Chapter 9
Wetland Ecosystems of the Yellow River Source Zone

Zhiwei Li, Zhaoyin Wang and Baozhu Pan

Abstract Diverse wetlands such as alpine meadows, lakes and peatlands are extremely important resources for water supply and ecological protection of aquatic ecosystems in the source zone of the Yellow River. Field surveys (2010–2014) and interpretations of remote sensing images are used to provide insights into the distribution of wetlands in this region and associated notions of landscape connectivity. Ruoergai Swamp (Zoige) at the eastern margin of the Qinghai–Tibet Plateau in the Yellow River Source Zone is the world's largest plateau peat wetland. A case study of this swamp shows that it has shrunk greatly since the 1950s. Environmental suffering (i.e. desertification, grassland degradation and run-off reduction) is leading to the ecological degradation of the Ruoergai peatlands, severely affecting local herdsmen and the surrounding communities (e.g. Sichuan Basin and the Upper Yellow River). Wetland degradation is affected by both global climate warming and human activities (especially construction of artificial ditches). The second half of this chapter uses field and laboratory analyses of benthic macroinvertebrates as indicators to assess the status of aquatic ecosystems

Z. Li (✉)
School of Hydraulic Engineering, Changsha University of Science & Technology, Changsha, Hunan 410114, China
e-mail: lizw@mail.tsinghua.edu.cn; lzhiwei2009@163.com

Z. Wang
State Key Laboratory of Hydroscience and Engineering, Tsinghua University, Beijing 100084, China
e-mail: zywang@mail.tsinghua.edu.cn

B. Pan
Changjiang River Scientific Research Institute, Wuhan 430010, China
e-mail: zhuzipan@163.com

in the region, contrasting findings from the Upper Yellow and Yangtze rivers. Human activities such as artificial ditch system, overstocking and peat mining have impacted upon local aquatic ecosystems. Such pressures have been accentuated by the economic and social development since the 1980s.

Keywords Wetland · Peatland   degradation · Aquatic   ecosystem · Macroinvertebrates · Human activities

9.1 Introduction

The shrinkage of wetlands on the Qinghai–Tibet Plateau since the 1950s endangers both local terrestrial and aquatic ecosystems as well as the supply of water from the Yellow River Source Zone to the 110 million people who live in downstream catchments. The degradation and loss of wetlands has intensified across the world in recent decades. Recently, Niu et al. (2012) reported that the area of wetlands in China in 2008 was around 324,097 km^2, a reduction of about 33 % since 1978. The rapid shrinkage of wetlands in China is attributed to fill, drainage and agricultural development associated with population growth and rapid economic development and to climate warming (Qiu et al. 2009; Zhou et al. 2009).

Wetlands provide many ecological functions and ecosystem services, including the provision of habitat for diverse wildlife, acting as carbon sinks that mitigate effects of climate warming and, filtering sediments in ways that buffer nutrient loads and improve water quality and reducing the frequency and extent of flooding (Barbier et al. 1997; Brinson and Malvarez 2002). Enhanced protection and restoration of wetlands is therefore an important component of programmes to support the maintenance of aquatic and terrestrial ecosystems, mitigate floods and improve water quality (Cao and Fox 2009; Niu et al. 2011).

We start this chapter by considering the different types of wetlands found on the Qinghai–Tibet Plateau, describing the processes that have generated them and ways they have changed in recent years. Next, we focus on the Zoige (Ruoergai) peatlands, a vast wetland of international importance with a rich cultural history. The various processes contributing to the shrinking of wetland area and the degradation of the remaining peatlands are discussed. Several methods to assess erosion rates in this region are outlined. The latter part of this chapter is concerned with the biota found within the aquatic ecosystems of the areas through systematic analysis of macroinvertebrate assemblages in the source regions of the Yellow and Yangtze rivers.
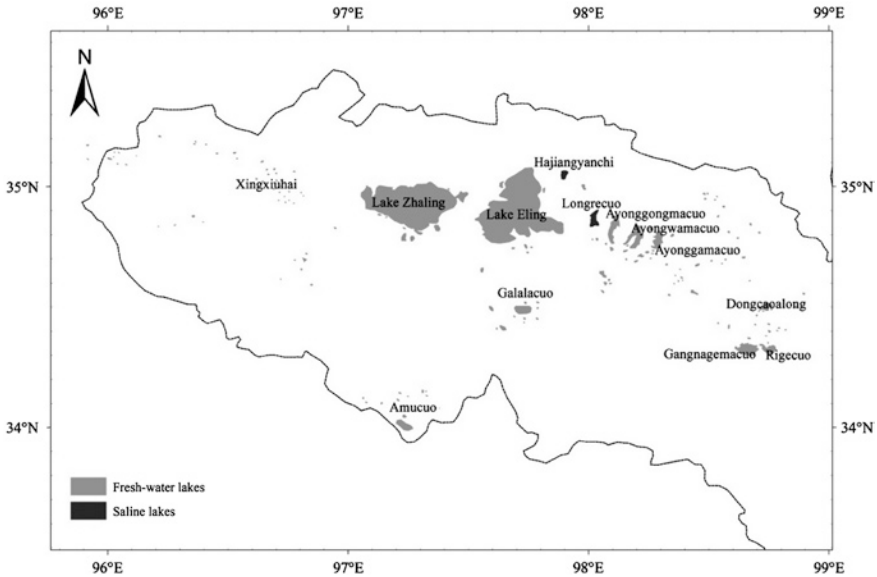
Fig. 9.1 Lakes in the Yellow River Source Zone

9.2 An Overview of Wetlands in the Yellow River Source Zone

Wetlands in the Yellow River Source Zone are concentrated mainly in the upper segments of the river (Fig. 9.1; Brierley et al. 2016, Chap. 3; Huang et al. 2016, Chap. 4). Several different wetland types are found in this region, including alpine meadow wetlands, lakes and peatlands (Blue et al. 2013; Gao et al. 2013; Nicoll et al. 2013). Each of these types of wetland is generated and sustained by different processes and is therefore vulnerable to different risks and likely to respond in different ways to the impacts of climate change and human activities.

