Chin. Geogra. Sci. 2010 20(4) 318–329
DOI: 10.1007/s11769-010-0404-6

Estimation of Ecological Water Requirements Based on Habitat Response to Water Level in Huanghe River Delta, China

CUI Baoshan[1], HUA Yanyan[1], WANG Chongfang[1], LIAO Xiaolin[1], TAN Xuejie[1] , TAO Wendong[2]

(*1. State Key Joint Laboratory of Environmental Simulation and Pollution Control, School of Environment, Beijing Normal University, Beijing 100875, China; 2. Department of Environmental Resources and Forest Engineering, College of Environmental Science and Forestry, State University of New York, NY 13210, USA*)

Abstract: In recent years, wetland ecological water requirements (EWRs) have been estimated by using hydrological and functional approaches, but those approaches have not yet been integrated for a whole ecosystem. This paper presents a new method for calculating wetland EWRs, which is based on the response of habitats to water level, and determines water level threshold through the functional integrity of habitats. Results show that in the Huanghe (Yellow) River Delta water levels between 5.0 m and 5.5 m are required to maintain the functional integrity of the wetland at a value higher than 0.7. One of the dominant plants in the delta, *Phragmites australis*, tolerates water level fluctuation of about ± 0.25 m without the change in wetland functional integrity. The minimum, optimum and maximum EWRs for the Huanghe River Delta are 9.42×10^6 m^3, 15.56×10^6 m^3 and 24.12×10^6 m^3 with water levels of 5.0 m, 5.2 m and 5.5 m, corresponding to functional integrity indices of 0.70, 0.84 and 0.72, respectively. A wetland restoration program has been performed, which aims to meet these EWRs in attempt to recover from losses of up to 98% in the delta's former wetland area.

Keywords: water level-habitat response; functional integrity; ecological water requirements; wetland; Huanghe River Delta

1 Introduction

Ecological water requirements (EWRs) describe the water regimes needed to sustain the ecological values of water-dependent ecosystems at a low level of risk (Tharme and King, 1998; Smakhtin *et al*., 2004). Information about EWRs will provide wetland managers with increased inference about the best use of water resources and will provide a better understanding of hydrological processes (Wilcox *et al*., 2006). It allows us to make well-informed decisions in the process of management of wetlands, leading ultimately to the improvement of maintenance of ecosystems and thus to sustainable development of wetlands.

Early studies on EWRs have mainly been focused on instream flows. Most of them addressed the relationships between river discharge and fish inhabitation, and indicator species of aquatic organisms and river flow (Smakhtin, 2001; Middleton, 2002; Duvail and Hamerlynck, 2003; Hughes, 2005; Yang *et al*., 2005). The approaches of assessing EWRs include look-up table method, biological-hydraulic analysis method and simulation of biological responses. Simulation of biological responses is regarded as the best potential methodology among them since it couples river flow with habitat use and biological characteristics (King and Louw, 1998; Hughes, 2001; Robertson and James, 2002; Duvail and Hamerlynck, 2003). However, none of them can be directly applied to calculating EWRs for wetlands due to different water regimes, such as duration, frequency, water depth, timing and extent of flooding (White *et al*., 2007; Powell *et al*., 2008) compared with instream ecosystem.

Water regimes have been shown to affect the composition, diversity and distribution of macrophyte communities in wetlands (Casanova and Brock, 2000; Nicol *et*

Received date: 2009-12-17; accepted date: 2010-04-28

Foundation item: Under the auspices of Major State Basic Research Development Program of China (No. 2006CB403303), National Natural Science Foundation of China (No. U0833002, 40571149), Scientific Research Foundation of Beijing Normal University (No. 2009SD-24)

Corresponding author: CUI Baoshan. E-mail: cuibs@bnu.edu.cn; cuibs67@yahoo.com

al., 2003; Kennedy *et al.*, 2003; Boar, 2006). All coastal or shoreline wetland plant communities have one or more bird habitats influenced by water depth and flood duration (Desgranges *et al.*, 2006). Slight increases in the water level of wetlands could progressively delay the egg-laying dates of birds (for example, *Chlidonias hybrida*) up to 40–50 days in relation to late appearance of suitable aquatic biomass (Pallisson *et al.*, 2006). Although the hydrological conditions greatly influence the nature of plant growth and bird distribution in wetlands, it is unknown how hydrology interacts with the other features of plant growth, e.g. nutrient and energy pathways, germination and establishment (Wolanski *et al.*, 2004; Finlayson, 2005; Finlayson *et al.*, 2006). The currently available methods are the water level method and functional method (Krstolic *et al.*, 2006). The former method is based on historical records to establish the optimal water levels, and strongly depends on hydrological data, while neglects ecological data, weakening its feasibility. The latter method is mainly based on such ecological functions of wetlands as water quality protection, fish and wildlife protection, and recreation of wetlands. For each function, a quantitative value of EWRs is needed to reach its desired state, which indicates the need for some foresight and commitment from policy makers and managers as to the health condition, in which an ecological system should be maintained. However, the desired state should be an integrated one, which is known as functional integrity, representing the intactness of environmental, ecological and productive functions.

Therefore, a growing field of research is dedicated to examining a new method that integrates hydrological condition into ecosystem for calculating EWRs of wetlands. Some researchers once recommended a group of physical, chemical and biological indicators, including water depth, area and salinity, as well as a variety of other components of wetlands, such as zooplankton, birds, fish, mammals and fringe vegetation, for monitoring wetland responses to changes in water management practices (Allan and Lovett, 1997; Hayashi and van der Kamp, 2000; Reid and Brooks, 2000). However, these indicators were not rationally integrated into an entire ecosystem to assign uniform water requirements for wetland management.

The objective of this paper is 1) to simulate the responses of wetland habitats to different water levels by analyzing the relationship between the spatial distribution and ecological characteristics of plant communities and water depth; and 2) to determine the reasonable thresholds of water level and its corresponding EWRs based on functional integrity assessment, and then to achieve satisfactory tradeoffs in water allocation among wetland functions. Based on different purposes in different regions, EWRs have been expressed in terms of water level or water depth (Froend and Loomes, 2004; 2006), discharge and flow or volume (Hughes, 2005; Smakhtin and Eriyagam, 2008; Sun *et al.*, 2009). In this study, to make management more convenient and practical we used water volume to express EWRs, so that the wetland managers can hold the desired EWRs of the wetlands by controlling the water volume flowing into the restored area of the Huanghe River Delta.

