temperature of channel 36.5 GHz, and $18.7H$ is the H-polarized brightness temperature of channel 18.7 GHz. $F > T$ indicates frozen soil, and otherwise, indicates thawed soil.

2.2.2. Average Diurnal Phase-Changed Water Content. In the FT regions, the phase-changed water content reflects the difference in soil liquid water content during the FT process. The changes in potential land surface emissivity were extracted from AMSR-E data and used to estimate phase-changed water content at the land surface. Microwave remote sensing has potential ability to monitor the soil freeze-thaw process and accompanying water phase change. Ground experiments and model simulation of the characteristics of microwave radiation of soil freeze-thaw process constitute the base of microwave detection of soil freeze/thaw process. Soil freezing process is essentially a phase change process of liquid water. The microwave emission of soil is sensitive to soil liquid water content mainly because of the large difference of the dielectric constant between liquid water and dry soil and ice. The total phase-changed water content was accumulated based on the defined FT erosion zone and FT cycle days and used to calculate the average diurnal phase-changed water content [28–33] using the following formula:

$$m_{\mathrm{pcv}} = A\left(\frac{\mathrm{Tb}_{d,10.65H}}{\mathrm{Tb}_{d,36.5H}} - \frac{\mathrm{Tb}_{a,10.65H}}{\mathrm{Tb}_{a,36.5H}}\right) + B, \quad (2)$$

where m_{pcv} is phase-changed water content, A and B are the regression coefficients, subscript d indicates down rail, and subscript a indicates a rise track.

2.2.3. Annual Average Precipitation. TRMM (*Tropical Rainfall Measuring Mission*) 3B42 jointly developed by US NASA and Japanese NASDA was used to measure the tropical rainfall. The system has three resolutions of 3 hours, 1 day, and 1 month. The monthly data was used in this paper. The average annual precipitation was calculated based on the monthly precipitation during 1998–2010.

2.2.4. Slope and Aspect. A 30 m × 30 m grid slope and aspect data were generated using ArcGIS software based on the DEM data, which was obtained jointly by US NASA and NIMA at 30 m using SRTM1 because of its higher spatial resolution.

2.2.5. Vegetation Coverage. The vegetation coverage was converted from the data obtained by a NDVI (*Normalized Difference Vegetation Index*) MOD13Q1 instrument, which is a product of MODIS with resolution of 250 m and 16 days, based on the following formula:

$$\mathrm{FVC} = \left(\frac{(\mathrm{NDVI} - \mathrm{NDVI}_{\min})}{(\mathrm{NDVI}_{\max} - \mathrm{NDVI}_{\min})}\right)^{K}, \quad (3)$$

where FVC is the fractional vegetation coverage, NDVI is the normalized vegetation index, $\mathrm{NDVI}_{\max}$ is the maximum value of pure vegetation, $\mathrm{NDVI}_{\min}$ is the minimum value of pure bare soil, and K is an experience factor and its value is 1 in this paper. The resampling resolution of vegetation coverage is 30 m.

2.2.6. Estimation Model of FT Erosion Content. The comprehensive evaluation of FT erosion quantity is a complex process of a number of affecting factors to make it an integrated single index. In fact, the evaluation classification is only possible in one dimension. This requires integration of multiple factors impacting the FT erosion to obtain a comprehensive evaluation index. In this paper, we adapted a weighted summation method to calculate the comprehensive evaluation index using the following formula:

$$I = \frac{\sum_{i=1}^{n} W_i I_i}{\sum_{i=1}^{n} W_i}, \quad (4)$$

where I is the dimensionless comprehensive evaluation index of FT erosion corresponding to erosion intensity, n is the six indicators, W_i is the weight of each index, and I_i is the dimensionless value of each index in different ranges. The weights of classification factors for FT erosion were determined by using analytic hierarchy process. First, the six factors (annual FT cycle days, average diurnal phase-changed water content, average annual precipitation, slope, aspect and vegetation coverage (Figure 2)) were pairwise compared and used to build a comparison matrix. Then their weights and consistency were calculated and tested using the square root method. Table 1 lists the weight values of the six factors.

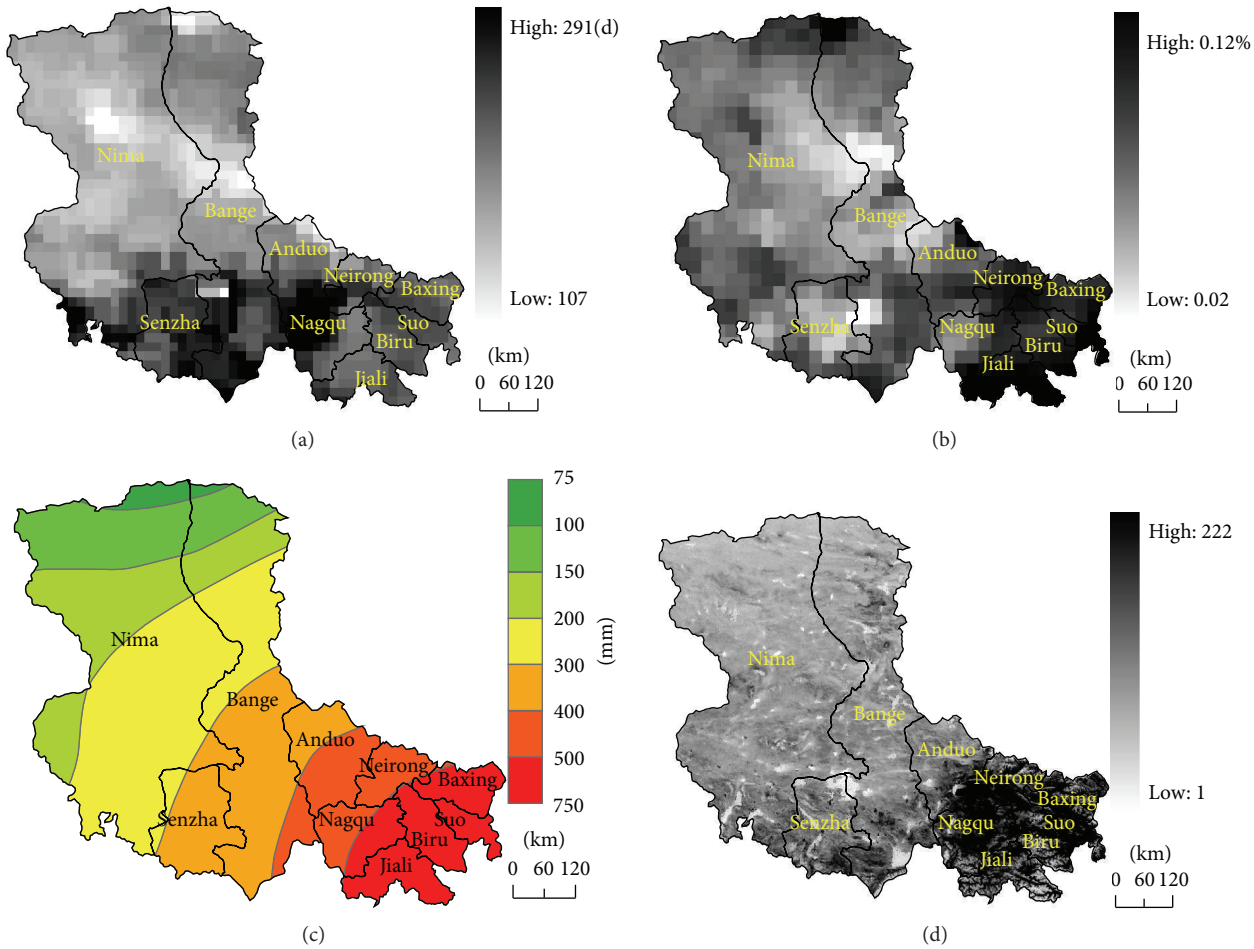

Figure 2: Distribution maps of annual FT cycle days (a), average diurnal phase-changed water content (b), average annual precipitation (c), and vegetation coverage (d).

3. Results

3.1. Classification Map and Analysis of Each Indicator. By introducing the indexes, we obtained the annual FT cycle days and the average diurnal phase-changed water content in eight years from 2003 to 2010, the average annual precipitation, slope, and aspect with 30-meter resolution in 12 years from 1998 to 2010, and the average fractional vegetation coverage from 2003 to 2010 (Figure 1). Table 1 lists the assigned classification value of each evaluation factor.