9.2.1 Alpine Meadow Wetlands

Alpine meadow wetlands are found in the lowland areas on the gentle footslopes of alpine watersheds (see Gao 2016, Chap. 10). Snowmelt and rainfall produce groundwater seepage at the base of hillslopes, thereby forming a shallow water zone, where flow velocity is very slow due to the narrow outlet and blocked drainage. For example, the Xingsuhai region in the Yellow River Source Zone area has an arid-cold climate with little precipitation, but evaporation is relatively low due to the low-temperature regime. Based on the above conditions, many alpine
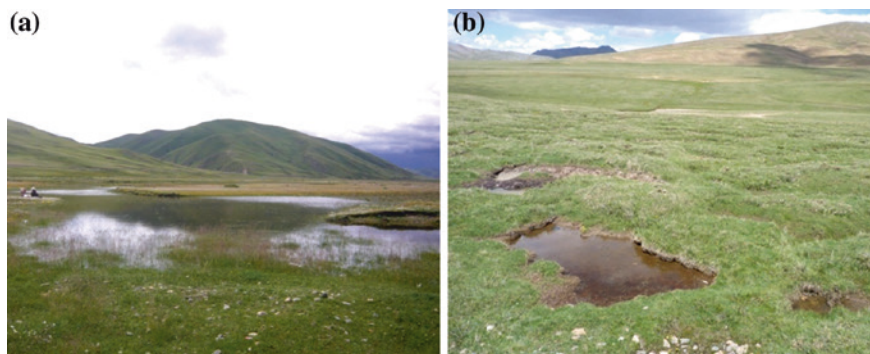
Fig. 9.2 Alpine meadow wetlands in the Yellow River Source Zone. **a** A typical alpine meadow wetland in 2007. **b** A relatively dewatered meadow in 2013

meadows form in the zone where run-off accumulates on low-lying land and in areas with high groundwater levels (Tane et al. 2016, Chap. 13). As the primary vegetation type, perennial herbaceous phreatophytes thrive in such conditions. Typical alpine meadow plants include *Kobresia schoenoides, Kobresia pygmaea, Kobresia humilis, Gramineae, Poaceae, Cyperaceae, Medulla junci, Leguminosae* and broad-leaved herbs. The area covered by alpine meadow wetlands in the Yellow River Source Zone has markedly decreased over the past fifty years (Fig. 9.2). Many wetland areas have been transformed into grazing pasture or have been impacted by road construction.

9.2.2 Lake Wetlands

There are 31 lakes with an area >1 km^2 in the Yellow River Source Zone, totalling around 1456 km^2 in 2012–2013 (Fig. 9.1). Zhaling and Eling lakes are the largest, making up almost three-quarters of the total lake area in the region. Eling Lake has an area of 628 km^2, a mean depth of 17.6 m (maximum 30 m) and water storage capacity of 10.7 billion m^3, while Zhaling Lake has an area of 526 km^2, a maximum depth of 8.6 m and a storage capacity of 4.6 billion m^3. The Yellow River connects these two lakes, running through a 20 km long and 300 m wide river valley. The river exits in the south-west corner of Eling Lake, where the mean annual discharge is about 485 million m^3. There are many small lakes to the east of Eling Lake, including Longrecuo (19 km^2), Aronggongmacuo (30 km^2), Arongwamacuo (31 km^2) and Aronggamacuo (23 km^2) (Fig. 9.1). In addition, many small lakes are located within the Xingxinghai wetland, which is not connected to the Yellow River trunk stream.

Remote sensing images indicate that although the total lake area in the Yellow River Source Zone has not changed in the last 20 years, the area of some lakes has
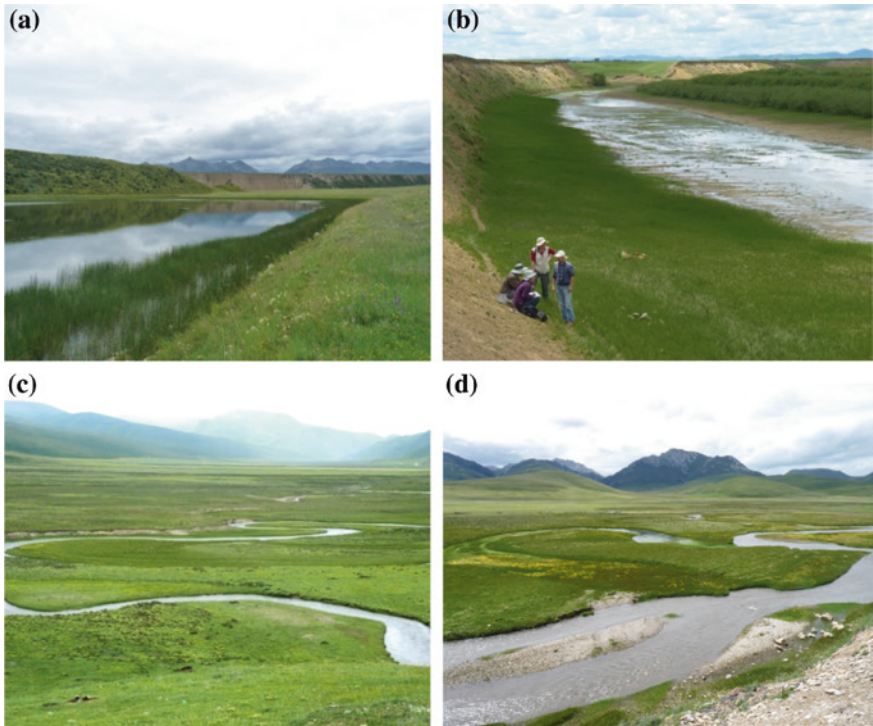
Fig. 9.3 Oxbow lakes in the Yellow River Source Zone. **a** An oxbow lake near Kesheng Town, Henan County. The lake is 1 km long and 100 m wide. It is separated from the contemporary channel by a natural levee. Its surface is covered by aquatic plants. **b** An oxbow lake near Tangke town on the first bend of the Yellow River trunk stream. The lake is partially connected to the trunk stream. It is 3 km long and 100 m wide. Rapid siltation is evident at this site. **c** Oxbow lakes along the Lanmucuo River, a tributary of the Yellow River. **d** An older oxbow lake that is infilling with aquatic vegetation along a tributary of the Upper Yellow River near Kesheng