2 Materials and Methods

2.1 Study area

The Huanghe (Yellow) River Delta Nature Reserve (37º35′–38º12′N, 118º33′–119º20′E) is located in the estuary of the Huanghe River in Dongying City, Shandong Province, China. It has a warm temperate continental monsoon climate with distinctive seasons and a rainy summer. It has an average annual temperature of 12.1℃, frost-free period of 196 days, average annual precipitation of 551.6 mm, and average annual evaporation of 1 962 mm. The annual runoff of the Huanghe River decreased since the 1980s, and reached its maximum and minimum value of 49.1×10^9 m^3 and 10.0×10^9 m^3 in 1983 and 2002, respectively. After 2002, due to scientific management strategies of the Huanghe River, its runoff stopped decreasing and increased to around 20.0×10^9 m^3. As a result, the perennially waterlogged area decreased, while the seasonally waterlogged one increased. Currently, the area of the perennially and seasonally waterlogged wetlands is 210 274 ha and 123 153 ha, accounting for 63.06% and 36.94% of the total area of the Huanghe River Delta wetlands, respectively. The topography features fluvial lowland due to riverbed siltation and fluvial deposition. The terrain is flat with an elevation of 3–6 m above sea level. Its main soil type is alluvial soil. The dominant vegetation is composed of herbaceous species such as *Lepiironia articulata*, *Phragmites australis*, *Suaeda heteroptera*, and shrub species *Tamarix chinensis*.

The Huanghe River Delta is an important wintering and breeding site for migratory birds in the Northeast Asian Inland and the Western Pacific Rim. The Huanghe River Delta Nature Reserve is rich in bird species, many of which are among the first class of state key protected birds in China and endangered birds all over the world, such as *Ciconia boycia* and *Grus japonensis*. With the objective of improving wetland functions and protecting the natural habitat for rare birds, a project was implemented by Dawenliu Management Station to restore 26.50 km^2 of wetlands in the Huanghe River Delta Nature Reserve in July 2002 (Cui *et al*., 2009). The project covers an area (central geographic coordinates at 37°45'48"N and 119°03'07"E) extending 4 km south from the current channel of the Huanghe River and 15 km west from the estuary, which is the site of this study (Fig. 1) (Cui *et al*. 2009). Q section, where water level records ranged from 4.4 m to 5.8 m during 1976–2001, is only the inflow channel, by which freshwater is directed to the wetlands. At the same time, considering that there are not water level records for monitoring area, we assumed that the simulated range of water level is consistent with the records in the paper.

2.2 Monitoring and surveying methods

Hydrological and ecological monitoring was carried out in the study area from May to October in 2005. Monitoring and sampling were performed along six transects of 1800 m in length (Fig. 1). Within each transect four 1 m × 1 m sampling sites in inundated site and six wells in dry ground with approximately interval of 200 m were arranged along gradients of water depth. Three sampling spots were selected randomly in each inundated sampling site, and water depth was measured with a GSK-4 Portable Digital Ultrasonic Sounder with 0.05 m in precision for each spot positioned by GPS once a month. Simultaneously, vegetation data, including coverage, height and density of herb species and dominant species, were also measured once a month. Considering the slight temporal variations in water depth and plant community characteristics, field measurements were averaged for corresponding monitoring spots during the seasons measured. Water tables were also measured in six wells for each transect randomly chosen once a month. Both water depths and water tables in six transects were then converted to 60 water levels in total by ground and water surface elevations from available DEM (data of digital elevation model) with a contour interval of 0.1 m provided by the Administration of the Huanghe River Delta Nature Reserve.

Bi-weekly waterbird counts were conducted by using the line transect method. The line transect was 15 km in length along the study area. Field surveys were conducted between 6 AM and 11 AM with 20–60× spotting scope and 80× binoculars. *Ciconia boyciana* was particularly selected as an indicator species observed at the spots where they frequently visited.

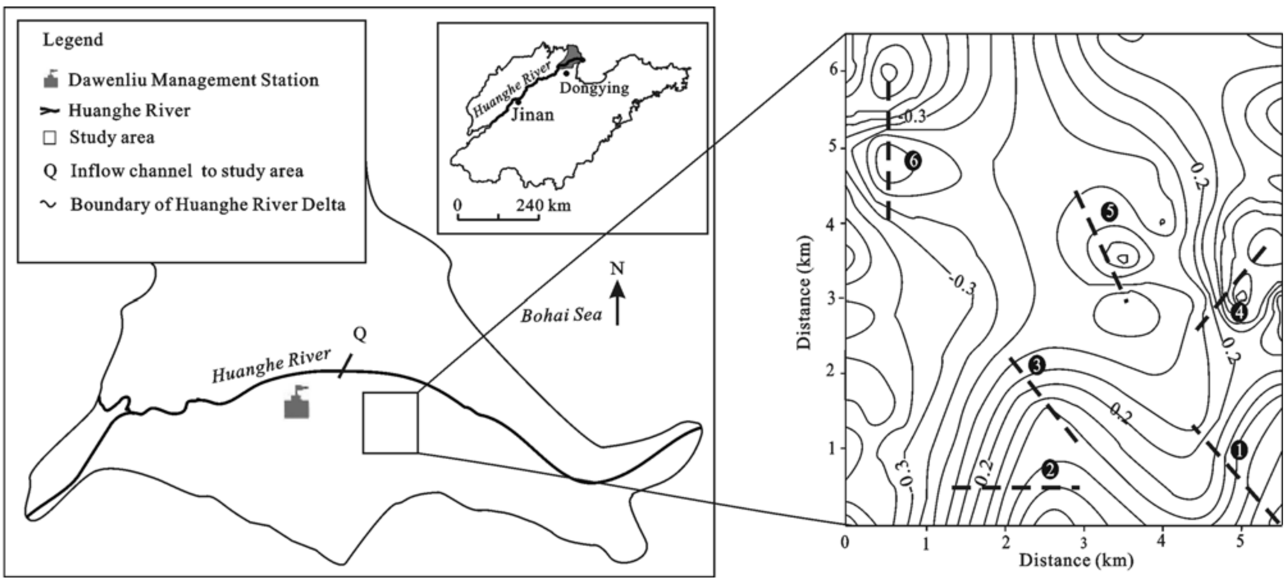


Fig. 1 Location sketch of study area and its contour map with six transects (❶, ❷, ❸, ❹, ❺, ❻)

2.3 Assessment of functional integrity

In this paper, the assessment of functional integrity for wetlands was based on the wetland habitat simulation at different water levels. Water levels within the thresholds were supposed to maintain wetlands in the best quality or comply with ecological management goals. For assessing the functional integrity for wetlands, we first divided the fundamental functions of wetlands into three evaluation elements, i.e. environmental, ecological and productive functions. Each element was normalized to eliminate the effects of different dimensions of original data and five indicators were selected. The weights of the indicators were determined by using the Analytic Hierarchy Process (AHP) method. Then an indicator system was established to include more details for each function (Table 1).

Functional Integrity Index (*I*) of wetlands at different water levels can be calculated by the following equation:

$$I = \sum_{i=1}^{n} I_i \mu_i \quad (1)$$

where I_i is the standardized value of indicator *i*, μ_i is the weight of indicator *i* (i = 1, 2, …, *n*, and *n* is the total number of wetland functions).

Area Proportion Index (*W*), which is an index to express the percentages of areas occupied by freshwater and salt water, can be expressed as:

$$W = 1 - W_s / W_t \quad (2)$$

where W_s is the salinized area in dry ground without flooding, which can be calculated by simulated diagram at various water levels; W_t stands for the total area of a wetland.