As shown, the annual FT cycle day did not have similar distribution pattern with diurnal phase-changed water content. Shenzha County and Anduo County in south Silingco watershed wetland had higher FT cycle day while Neirong, Biru, Jiali, and Suoxian in the east Silingco watershed wetland had higher diurnal phase-changed water content. The 12-year average annual precipitation increased mainly from west to east. The areas with relatively flat terrain, abundant rainfall, and relatively low altitude in the east Silingco watershed wetland had higher vegetation coverage.

3.2. Estimation of the Quantity of FT Erosion. Based on the erosion index calculated from formula (4), the FT erosion in Silingco watershed wetland could be divided into 6 categories: no erosion ($I < 1$), slight erosion ($1 < I < 2.2$), mild erosion ($2.2 < I < 2.5$), moderate erosion ($2.5 < I < 2.8$), intensive erosion ($2.8 < I < 3.3$), and severe erosion ($3.3 < I < 4$). The index standard distribution of erosion zones in Silingco watershed wetland is shown in Figure 3.

Erosion is widely distributed in Silingco watershed wetland of Northern Tibet, with FT erosion intensity mainly in moderate and intensive categories. The total FT erosion area in the region is 3.91×10^5 km^2. Among them, the slight, mild, moderate, intensive, and severe erosion areas are 3.07×10^4 km^2, 6.89×10^4 km^2, 1.7×10^5 km^2, 1.06×10^5 km^2, and 1.55×10^4 km^2, accounting for 7.8%, 17.62%, 43.47%, 27.1% and 3.9% of the total erosion area, respectively.

3.3. The Spatial Distribution of FT Erosion. The spatial distribution of the different FT erosion zones differed significantly. The slight erosion zone was concentrated in the southwest areas and southeast river of Silingco watershed wetland (Figure 3) and distributed in the low-lying potential of lakes, rivers, and wetlands. Its rainfall is between 200 and 300 mm, slope was less than 3°, and aspect was less than 45°.

Table 1: Classification assignment of evaluation indices and their weights.

Index	Assignment criteria				Weight (W_i)
Annual FT cycle days (day)	≤60	100–120	120–200	>200	0.15
Average diurnal phase-changed water content (%)	≤0.03	0.03–0.05	0.05–0.07	>0.07	0.15
Average annual precipitate (mm)	≤150	150–300	300–500	>500	0.05
Slope (°)	0–3	3–8	8–15	>15	0.35
Aspect (°)	0–45, 315–360	45–90, 270–315	90–135, 225–270	135–225	0.05
Vegetation coverage (%)	60–100	40–60	20–40	0–20	0.25
Assigned value	1	2	3	4	

Table 2: Error matrix of the quantity of FT erosion.

	FT erosion results being evaluated					
	Severe erosion	Intensive erosion	Moderate erosion	Mild erosion	Slight erosion	Sum
Level of sampling point						
Severe erosion	5	1	2	1	1	**10**
Intensive erosion	0	23	3	5	9	**40**
Moderate erosion	3	2	27	4	5	**41**
Mild erosion	2	0	4	16	6	**28**
Slight erosion	1	3	7	1	121	**133**
Sum	**11**	**29**	**43**	**27**	**142**	**252**

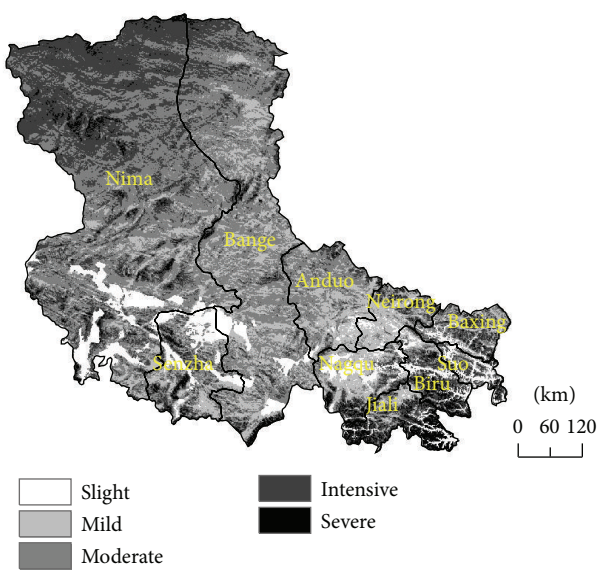

Figure 3: Classification map of the quantity of freeze-thaw erosion in Silingco watershed wetland.

In addition, the area has high vegetation coverage and the lowest spatial position of average diurnal phase-changed water content. For example, Gerencuo basin in Shenzha county, Jiali basin, and Suoxian basin had less FT erosion. Mild FT erosion was distributed mainly in the south of Silingco watershed wetland and located in the transition areas of the slight and moderate erosion zones, such as Nagqu, Neirong, and Anduo. These areas had rainfall between 300 and 500 mm and higher degree of vegetation coverage. The moderate FT erosion zones accounted for the most majority and were distributed mainly along Silingco watershed wetland. These areas had high incidence of avalanche, relatively high elevation, and different rainfall from 75 mm to 750 mm. The intensive FT erosion zones accounted for the second most majority and were distributed mainly in northern Silingco watershed wetland. These areas were uninhabited and had a greater degree of natural disasters and quite different average annual temperature. The severe FT erosion zones were surrounded by the intensive erosion zones and included Jiali, Biru, and north of Nimaxian. These areas had minimum proportion and distributed mostly around the watershed.

3.4. Accuracy Assessment. To test the accuracy of the calculated freeze-thaw erosion rank, we adopted the basic error matrix and precision index in the field survey. A total of 252 points were sampled with different spatial location, land use types, fractional vegetation cover change, landforms, aspect, slope, and erosion types, and so on, and analyzed using the principal component analysis to obtain the optimum index and weights. Table 2 lists the error matrix of the calculated erosion category and the field survey.

The map precision proved the effectiveness of the classification method. The mild intensity erosion map reached 91% precision and the extreme intensity, high intensity, and middle intensity erosion maps reached 60% precision (Table 3), reflecting the high credibility of the classification

Table 3: Precision index of the quantity of FT erosion.

	Cartographic precision	Omission	User accuracy	Commission
Severe erosion	0.50	0.50	0.45	0.55
Intensive erosion	0.58	0.43	0.79	0.21
Moderate erosion	0.66	0.34	0.63	0.37
Mild erosion	0.57	0.43	0.59	0.41
Slight erosion	0.91	0.09	0.85	0.15

results. The mild and extreme intensity erosion maps also had high preciousness, although the area distribution of sharp erosion was low. The overall precision indicated that the classification results were in agreement with the actual erosion level. Their probability reached 76.2%, illustrating that the freeze-thaw erosion model of Silingco watershed wetland is effective and accurate.

4. Discussions and Conclusions

FT erosion, as one of the of major erosion types in China, is mainly located in the western China and has not yet attracted enough attention in China and abroad. Studies on FT erosion are rare and far behind those on the water and wind erosion. In particular, usually occurred freeze-thaw erosion research of Plateau wetlands is insufficient.

Previous models for FT erosion estimation are not very accurate and lack scientific basis due to poor selection of impact factors, such as temperature, slope, aspect, vegetation coverage, annual precipitation, and soil. In this paper, we, from the scientific point of view, introduced microwave remote sensing techniques to determine the state of FT erosion and monitor the FT process, which allow us attribute FT erosion to the development of surface permafrost and parameters monitoring. By using the microwave remote sensing techniques, we can detect the soil sensitivity to the changes of moisture and calculate the annual FT cycle days and average diurnal phase-changed water content. These selected factors are reasonable and scientific by the high erosion accuracy.

The FT erosion estimation model established in this paper has certain scientific basis and strong operationality and practicability. The results have certain significance for the in-depth understanding of the occurrence and development process of FT erosion in Silingco watershed wetland of Northern Tibet, and for the rational use and preservation of the ecological environment. Follow-up studies will further strengthen the mechanisms underlying the capacity of a typical FT erosion zone, the causes for the spatial distribution of moderate erosion area, the evaluation precision, and the ecological consequences of erosion.

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Review Article

A Review of Surface Water Quality Models

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Surface water quality models can be useful tools to simulate and predict the levels, distributions, and risks of chemical pollutants in a given water body. The modeling results from these models under different pollution scenarios are very important components of environmental impact assessment and can provide a basis and technique support for environmental management agencies to make right decisions. Whether the model results are right or not can impact the reasonability and scientificity of the authorized construct projects and the availability of pollution control measures. We reviewed the development of surface water quality models at three stages and analyzed the suitability, precisions, and methods among different models. Standardization of water quality models can help environmental management agencies guarantee the consistency in application of water quality models for regulatory purposes. We concluded the status of standardization of these models in developed countries and put forward available measures for the standardization of these surface water quality models, especially in developing countries.