increased (e.g. Zhaling, Eling and Xingxing lakes), while others have decreased in area. The expansion of some lakes has been attributed to increasing snow and glacier melts and/or dam construction.

9.2.3 Oxbow Lakes

The Yellow River trunk stream and its tributaries contain numerous oxbow lakes formed when meander loops are cut off from the main channel (Fig. 9.3). For instance, the Bai River has more than 90 oxbow lakes and the Hei River has more than 250 oxbow lakes.

Fig. 9.4 Early development stages of two crescent lakes in Maduo County

9.2.4 Crescent Lakes

Crescent lakes occur in the desert regions of the Yellow River Source Zone and are formed by groundwater rising above the crescent base of sand dunes (Fig. 9.4). Satellite imagery indicates that there are more than 8400 crescent lakes in the desert areas of Yellow River Source Zone (e.g. the Maduo desert between Eling Lake and the Yellow River). These lakes are typically 100–300 m long, 30–100 m wide and 1–10 m deep. Perhaps, the most famous example is Crescent Lake in the Dunhuang Desert, Gansu Province at the northern margins of the plateau. This lake has the shape of a crescent moon. It is nearly 100 m long, 25 m wide and has a maximum depth of about 5 m.

These distinctive hydrogeomorphic phenomena rely upon constant recharge by groundwater, so that the water table continues to lie above the base of the dunes. Climate warming since 1960 has increased the volume of meltwaters from snow and glaciers, leading to rising groundwater levels and an increasing number and area of crescent lakes. Rising groundwater levels also result from dam construction. For example, construction of the 20-m-high Huangheyan Dam downstream of Eling Lake in 2002 has increased the number of crescent lakes in this area from about 6800 to over 8400.

9.2.5 The Ruoergai Peatland

Peatland in the Yellow River Source Zone is found mainly in the Zoige (Ruoergai) Basin located at the eastern margin of the Qinghai–Tibet Plateau. Ruoergai peatland covers a total area of 16,000 km^2, of which about 80 % is in Sichuan Province and 20 % is within Qinghai and Gansu provinces.

Ruoergai is the largest high-altitude peatland in the world. Because of the long-term inundation and hypoxic or anoxic conditions, dead aquatic plants in the

peatland cannot be oxidised. These plants eventually form a peat layer composed of densely interwoven dead grass roots that is typically 0.1–3.0 m thick at the bottom of the peatland. Prior to the 1950s, the Ruoergai peatland covered more than 4600 km^2. Today, it covers just 2200 km^2. This 52.2 % decrease in area has seriously degraded the functionality of this wetland ecosystem (Yan and Wu 2005; Li et al. 2011). Historical records and documents indicate that prior to the 1930s, the peatland was almost-pristine, with negligible impacts from human activities such as herding in the adjacent Riganqiao peatland. At this time, access to the main peatland was restricted by the presence of perennial water more than 1 m deep. The Ruoergai peatland has an infamous place in history as more than 23,300 members of the Chinese Red Army died while trying to cross it during the rainy season in August 1935 and July 1936 (Baidu Encyclopaedia 2014). Today, much of the peatland has been converted to grasslands, and some local areas are even prone to desertification.

9.3 Degradation of the Ruoergai Peatland

Shrinkage and degradation of the Ruoergai peatland is evident from analysis of photographic images taken in the early 1960s, which reveal a marked reduction in surface water area. Declining groundwater levels were also experienced at this
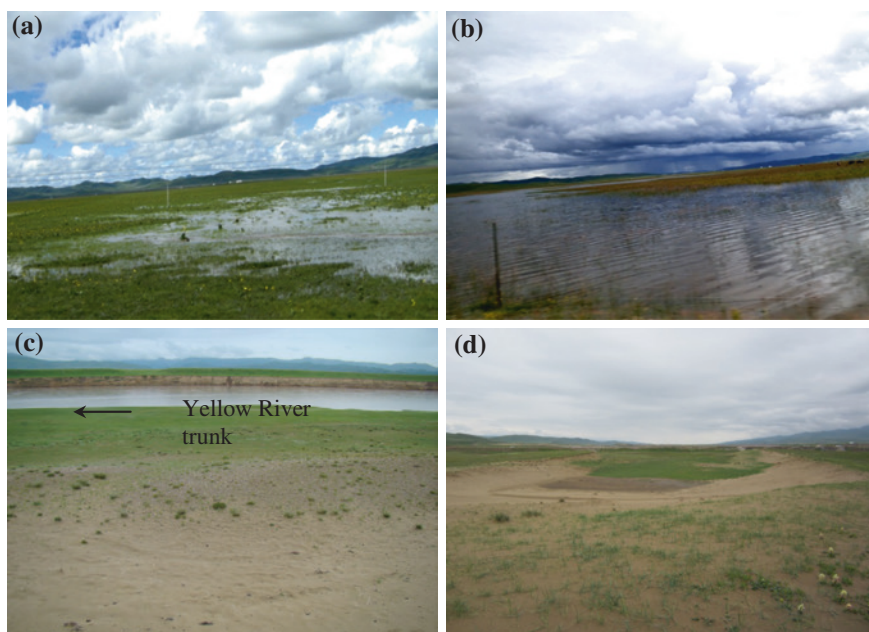