Habitat Suitability Index (*E*) represents the habitat suitability for rare waterfowls of wetlands. The higher the index, the more suitable the habitat is for rare birds. *E* is expressed as follows (Legendre *et al*., 2002):

$$E = \begin{cases} (S/S_0)(P_0/P)^m \sum_{i=1}^{n} A_i \lambda_i & (P > P_0) \\ \sum_{i=1}^{n} A_i \lambda_i & (P \le P_0) \end{cases} \quad (3)$$

where *S* is the total area of a specified plant (*Phragmites australis* in this study) and open water surface at a certain water level; S_0 represents the total area of the specified plant and open water at the base water level (4.4 m in this study); *P* represents the total number of specified plant patches and water surface patches at a certain water level; P_0 equals the total number of specified plant patches and water surface patches at the base water level; A_i denotes the area percentage of habitat *i* at different water levels, i = 1, 2, …, *n*, and *n* is the total number of habitats; *m*, is a constant; and λ_i is the suitability coefficient of habitat *i*.

In Equation (3), *m* can be expressed as follows (Lin *et al*., 2005):

$$m = \frac{\lg\left(\frac{k_0 S}{k S_0}\right)}{\lg\left(\frac{P_0}{P}\right)} + 1 \quad (4)$$

where k_0 and *k* are the numbers of the species (*Ciconia boycia* in this study) that the habitat can carry at the base water level and a certain water level, respectively.

λ_i can be quantified between 0 and 1 by surveys of bird species and species numbers in four different habitats, which are divided by the vegetation types (Table 2).

Biodiversity Index (*H*) expresses the potential ability of wetlands to maintain biodiversity. The higher the index, the higher the capacity of the wetlands to maintain plant biodiversity. The *H* is formulated (Bian and Zhang 2000) as:

$$H = -\sum_{i=1}^{n} (A_i\beta_i / \sum A_i\beta_i) \ln(A_i\beta_i / \sum A_i\beta_i) \quad (5)$$

Table 1 Indicators and weights for assessing functional integrity by AHP method

Evaluation element	Weight of element	Function	Indicator	Weight of indicator
Environmental function	0.23	Water-salt balance	Area Proportion Index (*W*)	1.00
Ecological function	0.53	Habitats for rare waterfowls	Habitat Suitability Index (*E*)	0.66
		Biodiversity maintenance	Biodiversity Index (*H*)	0.34
Productive function	0.24	Raw materials	*Phragmites australis* Yield Index (*B*)	0.50
		Water entertainment	Proportion of Deep Water Area (*P*) (> 0.5 m)	0.50

Table 2 Suitability coefficients of different habitats (λ_i)

Habitat	Frequency of bird occurrence	Number of individuals	λ_i
Non-inundated area with *Phragmites australis*	6	24	0.185
Shallow water area with *Phragmites australis*	15	98	0.754
Deep water area (> 0.50 m)	2	6	0.046
Non-inundated area with other plants communites	1	2	0.015

where A_i is the area percentage of habitat i at different water levels (i = 1, 2, …, n); β_i is the biodiversity index of Whittaker β of habitat i at different water levels. Whittaker β_i is expressed as follows:

$$\beta_i = S/ma_i - 1 \quad (6)$$

where S is the total number of species in the study area, and ma_i is the number of species in habitat i (Whittaker, 1960).

Phragmites australis Yield Index (B) can be expressed by the percentage of *Phragmites australis* area as follows:

$$B = \sum_{i=1}^{n} A_{pi} \quad (7)$$

where A_{pi} is area percentage of *Phragmites australis* in habitat i, i = 1, 2, …, n, and n is the total number of habitats with *Phragmites australis*.

The water entertainment function is a direct use value of wetlands. It can be developed where water depth is greater than 0.5 m, as only deep water can be sprayed, projected, directed, and divided into jets or drops to create an incredibly enchanting landscape. The Proportion of Deep Water Area (P) can be described by the following equation:

$$P = S_d / S_w \quad (8)$$

where S_d is the deep water area in wetlands, and S_w is the total area of wetlands.

The final step is to classify wetland states based on the values of the functional integrity index. No unified methods have been approached for ecological assessment other than the ecosystem integrity (Boyce and Elison 2001; Wolfgang, 2004). With consideration to the special geographic and ecological characteristics of the study area as well as some national and international research methods (Dungan *et al*. 2002; Ulrich *et al.* 2004), we develop a system to classify wetland states. When the functional integrity index is below 0.5, wetland ecosystem is in poor quality and wetland functions are poorly performed; when the index is between 0.5 and 0.7, wetland ecosystem is in medium quality while wetland functions are performed on average; and when the index is higher than 0.7, wetland ecosystem is in good quality while wetland functions are perfectively performed. So the lowest and the highest water levels with the index value higher than 0.7 were selected as the calculation bases of the minimum and the maximum EWRs that maintained the desired wetland functions.

2.4 Water level-habitat simulation method

Both animals and plants are very sensitive to the influence of hydrological processes, and too much water or a serious shortage will threaten the existence of certain species (Jean *et al*., 2006). The water level-habitat response simulation method associated with functional integrity determines the thresholds of water levels and it was performed as follows: 1) preparing the DEM with a contour interval of 0.1 m; 2) creating a GRD surface model file in Surfer by spline interpolation (Zhang and Takeuchi, 2004) with the 60 mean annual water levels, and extracting the trend surface of water level by converting Surfer GRD to ArcGIS GRD; 3) obtaining water depths corresponding to certain water levels (4.4–5.8 m a. s. l.) by subtracting topography from water table map; 4) simulating the spatial distribution of plant communities at different water levels based on the relationship between water depth and plant community characteristics.

2.5 Calculation of EWRs

A specific quantitative relationship between water level and its corresponding water volume in wetlands was determined by the landform and structure of wetlands. Because natural conditions and dominant organisms differ from region to region, site-specific correlations between EWRs and water level have to be determined. By linking water level data with DEM data, this study derived a series of water volumes corresponding to water levels (water depths), and then determined the EWRs according to the threshold of water levels (water depths) as follows (Hayashia and van der Kamp, 2000):

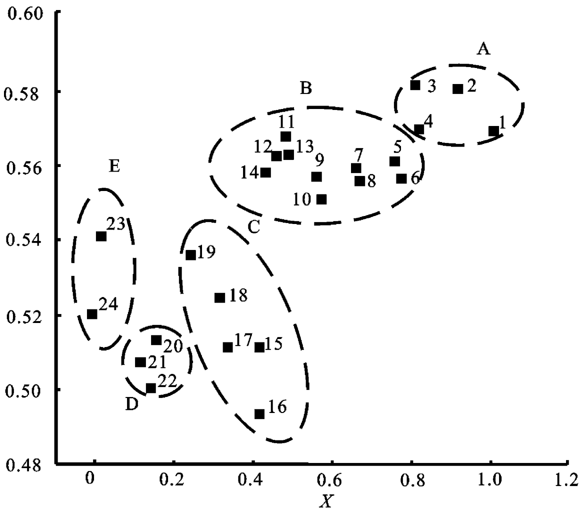
$$EWRs_j = \sum_{i=1, j=1}^{i=n, j=m} S_t B_{ij} \overline{H}_{ij} \quad (9)$$

where $EWRs_j$ is the ecological water requirements of water level j; S_t is the total study area; B_{ij} is the water area percentage of habitat i at water level j; $\overline{H}_{ij}$ is the mean water depth of habitat i at water level j, which can be acquired from the simulated figure.