1. Signature of Water Quality Models

Water quality models can be effective tools to simulate and predict pollutant transport in water environment [1–3], which can contribute to saving the cost of labors and materials for a large number of chemical experiments to some degree. Moreover, it is inaccessible for on-site experiments in some cases due to special environmental pollution issues. Therefore, water quality models become an important tool to identify water environmental pollution and the final fate and behaviors of pollutants in water environment [3]. These construction projects such as petrochemical, hydrological, and paper-making projects can bring serious effects on aquatic environment after enforcement [4, 5]. Therefore, these environmental effects have to be simulated, predicted, and assessed using numerical models before these construction projects are implemented. These modeling results under different pollution scenarios using water quality models are very important components of environmental impact assessment. Moreover, they are also the important basis for environmental management decisions as they not only provide data assistance for environmental management agencies to authorize the construction projects but also provide technical supports for water environmental protection agencies [6, 7]. Whether these model results are right or not can greatly impact the reasonability and scientific significance of the authorized construction projects and the availability of pollution control measures.

With the development of model theory and the fast-updating computer technique [8], more and more water quality models have been developed with various model algorithms [3, 4]. Up to date, tens of types of water quality models including hundreds of model softwares have been developed for different topography, water bodies, and pollutants at different space and time scales [3, 9]. However, there are often big differences between these modeling results due to different theories and algorithms of these models, which can lead to the insistency of the predicted results using different models, and thus bringing different environment management decisions as these modeling results cannot be referred or compared to each other [10].

The uniform model standardization system has not been established yet in most developing countries [9, 11], which limits the wide applications of these models to environmental management due to no references and comparisons among different modeling results. Therefore, it is very necessary for most developing countries to better understand the availability and precisions of different water quality models and their methods of calculation and calibration and progress in the model standardization in order to apply effectively

these models and form a good model regulation system [11, 12]. In particular, this work can contribute to making better environmental management policies and authorizing reasonable construction projects.

2. Development of Surface Water Quality Models

Surface water quality models have undergone a long period of development since Streeter and Phelps built the first water quality model (S-P model) to control river pollution in Ohio state of the US [13]. Surface water quality models have made a big progress from single factor of water quality to multifactors of water quality, from steady-state model to dynamic model, from point source model to the coupling model of point and nonpoint sources, and from zero-dimensional mode to one-dimensional, two-dimensional, and three-dimensional models [31, 32]. More than 100 surface water quality models have been developed up to now. Cao and Zhang [11] classified these models based on water body types, model-establishing methods, water quality coefficient, water quality components, model property, spatial dimension, and reaction kinetics. However, each surface water quality model has its own constraint conditions [33]. Therefore, water quality models still need to be further studied to overcome the shortcomings of these current models. Generally, the surface water quality models have undergone three important stages since 1925 to now.

2.1. The Primary Sage (1925–1965). Water quality of water bodies has been paid much more attention to at this stage. The water quality models focused on the interactions among different components of water quality in river systems as affected by living and industrial point source pollution [9, 11, 34]. Like hydrodynamic transmission, sediment oxygen demand and algal photosynthesis and respiration were considered as external inputs, whereas the nonpoint source pollution was just taken into account as the background load [35, 36].

At the beginning of this stage (from 1925 to 1965), the simple BOD-DO bilinear system model was developed and achieved a success in water quality prediction, and the one-dimensional model was applied to solve pollution issues in rivers and estuaries [33]. After that, most researchers modified and further developed the Streeter-Phelps models (S-P models). For example, Thomas Jr. [14] believed that BOD could be reduced without oxygen consumption due to sediment deposition and flocculation, and the reduction rate was proportional to the number of remained BOD; thus, the flocculation coefficient was introduced in the steady-state S-P model to distinguish the two BOD removal pathways. O'Connor [15] divided BOD parameter into carbonized BOD and nitrified BOD and added the effects of dispersion based on the equation. Dobbins-Camp [16, 17] added two coefficients, including the changing rate of BOD caused by sediment release and surface runoff as well as the changing rate of DO controlled by algal photosynthesis and respiration, to Thomas's equation.

2.2. The Improving Stage (1965–1995). From 1965 to 1970, water quality models were classified as six linear systems and made a rapid progress based on further studies on multidimensional coefficient estimation of BOD-DO models. The one-dimensional model was updated to a two-dimensional one which was applied to water quality simulation of lakes and gulfs [37, 38]. Nonlinear system models were developed during the period from 1970 to 1975 [39]. These models included the N and P cycling system, phytoplankton and zooplankton system and focused on the relationships between biological growing rate and nutrients, sunlight and temperature, and phytoplankton and the growing rate of zooplankton [35, 37, 39]. The finite difference method and finite element method were applied to these water quality models due to the previous nonlinear relationships and they were simulated using one- or two-dimensional models.

After 1975, the number of state variables in the models increased greatly, and the three-dimensional models were developed at this stage, and the hydrodynamic mode and the influences of sediments were introduced to water quality models [40, 41]. Meanwhile, water quality models were combined with watershed models to consider nonpoint source pollution input as a variable [42, 43]. The effects of sediments were coped with inner interaction processes of the models [43]; so, the sediment fluxes could vary accordingly under different input conditions. Therefore, the water quality management policies were greatly improved due to more constraint conditions and nonpoint source pollution simulation at watershed scale. The typical models including QUAL models [18, 19], MIKE11 model [22], and WASP models [23, 44] were developed and used at this stage. Meanwhile, the one-dimensional OTIS model developed by USGS was also applied to water quality simulation [45, 46].

2.3. The Deepening Stage (after 1995). Nonpoint source pollution has been reduced due to strong control in developed countries. However, the dry and wet atmospheric deposition such as organic compounds, heavy metals, and nitrogen compounds showed increasing effects on water quality of rivers [47–49]. Although nutrients and toxic chemical materials depositing to water surface have been included in model framework, these materials not only deposited directly on water surface but also they can be deposited on the land surface of a watershed and sequentially transferred to water body [20, 50], which has been an important pollutant source. From the viewpoint of management demands, an air pollution model has to be developed to introduce this proceed in the model, indicating that the static or dynamic atmospheric deposition should be related to a given watershed [51]. Therefore, at this stage, some air pollution models were integrated to water quality models to evaluate directly the contribution of atmospheric pollutant deposition [20].

With the exception of the typical models such as QUAL 2K model [52], WASP 6 model [24], QUASAR model [25, 53], SWAT model [21], and MIKE 21 [26] and MIKE 31 models [27] (Table 1), other water quality models have also been developed to simulate complicated water environmental conditions. For example, Whitehead et al. [54] developed

Table 1: Main surface water quality models and their versions and characteristics.

Models	Model version	Characteristics
Streeter-Phelps models	S-P model [13]; Thomas BOD-DO model [14]; O'Connor BOD-DO model [15]; Dobbins-Camp BOD-DO model [16, 17]	Streeter and Phelps established the first S-P model in 1925. S-P models focus on oxygen balance and one-order decay of BOD and they are one-dimensional steady-state models.
QUAL models	QUAL I [11]; QUAL II [18]; QUAL2E [19]; QUAL2E UNCAS [19]; QUAL 2K [20, 21]	The USEPA developed QUAL I in 1970. QUAL models are suitable for dendritic river and non-point source pollution, including one-dimensional steady-state or dynamic models.
WASP models	WASP1-7 models [22, 23]	The USEPA developed WASP model in 1983. WASP models are suitable for water quality simulation in rivers, lakes, estuaries, coastal wetlands, and reservoirs, including one-, two-, or three-dimensional models.
QUASAR model	QUASAR model [11, 24, 25]	Whitehead established this model in 1997. QUASAR model is suitable for dissolved oxygen simulation in larger rivers, and it is a one-dimensional dynamic model including PC_QUA SAR, HERMES, and QUESTOR modes.
MIKE models	MIKE11 [22]; MIKE 21 [26]; MIKE 31 [27]	Denmark Hydrology Institute developed these MIKE models, which are suitable for water quality simulation in rivers, estuaries, and tidal wetlands, including one-, two-, or three dimensional models.
BASINS models	BASINS 1 [11, 28]; BASINS 2 [11, 28]; BASINS 3 [11, 28]; BASINS 4 [28]	The USEPA developed these models in 1996. BASINS models are multipurpose environmental analysis systems, and they integrate point and nonpoint source pollution. BASINS models are suitable for water quality analysis at watershed scale.
EFDC model	EFDC model [29, 30]	Virginia Institute of Marine Science developed this model. The USEPA has listed the EFDC model as a tool for water quality management in 1997. EFDC model is suitable for water quality simulation in rivers, lakes, reservoirs, estuaries, and wetlands, including one-, two-, or three-dimensional models.

a semidistributed integrated nitrogen model (INCA) based on the effects of atmospheric and soil N inputs, land uses, and hydrology. More recently, Fan et al. [55] integrated QUAL 2K water quality model and HEC-RAS model to simulate the impact of tidal effects on water quality simulation. For the integration of point and nonpoint sources, the US Environmental Protection Agency (USEPA) developed a multipurpose environmental analysis system (BASINS), which makes it possible to assess quickly large amounts of point and nonpoint source [28]. Meanwhile, the USEPA also listed the EFDC model as a tool for water quality management.