Fig. 9.5 Images of the Ruoergai peatland at various states. **a** Peatland with shallow water. **b** Peatland with deep water. **c** Desertification. **d** Vegetation on desertified ground

time. Shrinkage of the peatland rapidly accelerated in the 1980s. By 2000, apart from the large area of lakes and swamps along upper and middle sections of the Hei River, the area had been transformed into "wet grassland", with no distinct surface water evident across the peatland, even during the rainy season (Yang 1999; Shen et al. 2003). Today, serious desertification is evident in the wetlands bordering both banks of the Upper Yellow River and along the Hei and Bai river tributaries (Fig. 9.5).

Rapid shrinkage of the Ruoergai peatland has not only affected ecological functions, but also caused loss of biodiversity, altered water supply to the Upper Yellow River and threatened local livelihoods, especially livestock farming (Zhang and Lu 2010). Some researchers contend that shrinkage of the Ruoergai peatland in recent decades is a product of global climate warming, as rising temperatures have enhanced evaporative losses (e.g. Yan and Wu 2005). Ning et al. (2011) showed that evaporation at the Maqu weather station showed a linear decreasing trend of 44.0 mm per decade from 1969 to 2008. This effect was most prominent in summer, reflecting reductions in the diurnal temperature range and changes to wind speed. However, Li et al. (2015) argue that human practices have been the primary driver of peatland shrinkage in recent decades. The excavation of artificial drainage ditches through the peatland breaks up the peat materials that naturally inhibit gully development and headward erosion. Once this surface layer is cut through, the exposed underlying materials become highly vulnerable to erosion, triggering a sequence of events that can culminate in desertification (see Fig. 9.5).

9.3.1 Impact of Ditch Excavation upon Ruoergai Peatland

Hundreds of kilometres of ditches were excavated in Ruoergai, Hongyuan and Maqu counties in the 1960s–1990s to rapidly drain wetlands and so increase pasture available for animal husbandry. Official records indicate that 380 km of ditches were excavated in Ruoergai County from 1965 to 1973, directly draining some 2.1 km^2
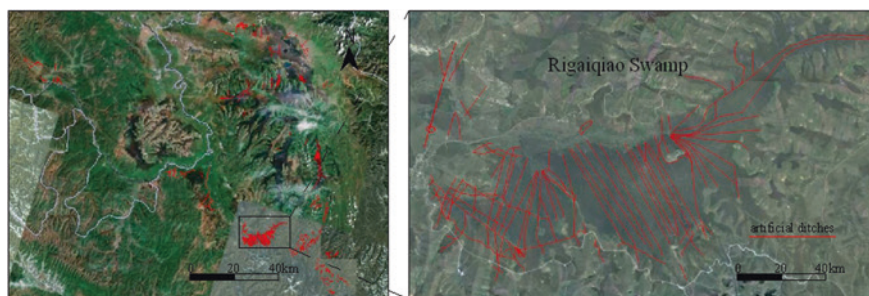

Fig. 9.6 Artificial ditches in Ruoergai and Riganqiao peatlands (Google Earth, 20 September 2010). The *centre line* of ditches is indicated by a *red solid line*, so that areas with many ditches appear as *solid red*

Fig. 9.7 Straight, artificial drainage ditches in Ruoergai peatland
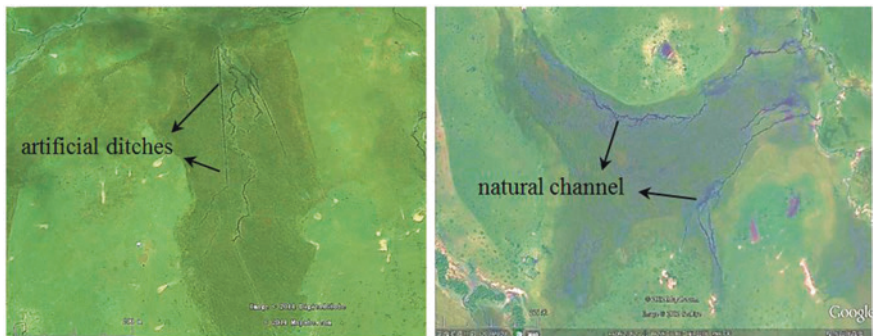

Fig. 9.8 Comparison of the drainage effects of artificial ditches and extended river networks

of perennial peatlands. Corresponding figures for the 1990s are 50.5 km of ditches converting a further 0.22 km^2 of peatland to grassland. Altogether, the length of artificial ditches in Ruoergai and Hongyuan counties now exceeds 1000 km, draining about 2000 km^2 (over 43 % of the peatlands). Figure 9.6 shows an example from the Riganqiao peatland in Hongyuan County where about 85 km^2 of peatland was converted to grazing land using artificial ditches. Today, interweaving artificial channels drain about 85 % of this area, so that surface water accumulates only in the central part of the peatland, and then only during the rainy season.