3 Results

3.1 Ecological characteristics of plant communities under different water depths

We recorded 14 herbaceous species with a coverage of over 1% within all of the 24 sampling sites (Table 3), which constituted a plant data matrix of 14 × 24 dimensions. Using Fuzzy Set Ordination, the resulting ordination graph of plant communities in response to water depth was created (Fig. 2).



1–24 denote sampling sites

X-value represents relative water depth, which is defined as gradient variation of water depth and increases from left to right; Y-value reflects community types corresponding to specific water depth from bottom up

Fig. 2 Ordination graph of plant communities to water depth

Figure 2 and Table 3 indicate that the spatial pattern of the regional wetland communities is controlled by water depth. Most plant species can not survive except for a few algae in the sampling sites with water depths over 0.7 m, where the plant coverage was also small. With the decline of the water depth, the coverages of *Typha orientalis* and *Phragmites australis* increase. In the sites with water depths of 0.55 m, *Typha orientalis* flourishes and its coverage increases to the peak value. However, further decrease of water depth makes the coverage of *Typha orientalis* decrease again. In the sites with a water depth of 0.45 m, the plant community underwent a transition from *Typha orientalis* to *Phragmites australis* with a smaller coverage. The transitional zone from water to land ranges from the water depth of –0.3 m to 0.4 m, in which both xeromorphic and aquatic vegetation co-exist with highest plant coverage. In the sites with water depths between –0.5 m and –0.3 m, salt crusts appear in most places due to salt accumulation induced by the lower ground water level. The most severe salinization even makes some places bare in those sites. In the sites with water depths lower than –0.5 m, arid environments appear, plant types are replaced by drought-resistant plants and the soil salinity is slightly mitigated due to getting rid of salting stain, all of which are due to insufficient water supply.

Figure 2 shows that sampling sites are clustered in five regions, with different plant communities in the study area. Plants of Type A and Type B occupy the regions with water depths above 0 m, where *Phragmites australis* and *Typha orientalis* are the dominant species of the sampling sites with water depths of 0–0.4 m and 0.4–0.7 m, respectively. The other three types of plants are distributed in the regions with water depths below 0 m. *Phragmites australis* and *Suaeda heteroptera* are the dominant species of Type C (with the water depth from –0.3 m to 0 m) and Type D (with the water depth

Table 3 Herbaceous species with projected coverage of over 1% within 24 sampling sites

Type	Inclusive sampling sites	Dominant species	Community composition	Water depth (m)
A	1, 2, 3, 4	P7	P1, P7	0.4–0.7
B	5, 6, 7, 8, 9, 10, 11, 12, 13, 14	P1	P1	0–0.4
C	15, 16, 17, 18, 19	P1	P1, P2, P3, P4, P5, P6, P8, P10, P12, P14	-0.3–0
D	20, 21, 22	P2	P1, P2, P3, P6, P10	−0.5–−0.3
E	23, 24	P9, P13	P1, P2, P4, P6, P8, P9, P13	−0.6–−0.5

Notes: P1, *Phragmites australis*; P2, *Suaeda heteroptera*; P3, *Setaria viridis*; P4, *Aeluropus sinensis*; P5, *Artemisia annua*; P6, *Limonium sinense*; P7, *Typha orientalis*; P8, *Cynanchum sibiricum*; P9, *Imperata cylindrica*; P10, *Sonchus oleraceus*; P11, *Cyperus microiria*; P12, *Chrysanthemum indicum*; P13, *Triarrhena sacchariflora*; P14, *Glycine soja*

From –0.5 m to –0.3 m), respectively. Type E (with the water depth from –0.6 m to –0.5 m) is mainly composed of xeromorphic vegetation with the dominant species of *Imperata cylindrica* and *Triarrhena sacchariflora*.

The Whittaker biodiversity index (β) also changes dramatically with water depth in the study area (Fig. 3). The value of β is small in the sites with water depths between 0.1 m and 0.3 m occupied by plants of Type B, while highest in the sites with water depths of about –0.3 m occupied by plants of Type C, a community dominated by *Phragmites australis*.

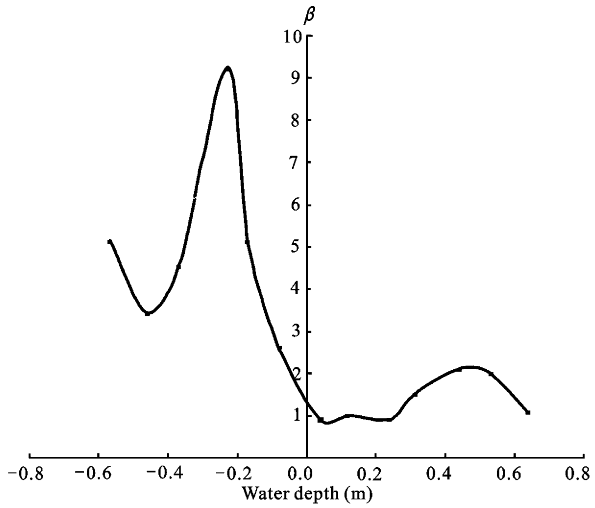


Fig. 3 Relationship between water depth and Whittaker biodiversity index (β)

Regression analysis shows that there is a single-peak trend of *Phragmites australis* coverage relative to water depth, which is expressed by the following regression equation (Fig. 4a):

$$y = -2.4052x^2 + 0.0066x + 0.6993, R^2 = 0.723$$

As shown in Fig. 4a, the peak value of coverage occurs at the depth of 0 m or so. However, the coverage decreases gradually when the water depth becomes higher or lower.

The regression curve of *Phragmites australis* height and water depth is given in the following equation (Fig. 4b):

$$y = 18.189x^4 - 1.6601x^3 - 8.856x^2 + 1.3137x + 2.579, R^2 = 0.819$$

It is indicated that the *Phragmites australis* height reaches the highest value at the water depth of approximately 0.05–0.10 m, which basically matches the depth of 0 cm corresponding to the peak coverage. At the water depth of approximately –0.4 m, *Phragmites australis* height was the lowest due to soil salinization.