Among the previously mentioned surface water quality models, these models including the Streeter-Phelps model, QUASAR model, QUAL model, WASP model, CE-QUAL-W 2 model, BASINS model, MIKE model, and EFDC model were widely applied worldwide [56, 57]. Recently, Kannel et al. [58] concluded that these public domain models (e.g., QUAL2EU, WASP7, and QUASAR) are the most suitable for simulating dissolved oxygen along rivers and streams. Generally, most developed countries (especially the US or European countries) have developed better and advanced surface water quality models [22, 27, 28, 30]. Some surface water quality models have also been established in some universities or institutes of China over the past years [11], but these models were still not widely utilized like MIKE models, EFDC model, and WASP models [59, 60].

3. Standardization of Surface Water Quality Models

Water quality models should be more available, standardized, and reliable when they are utilized to aid the important and valid reports (e.g., environmental impact assessment report). Therefore, it is very necessary for environmental management agencies to mandate or list some water quality models in order to guarantee the consistency of water quality models for regulatory purposes [61]. The models can be regulated and standardized through these pathways such as the establishment of the national model assessment indicator and validating system, published articles, workshops, or setting up local workgroup [62]. For example, The USEPA holds regular academic conferences on water quality models to identify and update regulatory models [62]. The European Union organizes regular workshops on the consistency of water quality models to evaluate the regulatory models. Moreover, the standardized models should be able to be downloaded free and have open origin codes.

Special research institutes of water quality models have been built to do a lot of researches on the regulation and standardization of water quality models in some regions or countries [62, 63]. They recommended some prediction models based on the requirements of environmental management. Compared to other countries, most water environmental models have been standardized in the US. The Water Science

Center belonging to the USEPA focuses on the following studies regarding water resources management and conservation, the theory and methods applied in water environments, numerical models, calculating tools, and databanks. Meanwhile, the USEPA also provides foundations for some universities, institutes, or companies to develop and compare related models and finish a series of research reports. In 2002, the USEPA mandated the Guidance for Quality Assurance Project Plans for Models, and some advices and guidance principles were given for the applications of water quality models in this guidance [64]. Additionally, the USEPA also authorized Tetra Tech Inc. to do the project of TMDL Model Evaluation and Research Needs, through which the modeling capacity, availability, and scopes of more than 60 models have been evaluated and compared using detailed appraisal forms [65]. Based on the above researches, the USEPA finally published the Guidance on the Development, Evaluation, and Application of Environmental Models in 2009. This guidance introduces concisely the characteristics and appropriate environment process modeling of these surface water environment models such as HSPF model, WASP model, and QUAL2E model and also gave the website links for more details of these models. The best practices for model evaluation are also appended to this guidance, which describes the methods, objectives, and procedures of model evaluation in detail [66]. Besides the guidance, the Council for Regulatory Environmental Modeling of the USEPA provides the model banks on its website. The United States Geological Survey, Federal Emergency Management Agency, and the United States Army Corps of Engineers also have similar model banks and detailed introduction for different types of models. The USEPA recommended its own developed models and those models developed by other research institutes or companies, but an announcement has been provided in the recommendation report that the recommended models do not denote that they have been authenticated by the USEPA [66]. The USEPA only suggested how to select appropriate models under different environmental conditions as each model has its own appropriate scope and scale. However, Kannel et al. [58] pointed out that the choice of a model depends upon availability of time, financial cost, and a specific application.

Similarly special research institute of model development and evaluation has been set up by the United Kingdom Environment Agency (UKEA). This institute helped the UKEA finish the Framework for Assessing the Impact of Contaminated Land on Groundwater and Surface Waterand and put forward the procedure, method, and prediction models of surface water environmental impact assessment of potential pollution sources, which can assess the influencing degree of pollution sources on water environment. The Her Majesty's Inspectorate of Pollution (HMIP) recommended 54 surface water quality models and limiting conditions for rivers, lakes, reservoirs, estuaries, and sea pollution assessment. Aspinwall and Company Limited recommended 11 models for different conditions including 1 one-dimensional model, 4 two-dimensional models, and 6 three-dimensional models, of which 11 models for steady-state simulation and 10 models for dynamic simulation [67]. In Korea, the Ministry of Environment made a general plan for water environmental management in 2006, which described 6 water quality prediction models in detail and recommended a series of numerical models including widely-used Qual2E model and EFDC model [68]. The MIKE models and Tuflow model were widely applied to predict surface water quality in Australia. MIKE models were adapted in Denmark to solve some issues in these fields such as ecology, environmental chemistry, water resources, hydraulic engineering, and hydrological dynamics. In China, the Delft 3D hydrological dynamic-water quality model has been used to simulate water environmental quality in Hong kong since 1970s and now become the standard model of Hong kong Environment Agency. Taiwan Environmental Protection Bureau issued the guidance on methods of water quality assessment of rivers and environmental impact assessment and provided a water quality model list for different conditions in this guidance. The Ministry of Environmental Protection of China formally published the Technical Guidelines for Environmental Impact Assessment (Surface water Environment) in 1993 and recommended some numerical models for rivers, lakes, estuaries, and marine environment under different conditions [69]. However, the standardized numerical models in China are still not provided yet up to date. Most models such as MIKE models, EFDC model, and Delft 3D model have been applied to simulate water environmental quality in most institutes of environmental impact assessment [70, 71]. However, little information is available on the differences in model results from different models and the suitability and parameter sensitivity of these models. Moreover, it is also an urgent task to standardize some numerical models to compare the modeling results among different regions efficiently. Additionally, Moriasi et al. [72] suggested to develop the consistent framework of model calibration and validation guidelines, as it is difficult to compare modeling results from different studies with different calibration and validation methods.

4. Measurements for the Standardization of Surface Water Quality Models

The appraisal techniques of the standardization of water quality models and their authentication system can provide an important scientific basis for the development of software informatization for water quality models and environmental impact assessment [68]. To improve the standardization of surface water quality models, the best way is to understand fully the status, progress, frame structure, assessing indicators, and authentication system of the standardization system of surface water quality models in developed countries, especially in some European or North American countries. Based on the previously mentioned, it is necessary for environmental management agencies of those countries without standardization models of water environmental quality to develop their own construction and frame structure of standardized model system of surface water quality, screen assessing indicators, procedures, and methods to establish their own authentication and standardization system for surface water quality models.

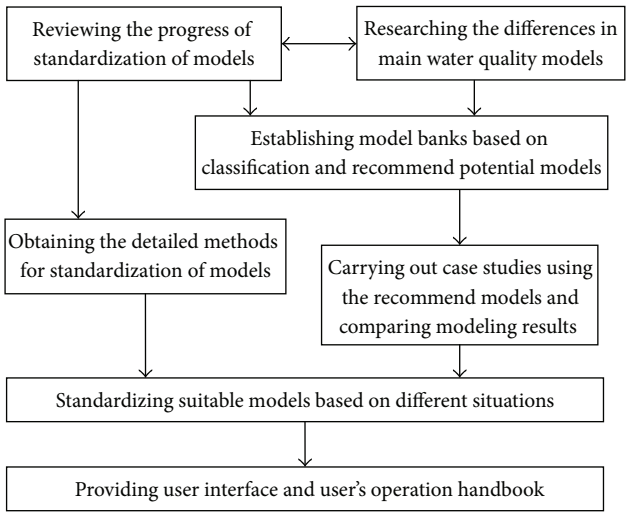

Figure 1: Flow chart of the standardization of water quality models.

The specific measures for the standardization of surface water quality models are given as follows (Figure 1).

(1) To research the water quality models which are widely used in the fields of surface water environmental impact assessment to know well the model mechanisms, suitable conditions, appropriate scopes, model parameters, stability, and the differences in modeling results.

(2) To develop case bank and data bank for surface water quality models through indoor experiments, case collection, and field monitoring.

(3) To compare the modeling results among different models and to conclude and analyze the input and output files, equations, theories, frames, and calculating methods of water quality models based on some case studies.