Based on the analyses of Google Earth imagery, on 20 September 2010, the total area of artificial ditches in Hongyuan, Ruoergai and Maqu counties stood at 1108.7, 884.7 and 834.1 km^2, respectively (Li et al. 2015). Two types of peatland drainage can be differentiated in this image:

(1)   Areas that are *completely* drained by artificial channels, such as the Riganqiao peatland (Fig. 9.7). These areas total 219.7 km^2, 4.8 % of the contemporary area of the Ruoergai peatland (4600 km^2);

(2)   Areas where artificial ditches *extend natural channel networks* to drain the wetlands, the Maiwa and Sedi wetlands in Hongyuan, the Heihe wetland
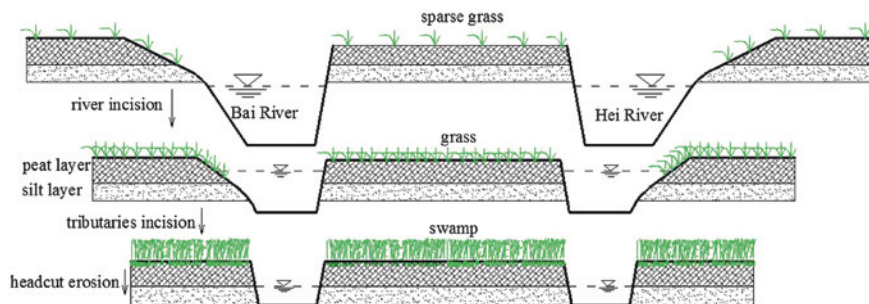
Fig. 9.9 Stages of river incision leading to the shrinkage of Ruoergai peatland

Fig. 9.10 A typical headcut in the Ruoergai peatland (33° 5′ 6.7″N, 102° 50′ 10.6″S)

in Ruoergai and the Awancang wetland in Maqu (Fig. 9.8). These areas total 428.6 km², some 9.3 % of the Ruoergai peatland.

Based on these figures, artificial ditches account for some 14.1 % of the total degraded area of the Ruoergai Swamp.

Drainage of peatland by artificial ditches and by channel extensions leads to wetland degradation and shrinkage, but different processes are involved in each case. Artificial ditches drain peatland along both banks of a channel, exposing the peat, which then dehydrates and hardens. The peatland is slowly transformed into a wet meadow, which then dries further to become grassland and can then eventually lose grass cover completely and become desertified. In contrast, the connection of artificial ditches to natural channels extends the drainage network, thereby strengthening the transport efficiency of the system. This leads to rapid headward incision of headcuts, undercutting of banks and subsequent channel expansion, which lowers the groundwater level and accelerates dehydration of wetlands (Safran et al. 2005; Schumm et al. 1984; Tooth et al. 2002; Fig. 9.9).

Extended channel networks in the Bai and Hei tributary systems have progressively drained the Ruoergai peatland and enlarged the desiccated area. Although the peat layer is resistant to erosion, once this layer is breached the underlying fine sand and silt layers are rapidly eroded. This promotes undercutting, bank collapse

Fig. 9.11 Examples of the Google Earth images used to estimate headcut erosion rates (20 September 2000 on the *left*; 15 June 2014 on the *right*)

and accentuated rates of headward erosion. A typical 10 m high drop at a headcut site is shown in Fig. 9.10.

The rate of headcut erosion is critical to the pace of wetland shrinkage. Although widespread incision is evident along Hei River tributaries (Gequ, Haqu and Jiqu), lack of historical incision depth data precludes accurate determination of the incision rate. However, estimates can be derived from measuring channel extension rates relative to the position of bridges and culverts along Provincial Highway S301 which runs through the Ruoergai peatland (constructed on 15 October 2010). Incision rates for 13 bridges and culverts over the 3 years indicate an average channel incision rate of 0.24 m year^{-1} (Li et al. 2015). These incision rates are considered to represent the upper limit of the actual rates, as there is no peat layer to protect the river bed at these locations, and flow scour is likely to be accentuated adjacent to bridges and culverts.

Other estimates of the rate of headcut erosion were made by comparing Google Earth images from 20 September 2010 and 15 June 2013 (an example is shown in Fig. 9.11). Based on measurements at 30 sites shown in these images, the average rate of headcut extension was 1.52 m year^{-1} (range from 0.68 to 17.12 m year^{-1}). Although rates can be easily measured across many sites using this method, accuracy is limited by the resolution and horizontal accuracy of the image (the images used in this analysis have a resolution of 0.65 m).

Rates of headcut erosion were also measured in the field between 2012 and 2013, where average rates of 0.42 m year^{-1} were observed. Each of the methods used to measure erosion rates is subject to errors of different kinds. Although field measurements appear the most credible, comparatively few sites can be measured, relative to analyses of remotely sensed images. This means that there is less opportunity to relate the observed rates to upstream catchment area, which is likely to be an important control on the individual rates measured at each site.

Fig. 9.12 Incised stream banks along the Upper Yellow River at Maqu Station, middle Hei River, Gequ and Haqu (an artificial ditch)

9.3.2 Long-Term Trends of River Incision and Headcut Erosion

Although human activities have undoubtedly accelerated the shrinkage and degradation of the Ruoergai peatland, an underlying long-term shrinking trend has been driven by river bed incision of the Yellow River, Bai River, Hei River and their tributaries in response to base level changes (Brierley et al. 2016, Chap. 3; Li et al. 2015). Groundwater levels have decreased and the drainage network has evidently expanded along both sides of river channels. For instance, terraces along the Upper Yellow River in the Maqu section are more than 10 m high, much higher than the 4 m height of corresponding features along the middle Hei River. As a comparison, the height of terraces along the tributaries of the upper Hei River, Gequ, Maqu and Haqu rivers are only around 0.5–3.0 m (see Fig. 9.12). Incision and channel expansion are also evident along the Dashui River, a tributary of Hei River. Upstream areas are yet to cut through the peat layer, with incised channels only about 0.5–1.5 m deep. Middle reaches have a depth of around 2 m, with channel widths of 10–30 m. The downstream section of channel is much deeper (about 5 m) and wider (over 500 m).