3.2 Water level-habitat simulation to determine EWRs

A horizontal zonation of aquatic plants and water bodies along water depth provides different habitats for birds and animals. Observations in this study show that there are more *Ciconia boycia* in the shallow water area with *Phragmites australis*, and less in the deep water area and the non-inundated area with other plants (Table 2). Based on water level-habitat simulation results, four habitats varied significantly with water level (Fig. 5). There are almost no deep water areas and less at the simulated water levels of 4.4 m and 4.5–4.8 m, respectively. Shallow water area or non-inundated area with *Phragmites australis* are found in all kinds of simulated water levels.

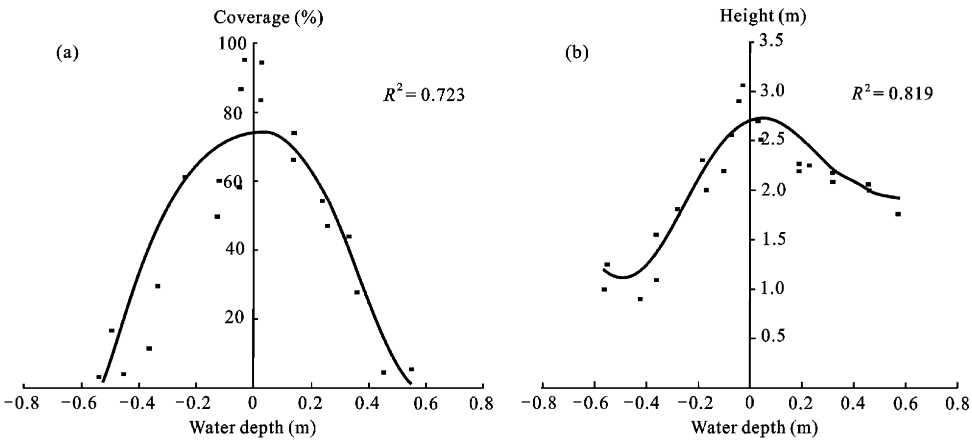


Fig. 4 Relationships between water depth and coverage (a) and height (b) of *Phragmites australis*

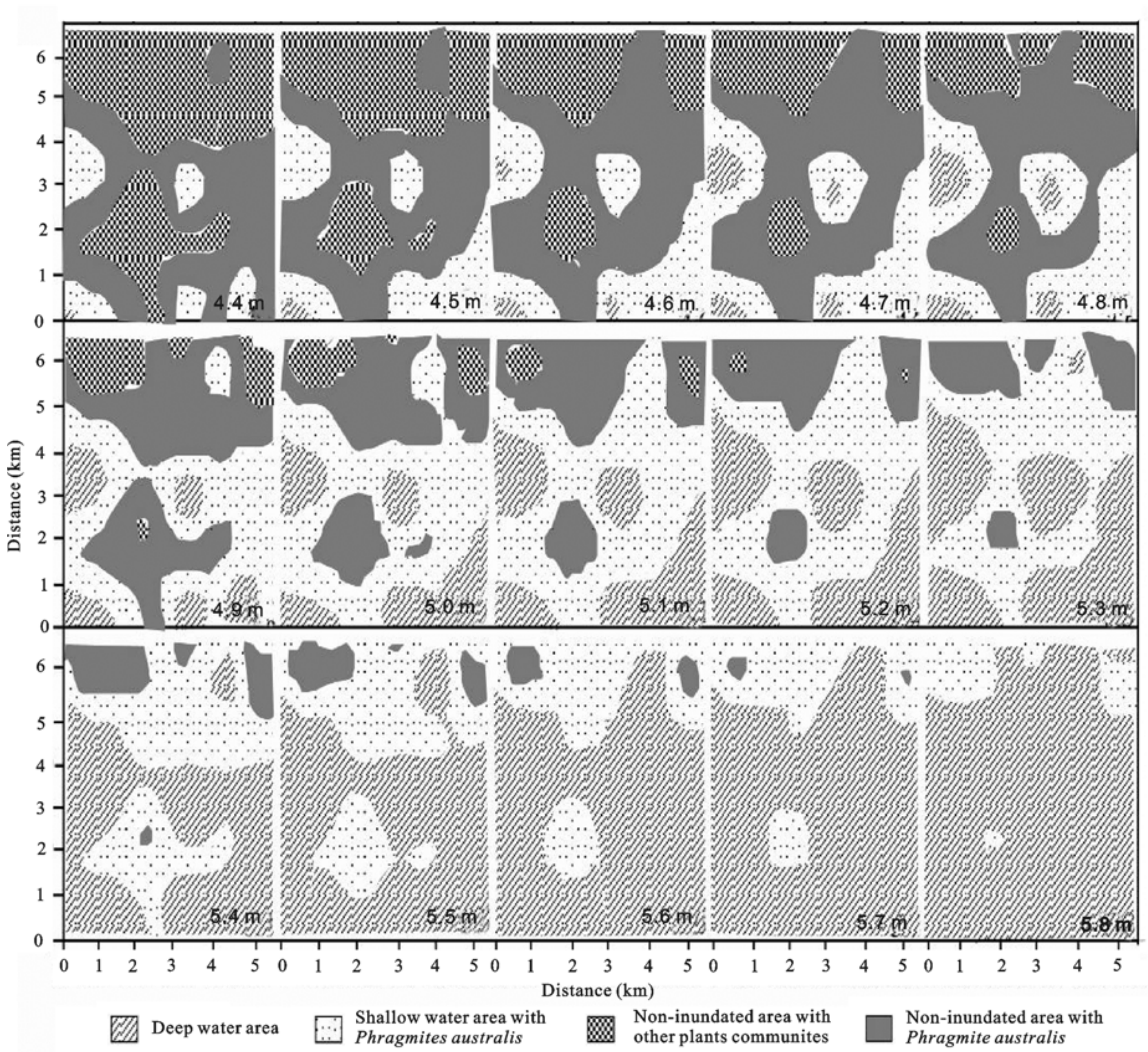


Fig. 5 Spatial distribution of various habitats simulated for various water levels

According to equations (2), (3), (5), (7) and (8), each function index at different water levels was calculated and standardized. The functional integrity index at different water levels was subsequently calculated with Equation (1) (Table 4). The functional integrity index higher than 0.70 can be considered as a symbol of good-quality wetland with theoretically optimum function and water level threshold of 5.0–5.5 m. Moreover, the functional integrity index reaches the peak value at 5.2 m of water level and has no significant drop when water level fluctuates in a range of ±0.2 m. Calculated by Equation (9), the water volumes of 9.42×10^6 m^3, 23.48×10^6 m^3, and 15.56×10^6 m^3 at the water levels of 5.0 m, 5.5 m, and 5.2 m respectively are the minimum, maximum and optimum EWRs to keep the wetland ecosystem in suitable and desired conditions (Table 5).