(4) To provide the screening indicators and appraisal methods for water quality models to establish the appraisal authentication system of these models and standardize the standard interfaces of input and output data for these models. To standardize some water quality models and list the standardized models for environmental impact assessment based on each country's actual conditions.

(5) To give the parameter calibration and validation methods and the access, sources, and recommended values of these parameters and put forward some standard proposals for typical model parameters considering the actual conditions of each country.

(6) To provide user interface of model graphs in native language and publish detailed model operation handbook including model inputs (data access, data processing), model structure, model calibration, model validation, parameter assessment, and model outputs.

5. Conclusions

Water quality models are very important to predict the changes in surface water quality for environmental management in the world. Worldwide, hundreds of surface water quality models have been developed. Moreover, some developed countries have mandated the guidance on water environmental quality assessment and provided some regulated models for surface water quality simulation. Therefore, it is very necessary for most developing countries to standardize some widely used water quality models for efficient environmental impact assessment. However, it is also a big challenge to standardize these models based on their own countries' actual conditions as a lot of investigations and researchers are still needed.

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Research Article

Application of Scenario Analysis and Multiagent Technique in Land-Use Planning: A Case Study on Sanjiang Wetlands

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Land-use planning has triggered debates on social and environmental values, in which two key questions will be faced: one is how to see different planning simulation results instantaneously and apply the results back to interactively assist planning work; the other is how to ensure that the planning simulation result is scientific and accurate. To answer these questions, the objective of this paper is to analyze whether and how a bridge can be built between qualitative and quantitative approaches for land-use planning work and to find out a way to overcome the gap that exists between the ability to construct computer simulation models to aid integrated land-use plan making and the demand for them by planning professionals. The study presented a theoretical framework of land-use planning based on scenario analysis (SA) method and multiagent system (MAS) simulation integration and selected freshwater wetlands in the Sanjiang Plain of China as a case study area. Study results showed that MAS simulation technique emphasizing quantitative process effectively compensated for the SA method emphasizing qualitative process, which realized the organic combination of qualitative and quantitative land-use planning work, and then provided a new idea and method for the land-use planning and sustainable managements of land resources.

1. Introduction

As a consequence of global increase of economic and societal prosperity, ecosystems and natural resources have been substantially exploited, degraded, and destroyed in the last century [1–3]. Land is one of the most valuable natural resources because of its close relation with human daily lives, and it is suffering high strength of landscape transformation activities such as mine exploitation, infrastructure construction, and agriculture cultivation, which have an important influence on the composition and quality of land resources [4]. The sustainable management of land resource has become the broadly accepted backdrop for policy and management decisions in most parts of the world [5–8]. Described as an activity that envisages future land arrangements [9], land-use planning has been recognized as a key instrument for identifying and ensuring sustainable land resource uses, improving the livelihoods of rural communities, and thereby achieving sustainable development [10].

Land use/cover change (LUCC) is the result of diverse interactions between society and the environment [11–13]. As such, land-use planning has triggered debates on social and environmental values and on the need for participatory processes to address individual differences in these values [14–19]. Over the past years, a number of efforts were undertaken for land-use planning with the consideration of individual participatory processes. For example, Ishii et al. proposed a new needs analysis method for the conceptual land-use planning of contaminated sites and illustrated this method with a case study of an illegal dumping site for hazardous waste [20]. Helbron et al. presented the use of indicators in a site-specific assessment method for strategic environmental assessment in regional land-use planning [21].

Koschke et al. presented a multicriteria assessment framework for the qualitative estimation of regional potentials to provide ecosystem services as a prerequisite to support regional development planning [2]. Lestrelin et al. examined the extent to which the evolution of Laos' village land-use planning has resulted in increased local participation and improved livelihoods [17]. Fitzsimons et al. aimed to create a quantitative, community-engaged basis for the evolution of multiple land uses [22]. Lagabrielle et al. considered participatory modelling to integrate biodiversity conservation into land-use planning and to facilitate the incorporation of ecological knowledge into public decision making for spatial planning [23]. Magigi and Drescher analyzed local communities' involvement in land-use planning to regulate land use change and customary land tenure challenges in a rapidly expanding city in Tanzania [24]. Those researches show that collaborative planning has become an increasingly popular approach in land-use decision making, particularly in situations where there are multiple actors with conflicting interests.

Over the last couple of decades, scenario analysis (SA) has become a broadly used tool to provide support and advice to policy makers [25]. In decision-making processes, scenarios can help the decision makers to anticipate possible or potential strategies according to different plausible scenarios, which is usually designed to identify a set of possible futures, where the occurrence of each is plausible, although not assured and not necessarily probable [26]. In this way, SA can be seen as a process of understanding, analyzing, and describing the behavior of complex systems consistently and completely. This kind of systematic analysis is crucial in collaborative planning, and it is widely used in land-use planning [27–33]. These approaches are mainly based on the elicitation of information from a set of people, or a panel of experts or stakeholders, and they are therefore characterized by a high level of subjectivity. Indeed, the quality and performance of SA as a basis for decision support become critically dependent on the quality and performance of the assessments expressed throughout the entire land-use planning process [34]. This represents the primary limitation of such qualitative approaches, particularly when the dynamic complexity of coupled systems is not well understood. Then different planners may get completely diverse planning results, and their scientific creditability has frequently been questioned. Hence, a land-use planning methodology based on a systems approach involving realistic computational modeling and metaheuristic optimization is still lacking in many current practices [35]. Through the above analysis, two key questions will be faced during land-use planning process: one is how to see different planning simulation results instantaneously and apply the results back to assist planning work interactively; the other is how to ensure that the planning simulation result is scientific and accurate.

Landscape digital reconstruction and spatial-temporal distribution simulation based on multiperiod regional land cover data can help to understand the mechanisms and laws of land use succession, recognize the relationship between human activities and land-use changes, predict the future trend of the land use, and ultimately provide strategies to decision maker for land-use planning. A multiagent system (MAS) can be defined as a set of agents that interact in a common environment, able to modify their attributes and their environment [36]. MAS may increase understanding of complex coupled social-ecological systems [37], more particularly in the context of land-use planning [38–41]. The MAS technique can simulate the different planning results based on a systems approach involving realistic computational modeling, so as to provide reference for planning work. More importantly, the data mining technique will be used in the process of multiagent simulation, which ensures the planning process can conform to various regions and get a more accurate, more scientific planning result.

The objective of this paper is to analyze whether and how a bridge can be built between qualitative and quantitative approaches for land-use planning work and to find out a way to overcome the gap that exists between the ability to construct computer simulation models to aid integrated land-use plan making and the demand for them by planning professionals. The specific topic is the integration of scenario analysis with the multiagent system technique, with its application in land-use planning process.

2. Theoretical Framework

MAS simulation and SA method both emphasize the role of human factors in the regional land use change and outstand human intervention in the simulation and planning process. This provides the theoretical basis for the combination of two methods. SA method aims to separate uncertain factors and establish system variables, which focuses on qualitative or qualitative and quantitative combination analysis process. The MAS simulation emphasizes the laws of landscape spatial-temporal distribution under influences of various geographic, economic, and other factors, which focuses on quantitative or quantitative and qualitative combination analysis process. This simulation process is based on actual multiperiod land cover data to obtain the transformation rules, which can reflect the regional actual change situations more effectively.

Through the analysis of MAS simulation and SA method basic principles, combination of them is mainly reflected in the qualitative analysis and design for different scenarios using SA, and actual simulation model and process are completed by MAS quantitatively. Furthermore, factors analysis in scenario design process will be affected by the data mining results of landscape dynamic knowledge database in the MAS simulation process. This makes the scenario analysis process more consistent with the actual situation of regional development and gain a strong geographical significance. The study presents a theoretical framework of land-use planning based on SA method and MAS simulation integration (Figure 1).