9.4 Protection and Restoration of Ruoergai Peatland

Degradation of wetlands in the Yellow River Source Zone can be attributed to both climate change and human activities. Unless urgent actions are taken, further intensification of human activities will only exacerbate degradation of these wetlands. The best strategy to protect the Ruoergai wetland is to convert it into protected natural areas or reserves. Hence, the Chinese government approved the establishment of a nature reserve for the Ruoergai peatland in Sichuan Province in 1998, which was listed as an "International important wetland" in 2008. This reserve has a total area of 1665.7 km^2. It measures 47 km from east to west and 63 km from north to south. Within the natural reserve, peatlands, meadows and marshes provide important habitat and breeding grounds for wildlife. Even though the Ruoergai Natural Reserve is now established, artificial drainage and peat mining, groundwater exploitation, overgrazing and tourism development continue, so the degradation of these wetlands will not stop unless additional measures are taken.

Drainage via artificial ditches and incision of rivers and ditches within the wetlands have been identified as the major forces behind the recent shrinkage of both peatland and meadow wetlands (Li et al. 2015). Hence, basic strategies to mitigate against future deterioration and to promote improvements in their condition include stopping artificial drainage, controlling incision through base control measures, prohibiting peat mining and reducing grazing pressure by limiting the expansion of grazing pastures. Roads should be elevated above the ground to protect the peatland rather than draining it. Concrete weirs can be placed at the main drainage outlets to control and reduce discharge of water from the peatland. Wherever appropriate, low-level dams can be built to increase the number of lakes, raise groundwater levels and restore wetlands. For instance, the Flower Lake in the Ruoergai wetland originally had a surface area of 156 ha that shrank by two-thirds in 2011. Some of the areas around the lake even became desertified. After the construction of a low dam downstream of the Flower Lake outlet, the watered area quickly expanded to 2000 ha. Today, Flower Lake is like a bright mosaic diamond in north-west Sichuan, where it has become a major tourist attraction. Construction of a high dam on the Yellow River downstream of Maqu County may also support efforts to restore the Ruoergai peatland.

In addition to protecting lakes and wetlands, it is also important to consider the aquatic ecosystems they contain, and it is to this challenge that we now turn our attention.

9.5 Aquatic Ecosystems of the Sanjiangyuan

9.5.1 Background

The source zones of the Yellow River and the Yangtze River are important water conservation areas in China. Not only do they supply water to millions of people, but they also play key roles in maintaining the unique biological communities of the region. However, given the harsh natural environment and the very fragile ecosystems in this area, it is critical to monitor their ecological condition, so that precautionary approaches to environmental management can be adopted.

In recent years, a significant body of research has documented changing environmental conditions in the source region of the Yellow River and Yangtze River basins, including climate (e.g. Li et al. 2006; Zhang et al. 2011), hydrology (e.g. Shi et al. 2007) and land use change (especially grassland productivity and degradation; Feng et al. 2009; Guo et al. 2008; Sheng et al. 2007; Yang et al. 2011). In the past 40 years, wetland areas in the source regions of the Yellow River and the Yangtze River have decreased by 13.6 and 28.9 %, respectively (Wang et al. 2009). This reduction reflects the combined effects of human activities and reduced rainfall resulting from climate change. Although spatio-temporal changes of habitat conditions for terrestrial organisms have been well documented (Liu et al. 2009), no systematic studies on aquatic biotic assemblages have been completed. Here, we present the first steps towards building a better understanding of aquatic ecosystems of the Yellow River Source Zone.

Benthic macroinvertebrates are important components of river ecosystems, cycling nutrients and providing food for higher trophic level creatures such as fish and birds. Because of their limited range and long life cycles, they are good indicators of human impacts and long-term environmental changes (Pan et al. 2012).

9.5.2 Macroinvertebrate Communities in the Sanjiangyuan

9.5.2.1 Method

Field investigations of benthic macroinvertebrates were carried out in August 2009 and July 2010 in the source regions of the Yellow and the Yangtze rivers (Fig. 9.13).