4 Discussion

4.1 Influence of water depths on plant communities

In wetlands, water depth directly influences the distribution, growth and suitability of specific species (Jackson and Colmer, 2005; Smith and Brock, 2007; Laitinen *et al*., 2008). Our study results showed that the biodiversity and coverage of macrophyte communities dominated by *Phragmites australis* reach their peaks at the water depths of –0.30 m and 0 m, respectively, and decrease either above or below that depth. Chow-Fraser (2005) also mentioned that there was a threshold water depth for emergent vegetation in a wetland ecosystem, above or below which the percentage cover was kept low or disproportionate effect. The difference of water depths is inferred responsible for a difference in the ri-

Table 4 Normalized index values and functional integrity index values at different water levels

Water level (m)	Area Proportion Index (*W*)	Habitat Suitability Index (*E*)	Biodiversity Index (*H*)	*Phragmites australis* Yield Index (*B*)	Proportion of Deep Water Area (*P*)	Functional Integrity Index (*I*)
4.4	0.11	0.31	0.73	0.68	0.00	0.35
4.5	0.22	0.48	0.77	0.86	0.01	0.46
4.6	0.25	0.56	0.72	0.96	0.01	0.50
4.7	0.31	0.58	0.79	0.92	0.05	0.53
4.8	0.37	0.62	0.71	1.00	0.07	0.56
4.9	0.50	0.62	0.87	0.92	0.13	0.61
5.0	0.65	0.69	0.96	0.86	0.20	0.70
5.1	0.73	0.98	0.96	0.82	0.27	0.82
5.2	0.79	1.00	0.97	0.78	0.32	0.84
5.3	0.85	0.83	0.98	0.68	0.42	0.79
5.4	0.91	0.83	1.00	0.56	0.55	0.81
5.5	0.94	0.58	0.94	0.42	0.67	0.72
5.6	0.97	0.48	0.78	0.32	0.78	0.66
5.7	0.99	0.38	0.52	0.22	0.89	0.59
5.8	1.00	0.27	0.31	0.12	1.00	0.51

Notes: The indices of *W*, *E*, *H*, *B, P* and *I* are gotten first from equations (1) to (8), then the highest index value in each column is considered as 1, and each value in corresponding column is divided by the highest value, respectively

Table 5 Estimated EWRs in different simulated water levels

Water level	Deep water area		Shallow water area with *Phragmites australis*		EWRs
(m)	B_{ij} (%)	$\overline{H_{ij}}$ (m)	B_{ij} (%)	$\overline{H_{ij}}$ (m)	($\times 10^6$ m^3)
4.4	0.002	0.65	0.11	0.27	1.13
4.5	0.01	0.59	0.21	0.20	1.74
4.6	0.01	0.69	0.24	0.27	2.61
4.7	0.04	0.61	0.27	0.29	3.74
4.8	0.06	0.66	0.31	0.33	5.17
4.9	0.11	0.66	0.39	0.30	6.90
5.0	0.18	0.68	0.47	0.29	9.42
5.1	0.24	0.72	0.50	0.35	12.66
5.2	0.28	0.78	0.51	0.41	15.56
5.3	0.37	0.80	0.48	0.41	17.94
5.4	0.48	0.82	0.43	0.41	20.74
5.5	0.59	0.85	0.35	0.41	23.48
5.6	0.69	0.89	0.28	0.41	26.53
5.7	0.78	0.94	0.21	0.41	29.82
5.8	0.88	0.98	0.12	0.41	33.18

Notes: Only water above ground is considered for calculating EWRs; B_{ij} is water area percentage of habitat *i* at water level *j*; $\overline{H_{ij}}$ is mean water depth of habitat *i* at water level *j*, which can be acquired from Fig. 5

chness and composition of aquatic macrophytes in this study, which is also consistent with the survey work carried out by White *et al*. (2007). This study confirms the previous studies that long-term inundation or permanently flooded conditions may directly reduce species diversity in wetlands and aquatic environments (Fortney *et al.*, 2004; Flinn *et al.*, 2008) or indirectly influence the water regime on competitive interaction (Riis and Hawes, 2002; Sim *et al*., 2006).

The peak value of *Phragmites australis* height in our study, however, is not consistent with the conclusion of Howard and Rafferty (2006), who articulated that the response of *Phragmites australis* height to water depth often changed over time, with negative effects of increased water depth appearing over longer exposure time, and the height peak occurred at the water depth of 0.2 m, which is higher than that of our study. It is likely that the salinity, which is associated with water depth, influences plant growth.

4.2 Water level-habitat simulation to determine EWRs

In this paper, EWRs, determined by water level which can deduce water depth, mainly contain the water volume of the areas with water surface. As water level simulated increases, the corresponding area percentage and mean water depth vary by different degrees. In deep water area, the area percentage and the mean water depth increase following the water level′s increase, and reach the highest values at the highest water levels simulated, respectively. While in shallow water area with *Phragmites australis,* the area percentage and mean water depth first increase until the water level of 5.2 m, and then decrease and keep the same level, respectively. The different responses to water levels are determined by the water exchange and fluidity when water level change (Riis and Hawes, 2002; Chow-Fraser, 2005). No matter how the habitats respond to the water levels simulated, the EWRs always have the increasing tendency when water levels increase. That exactly explains the determination of the water level to EWRs. However, the habitat responses which influence the functional integrity are complicated to the water levels.

Our study results indicate that different wetland functions achieve their highest values at different water levels, and the high functional integrity is enhanced by moderately water levels. As Ludwig *et al*. (2004) suggested that ecosystem with high functional integrity should maintain biodiversity, and vice versa. Our simulated results of biodiversity maintenance and raw material production are roughly consistent with the conclusion. So does the index of the habitat for rare waterfowls, which is of highest value at the water level of 5.2 m, where the functional integrity index achieves its highest value. As observed in this study, there is the largest amount of *Ciconia boycia* in the shallow water area with *Phragmites australis*, and less in deep water area as well as non-inundated area with other plants. It is the reason that shallow water area with *Phragmites australis* communities could provide an optimal habitat for *Ciconia boycia* to prey and move (Schaub *et al*., 2004; Liu *et al*., 2006; 2007). However, water-salt balance and the entertainment function are of sustained growth with the increase of water level, and achieve their greatest values at the highest water levels simulated. Raw material production which directly associates with the total area of the *Phragmites australis* reaches its highest value at the water level of 4.8 m. To sum up, the tendencies of individual wetland functions are not all consistent with those of water levels. As water levels can be designed for different purposes (Denis and Pauline, 2003), this study can provide guiding information for the wetland managers. In fact, if we accept that our desired goal is to maintain the functional integrity of wetlands rather than individual function, then a tradeoff strategy of estimating the optimal EWRs can be developed to ensure functional integrity of wetland ecosystems.

In the paper, to keep the method for simulation and calculation (model in short) relatively simple, some factors influencing EWRs have been excluded. Since it is not a temporal model, the method does not address the characteristics of hydrological regime such as duration of inundation, magnitude of seasonal water level fluctuations, frequency of inundation that play a significant role in maintaining wetland ecosystems. However, the model can be used to describe management procedures for examining functionally beneficial water levels using simulated water levels and GIS. EWRs estimated with this method reveal fundamental information on how wetlands perform their functional integrity to achieve a desired state, and provide valuable reference for wetland restoration and conversation. Furthermore, to achieve different management goals, management authorities can calculate and predict EWRs according to different ecological objectives, which are designed by different indices, such as the habitat suitability index and the biodiversity index. The model provides a framework for modeling other similar wetlands, where such data can be attained as topography map, DEM, measured water levels or water depths, vegetation conditions, and other information including weights of functions that is available in the historical literature.