3. Case Implementation and Results

3.1. Study Area and Data

3.1.1. Study Area. Wetlands are integral parts of the global ecosystem as they can prevent or reduce the severity offloods,

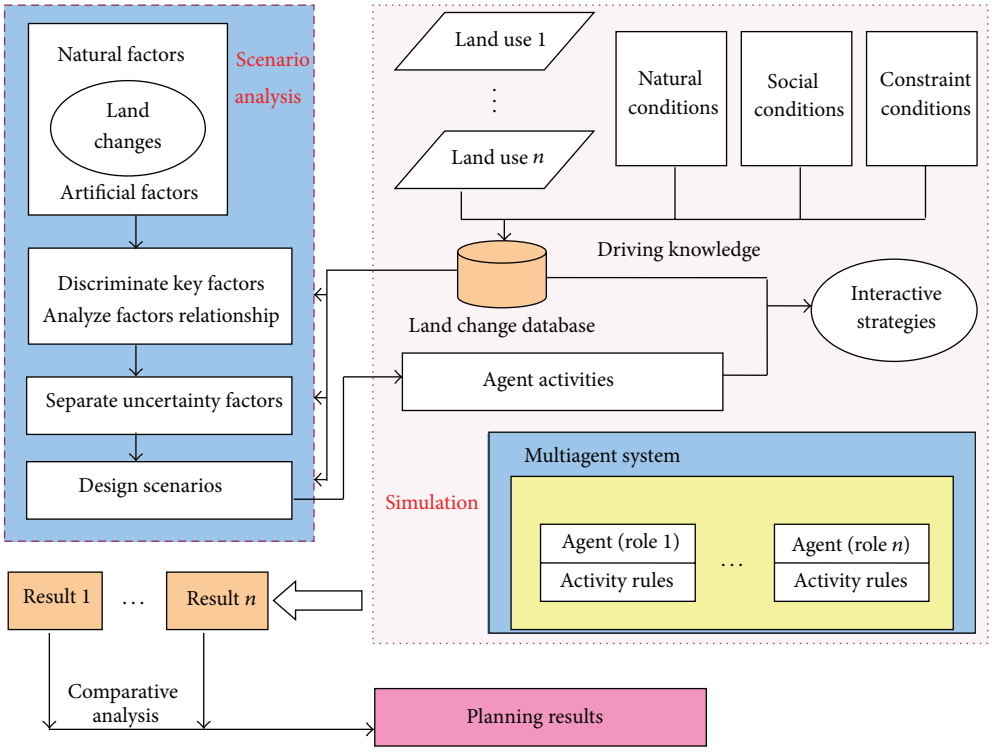

Figure 1: The theoretical framework of landscape planning based on MAS and SA integration.

feed ground water, and provide unique habitats for flora and fauna [42, 43]. The Sanjiang Plain, located in the Northeastern region of China, is one of the largest freshwater wetland in the country (Figure 2). Since the end of the 1950s, large-scale development in the Sanjiang Plain marsh land has occurred [44]. By 2003, about 80% of natural wetlands had been converted to farm land and the progressive loss of wetland is continuing [45]. With a local population of 7.8 million in this region, of which 53.4% is engaged in farming the Sanjiang Plain has become an important grain and bean production region for China [46]. The regional climate is mild humid to subhumid continental monsoon feature. The average temperatures range from −18°C in January to 21-22°C in July with a frost-free period of 120–140 days. Annual precipitation is somewhere around 500–650 mm with 80% occurring from May to September. Most of the rivers at the area have riparian wetlands supporting meadow and marsh vegetation. Sedge (*Carex* spp.) is the dominant plants with *Phragmites* spp. scattered across some parts of the landscape [47].

The study area is limited within 47°21′42″–48°15′9″ north altitude and 133°25′52″–134°33′37″ east longitude in the Northeast of Sanjiang Plain at (Figure 2). Several factors had been taken into consideration when this region was chosen to start this study. Firstly, the Sanjiang Plain is one of the largest marsh distribution region. Secondly, it is a typical representation in the global temperate wetland ecosystems. Thirdly, due to the relative cold weather, deep surface waters, large marsh patches, and sparse population, reclamation of marsh lands in this region is relatively late. Fourthly, study area contains two national nature reserves and three major river systems: Honghe Reserve, Sanjiang Reserve, and Yalu River, Dongjiang River, Bielahong River. They make the study area possess natural original scenery relatively. In addition, during the process of development and utilization in recent decades, the conflict between people and land is a constant game of war. Wetland degradation process under the disturbance from human activities is representative, which makes it suitable for carrying out simulation of wetland landscape spatial-temporal evolution.

3.1.2. Data. To complete the simulation using MAS, land-use data were collected during three-year period (1995, 2000, and 2006). The 1995 and 2000 datasets are used in decision ruling on transformation, while 2006 dataset is used to verify predicted results. Each land use data set contains 5 types of covers such as water, farmlands, resident area, forest, and wetlands. The data of soil, topography, terrain, location, and other thematic parameters are sorted to formulate the transformation probability under the influence of many geographical conditions. The soil data represent 22 different types; topography data contains 14 types of landforms; River distance is a grid file that reflects the distance to rivers and road distance reflects the distance to road. The units of river distance, road distance, and digital elevation model (DEM) data are meters, and the slope is degree. In addition, the existing data collections previously include planning, feasibility reports, scientific research reports, maps and documentation of Honghe National Natural Reserve and Sanjiang National Natural Reserve, and meteorology, hydrology, groundwater

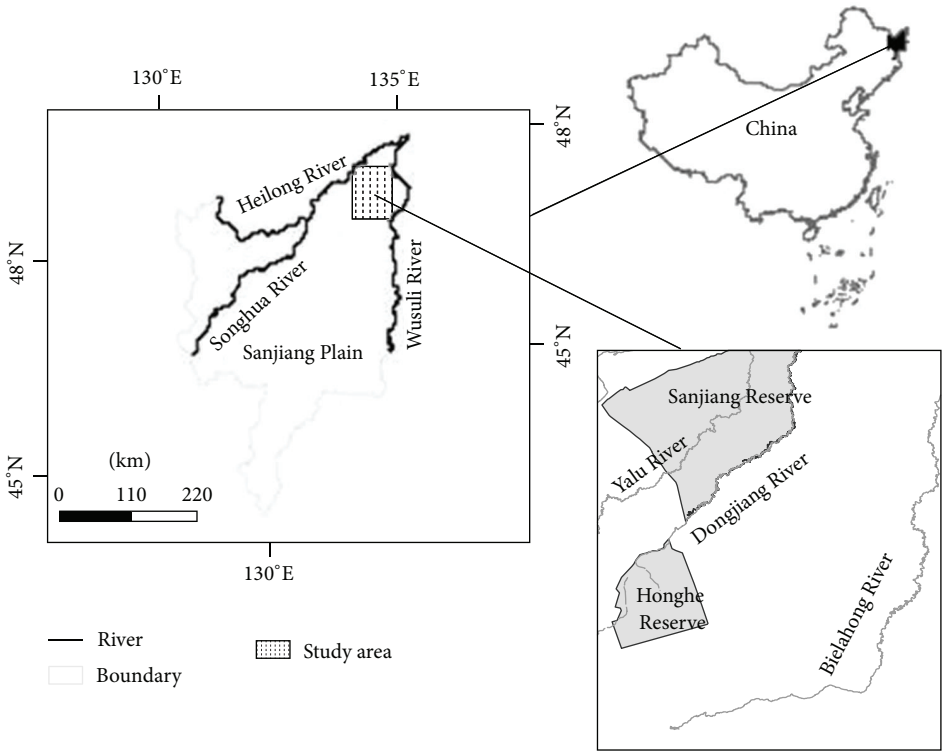

Figure 2: Location of the study area in Sanjiang Plain, China.

observations and other statistical records are available for reference in this research. The detail of each dataset is listed in Table 1. All of the data were coregistered and formatted as GRID format under ArcGIS 9.3.

3.2. Scenario Analysis and Design

3.2.1. The Key Variables and Their Interactions. There are mainly two kinds of factors influencing land use [48, 49]. The first one is direct, and it consists of various forms of activities including conservation and development. The second one is indirect, and it relies on the legal instruments of public policy to influence the behavior of landowners [50].

Considering the actual situation of study area, one direct factor is mainly performed by protectors who are the staff of national nature reserves. They prevent wetlands from developing to other landscape directly. Secondly, the statistic results of land use/cover change cells between the years of 1995 and 2006 show that total area changing from wetland to other cover types is 2402.68 km^2, among which 2215.75 km^2 is from wetlands to farmlands. The farmlands count about 92.22% of total altered area, and this indicates that the cultivation is the main factor that results in the wetland shrink. Then, another direct factor is mainly performed by farmers who change the land cover through reclaiming wetland. Thirdly, because local governments plan for and decide current and future land use, their role in land use/cover change is crucial. The behavior of the governments includes the government's macromanagement and policy establishment, which are indirectly influence regional landscape change. At last, three main variables that cause regional land use change can be simplified as protectors, farmers, and governments according to study area actual situations.

Although one variable has certain functions, however, relying on a single variable cannot always describe and solve complex large-scale problems in reality. Therefore, an application system often includes multiple variables. Each variable is not isolated but an interactive part of the group. Those variables can follow some kind of specific agreement and possess multilinguistic communication skill to complete a specific task. According to the actual situation of study area, logical interaction rules among governments, protectors, and farmers variables are designed as shown in Figure 3.