Water depth (Z) and Secchi depth (Z_{SD}) were measured in the field using a sounding lead and a Secchi disc, respectively. Flow velocity (U) was measured using a propeller-type current metre (Model LS 1206B). Analysis of suspended sediments (SS) followed the procedures outlined in the Standard Methods for Water and Wastewater Monitoring and Analysis (APHA) (2002). Conductivity was measured using a conductivity metre (Model DDS-11A). Total nitrogen (TN) was analysed using the alkaline potassium persulfate digestion-UV spectrophotometric method. Total phosphorus (TP) was analysed using the ammonium molybdate
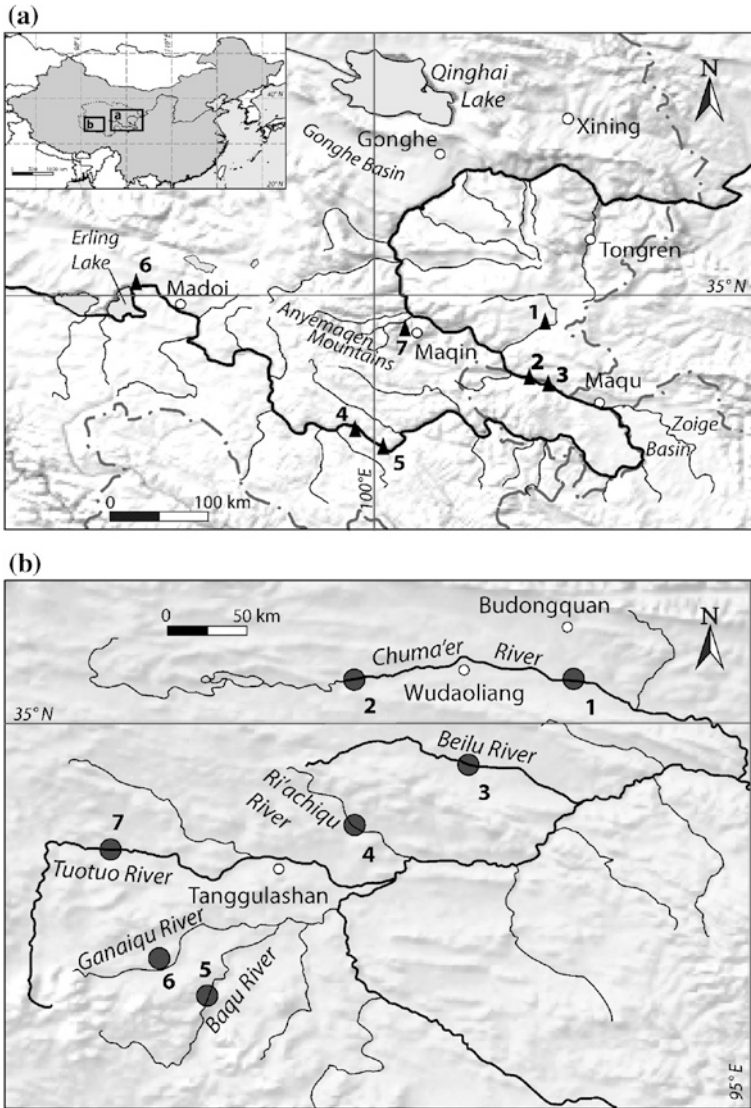
Fig. 9.13 Study areas and sampling sites in the source regions of **a** the Yellow and **b** the Yangtze rivers. Sampling sites in the source region of the Yellow River: *1* Zequ; *2* gravel bar along meandering Yellow River; *3* oxbow lake adjacent to the Yellow River; *4* riparian wetland of the Yellow River; *5* main river channel with low-sediment concentration; *6* shore of Erling Lake; *7* Jiangrang hydroelectric power station in a tributary of the Qiemuqu River. Sampling sites in the source region of the Yangtze River: *1* downstream Chuma'er River; *2* upstream Chuma'er River; *3* Beilu River; *4* Ri'achiqu River; *5* Buqu River; *6* Gananoque River; *7* Tuotuo River

Table 9.1 Environmental parameters of sampling sites in the source regions of the Yellow River and the Yangtze River (mean $\pm$ SE)

Environmental parameters	The source region of the Yellow River	The source region of the Yangtze River
Water depth (m)	0.4 ± 0.2	0.4 ± 0.2
Secchi depth (m)	0.30 ± 0.1	0.15 ± 0.10
Water velocity (m s^{-1})	0.26 ± 0.15	0.38 ± 0.05
Suspended sediments (mg L^{-1})	59.3 ± 39.3	351.6 ± 84.5
Conductivity (μS cm^{-1})	487 ± 12	2257 ± 838
Total nitrogen (mg m^{-3})	2940 ± 540	1967 ± 347
Total phosphorus (mg m^{-3})	8 ± 1	37 ± 17

method. All parameters were analysed according to APHA (2002). Near-surface and near-bed water samples were combined for laboratory analyses (Table 9.1).

At each sampling site, three replicate samples of macroinvertebrates were collected in a kick-net with 420 μm mesh size within an area of 0.33 m^2. Specimens were manually sorted from sediment on a white porcelain plate and preserved in 75 % ethanol. After blotting, the wet weight of the sampled animals was determined using an electronic balance. The dry weight was later calculated according to the ratio between wet weight and tissue-shell weight (see Yan and Liang 1999). All taxa were assigned to one of five functional feeding groups: shredders, collector-gatherers, collector-filterers, scrapers and predators (Morse et al. 1994). When a taxon could potentially be assigned to multiple feeding groups, its abundance was divided equally between these groups (e.g. if a taxon is both a collector-gatherer and a scraper, its abundance was divided equally between these groups). The Jaccard similarity coefficient (S_J) was then used to compare macroinvertebrate assemblages between the river source regions:

$$S_J = c/(a + b - c) \tag{9.1}$$

where a is the number of species in assemblage A; b is the number of species in assemblage B; and c is the number of species found in both assemblages.

Species richness (S, the number of species) is one of the most important characteristics of biodiversity, as it provides a measure of the diversity of both species and habitat. Another measure of diversity, the Shannon–Weaver index, H, integrates species richness with the evenness of the distribution of animals of different species. It is defined by Krebs (1978) as:

$$H' = -\sum_{i=1}^{S} \frac{n_i}{N} \ln\left(\frac{n_i}{N}\right) \tag{9.2}$$

in which S is the number of species (richness), N is the total number of individual animals, and n_i is the number of individual animal of the ith species. However, the Shannon–Weaver index can sometimes provide misleading results, since it provides no information on the total abundance of the biocommunity, which can
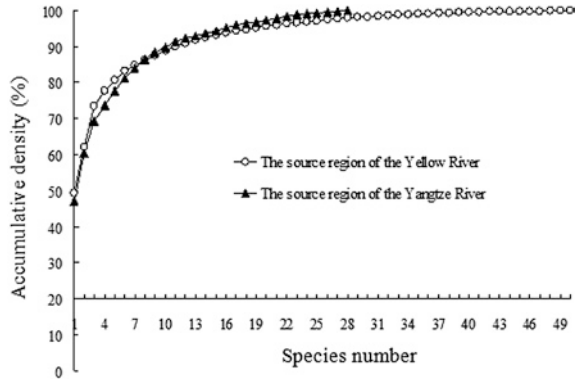
Table 9.2 Species composition of macroinvertebrates in the source regions of the Yellow River and the Yangtze River