5 Conclusions

The research on EWRs for wetlands is a new field requiring comprehensive use of multi-disciplinary techniques. Presently there is not a mature and widely acceptable theory and methodology. This paper is devoted to pioneer research work in this field. The established model focuses on the early development of EWRs for wetlands based on the simulation of habitat response to water level associated with functional integrity, thereby overcoming the dependence of some traditional methods on hydrological data. However, some factors, such as the time cumulative effect, seasonal water levels, sedimentation and water quality, were not taken into account in the simulating progress, so the result may be of some uncertainty. Future work may focus on connecting functional factors and examining more on the temporal and spatial variability in favor of sustainable management for wetland restoration.

References

Allan J, Lovett S, 1997. *Impediments to Managing Environmental Water Provisions*. Canberra: Bureau of Resource Sciences.

Bian Zhengfu, Zhang Guoliang, 2000. Application of bio-diversity indices to mined land reclamation. *Journal of China Coal Society*, 25(1): 76–80. (in Chinese)

Boar R R, 2006. Responses of a fringing Cyperus papyrus L. swamp to changes in water level. *Aquatic Botany*, 84(2): 85–92. DOI: 10.1016/j.aqubot.2005.07.008

Boyce R L, Elison P C, 2001. Choosing the best similarity index when performing fuzzy set ordination on binary data. *Journal of Vegetation Science*, 12: 711–720.

Casanova M T, Brock M A, 2000. How do depth, duration and frequency of flooding influence the establishment of wetland plant communities. *Plant Ecology*, 147(2): 237–250. DOI: 10.1023/A:1009875226637

Chow-Fraser P, 2005. Ecosystem response to changes in water level of Lake Ontario marshes: Lessons from the restoration of Cootes Paradise Marsh. *Hydrobiologia*, 539(1): 189–204. DOI: 10.1007/s10750-004-4868-1

Cui Baoshan, Yang Qichun, Yang Zhifeng *et al*., 2009. Evaluating the ecological performance of wetland restoration in the Yellow River Delta, China. *Ecological Engineering*, 35(7): 1090–1103. DOI: 10.1016/j.ecoleng.2009.03.022

Denis A H, Pauline H, 2003. A desktop model used to provide an initial estimate of the ecological instream flow requirements of rivers in South Africa. *Journal of Hydrology*, 270(3–4): 167–181. DOI: 10.1016/S0022-1694(02)00290-1

Desgranges J L, Ingram J, Drolet B *et al*., 2006. Modeling wetland bird response to water level changes in the Lake Ontario-St.Lawrence River hydrosystem. *Environmental Monitoring and Assessment*, 113(1–3): 329–365.

Dungan J L, Perry J N, Dale MR T *et al*., 2002. A balanced view of scale in spatial statistical analysis. *Ecography*, 25(5): 626–640. DOI: 10.1034/j.1600-0587.2002.250510.x

Duvail S, Hamerlynck O, 2003. Mitigation of negative ecological and socio-economic impacts of the Diama dam on the Senegal River Delta wetland, using a model based decision support system. *Hydrology and Earth System Sciences*, 7(1): 133–146.

Finlayson C M, 2005. Plant ecology of Australia′s tropical floodplain wetlands: A review. *Annals of Botany*, 96(4): 541–555. DOI: 10.1093/aob/mci209

Finlayson C M, Lowry J, Bellio M G *et al*., 2006. Biodiversity of the wetlands of the Kakadu Region, northern Australia. A*quatic Science*, 68(3): 374–399. DOI: 10.1007/s00027-006-085-2-3

Flinn M B, Adams S R, Whiles M R *et al*., 2008. Biological responses to contrasting hydrology in backwaters of Upper Mississippi River Navigation Pool 25. *Environmental Management*, 41(4): 468–486. DOI: 10.1007/s00267-008-9078-6

Fortney R H, Benedict M, Gottgens J F *et al*., 2004. Aquatic plant community composition and distribution along an inundation gradient at two ecologically-distinct sites in the Pantanal region of Brazil. *Wetlands Ecology and Management*, 12(6): 575–585. DOI: 10.1007/s11273-005-1763-0

Froend R, Loomes R, 2004. *Approach to Determination of Ecological Water Requirements of Groundwater Dependent Ecosystems in Western Australia—A Report to the Department of Environment*. Perth: Edith Cowan University.

Froend R, Loomes R, 2006. *Determination of Ecological Water Requirements for Groundwater Dependent Ecosystems—Southern Blackwood and Eastern Scott Coastal Plain*: *Report for the Department of Water*. Perth: Edith Cowan University.

Hayashi M, van der Kamp G, 2000. Simple equations to represent the volume-area-depth relations of shallow wetlands in small topographic depressions. *Journal of Hydrology*, 237(1–2): 74–85. DOI: 10.1016/S0022-1694(00)00300-0

Howard R J, Rafferty P S, 2006. Clonal variation in response to salinity and flooding stress in four marsh macrophytes of the northern gulf of Mexico, USA. *Environmental and Experimental Botany*, 56: 301–313.

Hughes D A, 2001. Providing hydrological information and data analysis tools for the determination of ecological instream flow requirements for South African rivers. *Journal of Hydrology*, 241(1–2): 140–151. DOI: 10.1016/S0022-1694(00)00378-4

Hughes D A, 2005. Hydrological issues associated with the determination of environmental water requirements of ephemeral rivers. *River Research and Applications*, 21(8): 899–908. DOI: 10.1002/rra.857

Jackson M B, Colmer T D, 2005. Response and adaptation by plants to flooding stress. *Annals of Botany*, 96(4): 501–505. DOI: 10.1093/aob/mci205

Kennedy M P, Milne J M, Murphy K J, 2003. Experimental growth responses to groundwater level variation and competition in five British wetland plant species. *Wetland Ecology and Management*, 11(6): 383–396. DOI: 10.1023/B:WETL. 00000-07194.01073.6b

King J, Louw D, 1998. Instream flow assessments for regulated rivers in South Africa using the building block methodology. *Aquatic Ecosystem Health and Management*, 1(2): 109–124. DOI: 10.1080/14634989808656909

Krstolic J L, Hayes D C, Ruhl P M, 2006. Physical habitat classification and instream flow modeling to determine habitat availability during low-flow periods, North Fork Shenandoah River, Virginia. *Scientific Investigations Report*: 5025.

Laitinen J, Rehell S, Oksanen J, 2008. Community and species responses to water level fluctuations with reference to soil layers in different habitats of mid-boreal mire complexes. *Plant Ecology*, 194(1): 17–36. DOI: 10.1007/s11258-007-9271-3

Legendre P, Dale M R T, Fortin M J *et al*., 2002. The consequences of spatial structure for the design and analysis of ecological field surveys. *Ecography*, 25: 601–615.