During the land-use change process, land cover status of certain position is determined by governments, farmers, and protectors variable jointly. First, farmers variable determine whether to reclaim wetlands under various environmental conditions. If the farmers wish to do so, a small part of them will illegally reclaim wetlands, and when this part of farmers goes around obstacles from protectors, the land cover status will be changed. On the contrary, if they are hampered successfully by protectors, then the land cover remains unchanged. The rest of them will apply to the governments variable to reclaim wetlands. Two total diverse consequences will result depending on the government approval to their petition: if the government approved and farmers avoided obstacles from protectors, the land cover status would be changed; if not, the land cover status will be unaffected and unchanged.

Table 1: List of data description.

Name	Content	Resolution	Time	Source	Size
Soil	Spatial distribution of soil types	30 m	1985	Digitizing	8.41 MB
Landform	Spatial distribution of geomorphologic types	30 m	1985	Digitizing	8.41 MB
River distance	Distance to rivers	30 m	1998	Euclidean distance calculation	33.66 MB
Road distance	Distance to roads	30 m	1998	Euclidean distance calculation	33.66 MB
DEM	Digital elevation model	30 m	1986	Digitizing	33.66 MB
Slope	Spatial distribution of slope	30 m	1986	Calculated from DEM	33.66 MB
Land use	Spatial distribution of land cover types	30 m	1995	TM image classification	16.82 MB
Land use	Spatial distribution of land cover types	30 m	2000	TM image classification	16.82 MB
Land use	Spatial distribution of land cover types	30 m	2006	TM image classification	16.82 MB

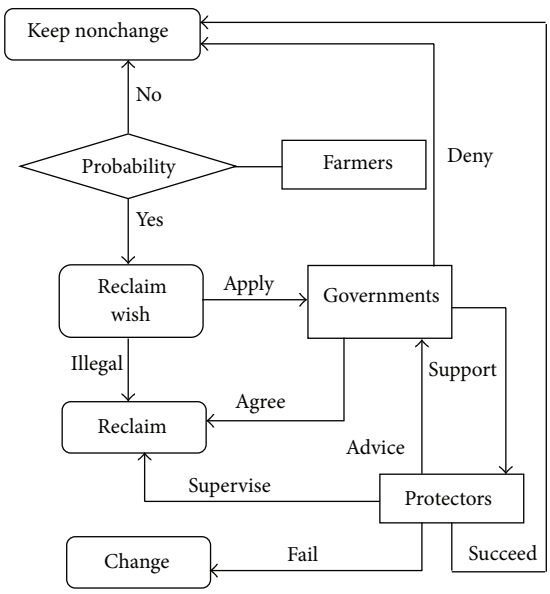

Figure 3: Interaction logistics of variables.

3.2.2. The Scenario Design. After finishing the key factors discrimination and interaction relationship analysis, study defined 3 specific planning scenarios in order to verify the availability of theoretical framework presented by this paper.

Undisturbed Scenario. According to actual land-use change rules of study region from 1995 to 2000, the land cover of year 2006 will be predicted based on MAS simulation technique.

Ecotype Scenario. In this scene, governments' criterions of wetland development approval become strict, and farmers and protectors' awareness of ecological environment protection is strengthened, which cause the reclaim desire to reduce and protection scrutiny to increase. Under these conditions, damage degree of wetland landscape will be degraded. However, it is also very likely to cause regional economy development to be slowed down, and then the income of farmers and governments may drop.

Economy Scenario. Governments encourage farmland development for the needs of economic construction, which affects the speed and manner of the entire regional land-use change. Being driven by economic interests, farmers also strongly destroy wetlands for increasing farmland quantity. Protectors abandon wetland conservation efforts and even join in the wetland destruction and agriculture development in action. Under these conditions, the regional land use subordinates the economic construction and ignores the protection of ecological environment. It is a nonsustainable development mode, but a certain degree of economic achievements may arise in this case.

Research realizes different planning scenarios through modifying decision-making behavior of governments, farmers, and protectors variables. It can clearly explain the specific reasons for the differences of diverse planning scenario simulation results. In detail, undisturbed scene is realized through MAS simulation based on the transformation rules that were gained by data mining technique. Governments, protectors and farmers variables are only used to reflect actual land-use change process. An ecotype scenario is realized through governments auditing standards or reducing the rate of approvals, protectors reinforcing supervision to prevent wetlands being destroyed, and famers reducing their cultivation will. On the other side, an economy scenario is realized through which governments lowering standards or increasing the rate of approvals, protectors reducing supervision to increase rate of development, and famers increasing willingness to reclaim wetlands. After finishing the discrimination of key factors, the analysis of their interactions, and the design of scenarios, the next step is how to quantitatively descript them based on MAS computer simulation models.

3.3. Model Construction of MAS

3.3.1. Construction of Environmental Factors Layer. Environmental factor layer in the model is the natural and social environment of MAS, the database for land cover spatial-temporal evolution simulation, and a key element of the model [51, 52]. In this model, environmental factor layer is defined as an integral body including the status of initial land cover, elevation, slope, soil, topography, distance to road, distance to river, and other environmental factors.

3.3.2. Definition of Roles and Conduct Rules. A key issue of multiagent model construction is how to abstract and

descript agents properly [53]. Analysis of study area land cover reveals the driving force of regional landscape changes that are caused by human activities. Therefore, the simulation of landscape spatial-temporal evolution using MAS is to link up human activities and agents based on multi-agent characteristics. According to scenario analysis and design results, three agent types are defined as, farmer agent, protector agent and government agent.

Protector Agent. Protectors who are the staff of national nature reserves in this research prevent farmers from agricultural developing. Protector agent becomes the main driving force slowing down wetland landscape degradation with the support from government agent. The protective efforts of the protector agent are directly reflected on wetland area changes in a specific period of time. The wetland area reduces enormously and quickly, and it is indicative of a poor effect of protection and a small effect on wetland protection. Thus, an equation assessing the protective effect can be quantitatively expressed as

$$P_{\text{omit}} = \frac{A^{t1}_{\text{marsh}} - A^{t2}_{\text{marsh}}}{A_{\text{marsh}}} \times 100\%, \quad (1)$$

where P_{omit} is the probability that omits hindering agriculture development, A^{t1}_{marsh} is the wetland area at time $t1$, A^{t2}_{marsh} is the wetland area at time $t2$, and A_{marsh} is the reserve total area.

Formula (1) will be used to assess the protect effect on Honghe and Sanjiang National Nature Reserves separately. When land cover change position (i, j) is within those reserves, the omit probability will be calculated using this equation. The omit probability will be one hundred percent if the position is out of the reserves. In such case, protector's activity can be expressed as

$$P_{\text{pro}}(i, j) = \begin{cases} P_{\text{omit}}(i, j) & (\text{within reserves}), \\ 1 & (\text{out of reserves}). \end{cases} \quad (2)$$

Farmer Agent. The behaviors of farmer agent can be classified into two categories: development and undevelopment. This behavior will cause two types of possible results: one is negative behavior that can reduce the wetland area, and the other is a positive activity that does not change the land cover. In reality, farmers perform such activities under the approval of government. In this model, one part of the farmers' behavior is carried out directly (illegal development), while the other part applies for government agent approval selectively. The whole behavior is also affected by protector agent. It must get the approval of government and avoid the hindering effect from protector agent. As a result, the land cover status can be changed eventually.

When the model starts running, the farmers' probability of reclaim wetlands will be calculated by the formula below

$$P_{\text{far}}(i, j) = \left(w_1 E_{\text{eleva}}, w_2 E_{\text{slop}}, w_3 E_{\text{soil}}, w_4 E_{\text{landf}}, w_5 E_{\text{rivd}}, w_6 E_{\text{road}}, w_7 R_{\text{disb}}\right), \quad (3)$$

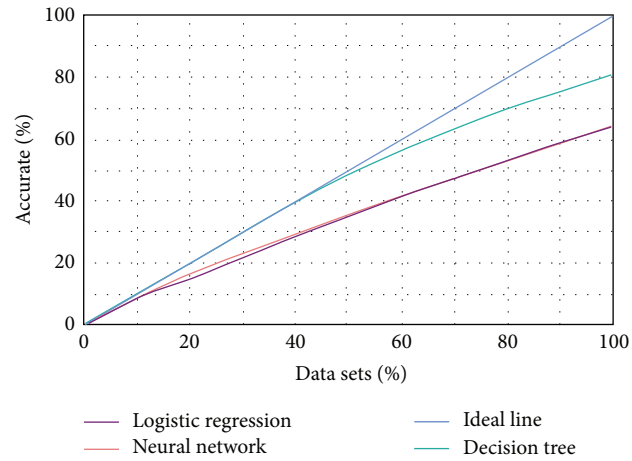

Figure 4: The lift map of data mining.

where $P_{\text{far}}(i, j)$ is the change probability for position (i, j); E_{eleva}, E_{slop}, E_{soil}, E_{landf}, E_{rivd}, E_{road}, and R_{disb} are the environment factors of elevation, slope, soil, landform, distance to river, distance to road, and random disturbance. These indicators were selected because they are representative of the most critical environmental issues of the study area, and they are easy to understand and communicate. $w_1, \ldots, w_6$ is the influence weight for each factor, which is calculated through data mining method. This instance completed the calculation of farmland development probability using Microsoft SQL Server 2008 software, data mining process used neural network, decision tree, and logistic regression mathematical model, respectively, and results of them described by lift chart (Figure 4). A lift chart is used for comparing the accuracy of each prediction model. The x-axis represents the percentage of the test data set for prediction, and the y-axis indicates the percentage of accurate prediction. An ideal line is a diagonal line, which means 50 percent of the data accurately predicted 50 percent of the cases (the expected maximum). Then we found that decision tree method was the best, and then it was used to calculate the probability of land use change under the influence of various geographical conditions.

The random disturbance is expressed as

$$R_{\text{disb}} = 1 + (-\ln \theta)^{\alpha}, \quad (4)$$

where θ is the random number between 0 and 1; α is the parameter that controls the size of random disturbance; weight w_7 for it is set as 1 [54].

Government Agent. Government achieves its own wish through the planning actions, while residents affect the probability of land cover conversion through the cooperation with the government [55]. The behavior of the government agent at the area includes the government's macromanagement, action planning, and decision making in response to farmer agent's application [56]. When farmers apply to convert wetlands for agricultural purpose, government agent will make decisions based on current land cover status and future planning of utilization. And degree of support from government also indirectly determines the specific action of

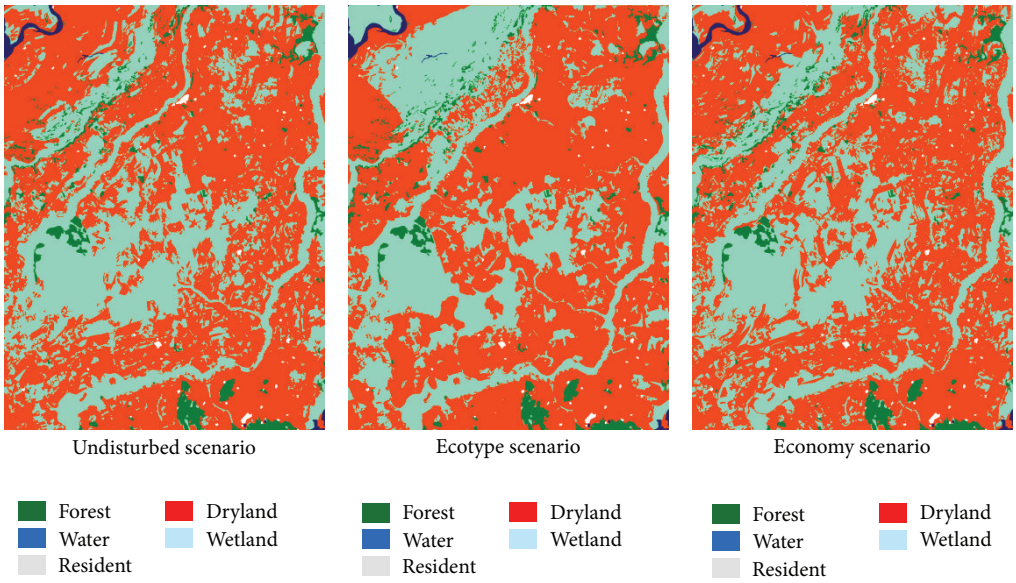

Figure 5: The results of scenarios planning based on multi-agent adjustment.

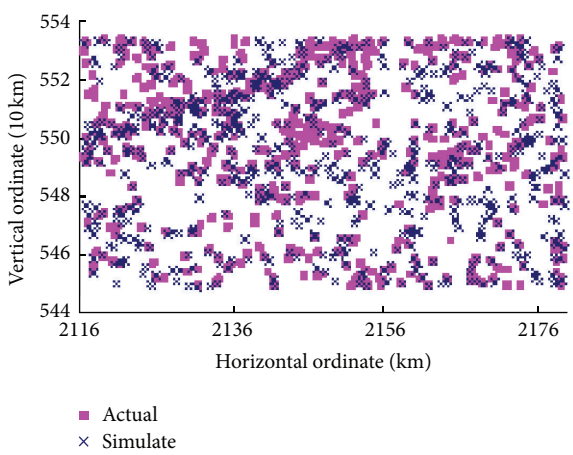

Figure 6: The spatial distribution of centroids.

protector. A strongly support will obtain an excellent performance of protection and vice versa. Then the behaviors of the government agent can be expressed through the activities of farmer and protector indirectly as

$$P_{\text{gov}}(i, j) = \left|1 - \left(aP_{\text{pro}}(i, j) + bP_{\text{far}}(i, j)\right)\right|, \quad (5)$$

where $P_{\text{gov}}(i, j)$ is the probability that government approves to change land cover for position (i, j); a is the adjustment coefficient for protector between 0 and 1; b is the adjustment coefficient for famers between 0 and 1; coefficients are varied with the government policy changing.

Different scenarios are realized through revising 3 kinds of agents, and the details are shown in Table 2.

3.4. Simulation Results. Based on the characteristics of three planning scenarios, combined with the framework of interaction relationships between governments, protectors, and farmers variables (Figure 4), MAS model is used to generate the different planning scenarios (Figure 5).

3.5. Assessment of Accuracy. In order to facilitate observation of the differences between simulation results and reality situations, centroids of each landscape patch including undisturbed scenario simulation results and reality situations of year 2006 are calculated. The spatial distribution of them is plotted, which is shown in Figure 6.

The centroids distribution plots show that simulation results of undisturbed scenario have a high degree of consistency with 2006 year's actual situation. For the quantitative evaluation of the simulation accuracy, study overlaps the simulation result map of undisturbed scenario and the actual land cover map of 2006 together to gain the simulation accuracy of points to points. The overall simulation accuracy of undisturbed scenario reaches 85.12%, and it validates the feasibility and effectiveness of land cover change simulation using MAS. At the same time, it assures the scientific creditability of the other two planning scenarios' simulation results based on SA method and MAS technique integration.

4. Discussions and Conclusions

Simulation results show that three different scenarios show vast differences, especially that the eco-type scenario is significantly different from the other two scenarios, which means that the current ecological conditions of the study area are not ideal, and that adjustment and optimization work should promptly be carried out to protect wetlands. The differences between economy scenario and undisturbed scenario are relatively small, but obvious differences still exist on closer inspection. Wetland patches distribution is loose in

Table 2: Realization of scenarios through agent parameter adjusting.

Scenarios	Protectors	Famers	Governments	Illustrations
Undisturbed	P_{pro}	P_{far}	P_{gov}	All the parameters keep unchanged
Eco-type	$1/3 * P_{pro}$	$1/3 * P_{far}$	$1/3 * P_{gov}$	Governments reduce the rate of approvals, protectors reinforce supervision, and famers reduce their cultivation will.
Economy	$3 * P_{pro}$	$3 * P_{far}$	$3 * P_{gov}$	Governments increase rate of approvals, protectors reduce supervision, and famers increase willingness to reclaim.

economy scenario, and many ecologically significant wetland patches are disappeared.

According to the simulation results of different scenarios, depending on the development goals, different planning strategies can be gained. Furthermore, the similarities and differences between actual situations and simulation scenarios can be used to assist land use optimization problem, even to provide reference for landscape reconfiguration including wetland sustainable development, returning farmland to wetland, and so forth.

To sum up, a theoretical framework was proposed for the land-use planning based on the integration SA method with MAS simulation. Taking Sanjiang Plain inland freshwater wetland areas as an example, the study verified the availability of this framework. Results showed that MAS simulation technique emphasizing quantitative process effectively compensated for the SA method emphasizing qualitative process, which realized the organic combination of qualitative and quantitative land-use planning work and then provided a new idea and method for the land-use planning and sustainable managements of land resources.

Applying SA method originated in enterprise management and MAS technique from the field of artificial intelligence into the complex geographical system problem, there is still a lot of refinement work that needs to be further completed. How to establish criteria for the classification of different planning scenarios in SA more scientifically based on regional characteristics and how to define the simulation variables and their interaction relationships based on land-use planning objective more accurately still need to be explored in further practical work.

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