Lin Zhenshan, Liu Huihong, Liu Hongyu, 2005. Non-autonomous population dynamics model for single species with Allee effect affected by human activities and its application: A case study of red-crowned cranes Grus japonensis. *Acta Ecologica Sinica*, 25(5): 945–951. (in Chinese).

Liu Hongyu, Li Zhaofu, Bai Yunfang, 2006. Landscape simulating of habitat quality change for oriental white stork in Naoli River Watershed. *Acta Ecologica Sinica*, 26(12): 4007–4013. (in Chinese).

Liu Hongyu, Li Zhaofu, LI Xiaomin, 2007. Effects of wetland landscape fragmentation on habitats of oriental white storks: A case study on northeastern Sanjiang Plain, China. *Journal of Natural Resources*, 22(5): 817–823. (in Chinese).

Ludwig J A, Tongway D J, Bastin G N J *et al*., 2004. Monitoring ecological indicators of rangeland functional integrity and their relation to biodiversity at local to regional scales. *Austral ecology*, 29(1): 108–120. DOI: 10.1111/j.1442-9993.2004.013-49.x

Middleton B A, 2002. *Flood Pulsing in Wetlands*: *Restoring the Natural Hydrological Balance*. New York: John & Wiley Sons.

Nicol J M, Ganf G G, Pelton G A, 2003. Seed banks of a southern Australian wetland: the influence of water regime on the final floristic composition. *Plant Ecology*, 168(2): 191–205.

Pallisson J, Reeber S, Carpentier A *et al*., 2006. Plant-water regime management in a wetland: consequences for a floating vegetation-nesting bird, whiskered tern Chlidonias hybridus. *Biodiversity and Conservation*, 15: 3469–3480. DOI: 10.1007/978-1-4020-5734-2

Pichancourt J B, Burel F, Auger P, 2006. A hierarchical matrix model to assess the impact of habitat fragmentation on population dynamics: an elasticity analysis. *Comptes Rendus Biologie*, 29(1): 31–39. DOI: 10.1016/j.crvi.2005.09.009

Powell S J, Letcher R A, Croke B F W, 2008. Modelling floodplain inundation for environmental flows: Gwydir wetlands, Australia. *Ecological Modeling*, 211(3–4): 350–362. DOI: 10.1016/j.ecolmodel.2007.09.013

Reid M A, Brooks J J, 2000. Detecting effects of environmental water allocations in wetlands of the Murray-Darling Basin, Austrilia. *Regulated Rivers*: *Research and Management*, 16(5): 479–496. DOI: 10.1002/1099-1646(200009/10)16:5<479:: AID-RRR599>3.0.CO;2-Y

Riis T, Hawes I, 2002. Relationships between water level fluctuations and vegetation diversity in shallow water of New Zealand lakes. *Aquatic Botany*, 74(2): 133–148. DOI: 10.1016/S0304-3770(02)00074-8

Robertson H A, James K R, 2002. Determining the water requirement for the rehabilitation of wetland habitat at Kanyapella Basin, Victoria. *Ecological Management and Restoration*, 3: 220–221.

Schaub M, Pradel R, Lebreton J D, 2004. Is the reintroduced white stork (*Ciconia ciconia*) population in Switzerland self-sustainable? *Biological Conservation*, 119(1): 105–114. DOI: 10.1016/j.biocon.2003.11.002

Sim L L, Davis J A, Chambers J M, 2006. Ecological regime shifts in salinised wetland systems. II. Factors affecting the dominance of benthic microbial communities. *Hydrobiologia*, 573(1): 109–131. DOI: 10.1007/s10750-006-0268-z

Smakhtin V, Revenga C, Doll P, 2004. Taking into account environmental water requirements in global-scale water resources assessments. *Comprehensive Assessment Report 2*. Colombo, Sri Lanka: Comprehensive Assessment Secretariat.

Smakhtin V U, 2001. Low flow hydrology: A review. *Journal of Hydrology*, 240(3–4): 147–186. DOI: 10.1016/S0022-1694(00)00340-1

Smakhtin V U, Eriyagam N, 2008. Developing a software package for global desktop assessment of environmental flows. *Environmental Modelling & Software*, 23(12): 1396–1406. DOI: 10.1016/j.envsoft.2008.04.002

Smith R G B, Brock M A, 2007. The ups and downs of life on the edge: the influence of water level fluctuations on biomass allocation in two contrasting aquatic plants. *Plant Ecology*, 188(1): 103–116. DOI: 10.1007/s11258-006-9151-2

Sun Tao, Yang Zhifeng, Shen Zhenyao *et al*., 2009. Environmental flows for the Yangtze Estuary based on salinity objectives. *Communications in Nonlinear Science and Numerical Simulation*, 14(3): 959–971. DOI: 10.1016/j.cnsns.2007.10.006

Tharme R E, King J M, 1998. Development of the building block methodology for instream flow assessments and supporting research on the effects of different magnitude flows on riverine ecosystems. *Water Research Commission Report No 576/1/98*. Pretoria, South Africa.

Ulrich K, Rainer B, Konrad H, 2004. Assessment of river habitat in Brandenburg, Germany. *Limnologica*, 34(3): 176–186. DOI: 10.1016/S0075-9511(04)80043-4

White S D, Deegan B M, Ganf G G, 2007. The influence of water level fluctuations on the growth of four emergent macrophyte species. *Aquatic Botany*, 86(4): 309–315. DOI: 10.1016/j.aquabot.2007.01.006

Whittaker R H, 1960. Vegetation of the Siskiyou Mountains, Oregon and Califonia. *Ecological Monographs*, 30(3): 279–338. DOI:10.2307/1943563

Wilcox D A, Sweat M J, Carlson M L *et al*., 2006. A water-budget approach to restoring a sedge fen affected by diking and ditching. *Journal of Hydrology*, 320(3–4): 501–517. DOI: 10.1016/j.jhydrol.2005.07.026

Wolanski E, Boorman L A, Chicharo L *et al*., 2004. Ecohydrology as a new tool for sustainable management of estuaries and coastal waters. *Wetlands Ecology and Management*, 12(4): 235–276. DOI: 10.1007/s11273-005-4752-4

Wolfgang O, 2004. New approaches to integrated quality assessment of lakeshores. *Limnologica*, 34(1–2): 160–166. DOI: 10.1016/S0075-9511(04)80036-7

Yang Zhifeng, Cui Baoshan, Liu Jingling, 2005. Estimation methods of eco-environmental water requirements: Case study. *Science in China* (*Series D*), 48(8): 1280–1292. DOI: 10.1360/02yd0495

Zhang Xiangwei, Takeuchi K, 2004. Methodology for modeling of groundwater flow in large area. *Journal of Hydraulic Engineering*, 35(6): 7–13. (in Chinese)