Phylum	Class	Family	Number of species	
			Yellow River Source Zone	Yangtze River source zone
Nematoda			ud	0
Annelida	Oligochaeta	Naididae	1	0
		Tubificidae	4	4
Mollusca	Gastropoda	Lymnaeidae	3	0
		Planorbidae	2	0
Arthropoda	Crustacea	Gammaridae	ud	ud
	Arachnoida		ud	0
	Insecta	Caenidae	(1)	0
		Baetidae	(1)	(1)
		Heptageniidae	(1)	(1)
		Ephemerellidae	(1)	0
		Leptophlebiidae	0	(1)
		Hydropsychidae	(1)	0
		Leptoceridae	(1)	0
		Brachycentridae	(1)	0
		Nemouridae	0	ud
		Taeniopterygidae	0	ud
		Dytiscidae	ud	0
		Elmidae	ud	ud
		Chrysomelidae	ud	0
		Naucoridae	ud	0
		Corixidae	ud	0
		Pyralidae	ud	0
		Sialidae	0	ud
		Tipulidae	(1)	(1)
		Simuliidae	(1)	0
		Culicidae	ud	0
		Ephydridae	ud	0
		Chironomidae	(20)	(16)
Total number of species			50	29

vary substantially in habitats with different physical conditions. This drawback is resolved in the biocommunity index, B, which considers both total community abundance and species biodiversity (Wang et al. 2008):

$$B = -\ln N \sum_{i=1}^{S} \frac{n_i}{N} \ln\left(\frac{n_i}{N}\right)$$

(9.3)

Fig. 9.14 K-dominant curve of macroinvertebrates in the source regions of the Yellow and Yangtze rivers

9.5.2.2 Results

(a)   *Composition and diversity of macroinvertebrate assemblages*

Altogether, 68 species of macroinvertebrates belonging to 29 families and 59 genera were identified in the two source regions (Table 9.2). Among them were 8 annelids, 5 mollusks, 54 arthropods and 1 other animal. In the Yellow River Source Zone, we found 50 species belonging to 25 families and 46 genera, while in the source region of the Yangtze River, we found 29 species belonging to 11 families and 24 genera. Only 11 species of macroinvertebrates coexisted in the two river source regions. The Jaccard coefficient between the source regions of the Yellow River and the Yangtze River was only 0.16, indicating little similarity between the macroinvertebrate communities of these two source regions.

The Shannon–Weaver index (*H*) of macroinvertebrates in the source regions of the Yellow River and the Yangtze River was very similar (2.06 and 2.05, respectively). However, the biocommunity index (*B*) of macroinvertebrates in the two regions was quite different [11.86 (Yellow River) and 8.41 (Yangtze)].

Species diversity can also be assessed using K-dominant curves which combine measures of diversity, species richness and distribution evenness. Using this method, dominant patterns are represented by plotting the accumulative abundance of each species (%) ranked in decreasing order of abundance (see, Fig. 9.14).

Biodiversity comparisons indicated that macroinvertebrate diversity in the source region of the Yellow River was higher than that in the source region of the Yangtze River. Although the Shannon–Weaver index suggests similar levels of macroinvertebrate diversity in the two areas, the number of macroinvertebrate species found in the source area of the Yellow River (50 species) was higher than that found in the Yangtze River (29 species), and both the biocommunity index and the K-dominant curve analysis indicate higher levels of diversity in the Yellow River Source Zone.
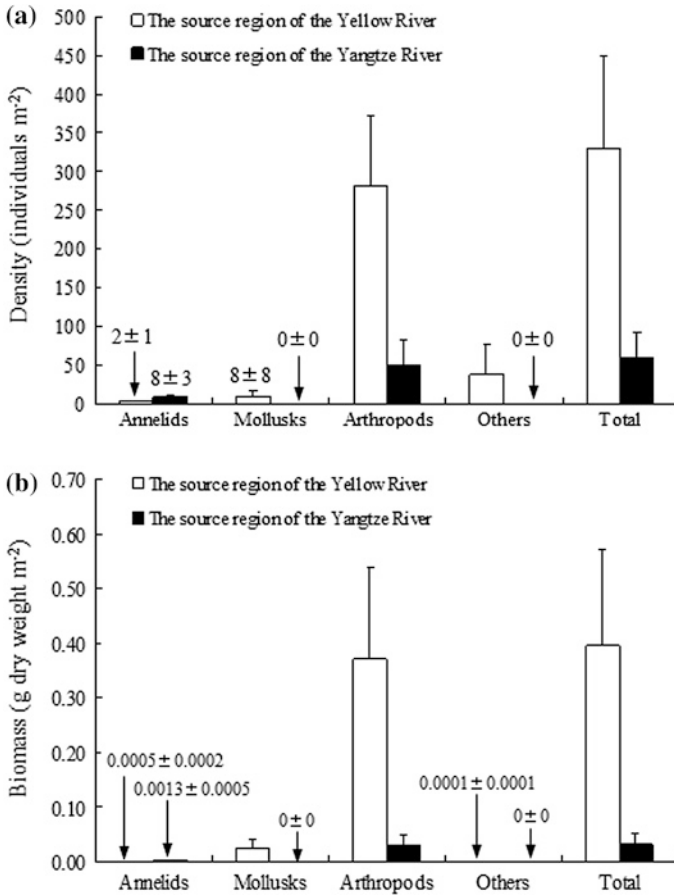
Fig. 9.15 Density (**a**) and biomass (**b**) of each phylum of macroinvertebrates in the source regions of the Yellow River and the Yangtze River (Mean ± SE)

(b) *Density and biomass of macroinvertebrate assemblages*

Overall, macroinvertebrates have a density of 329 ± 119 and 59 ± 32 individuals m^{-2} in the source regions of the Yellow River and the Yangtze River, respectively (Fig. 9.15). The biomass of total macroinvertebrates was 0.3966 ± 0.1763 and 0.0307 ± 0.0217 g dry weight m^{-2} in the source regions of the Yellow River and the Yangtze River, respectively. Arthropods were the dominant group in each source region, both in terms of the number of species present and in terms of density and biomass. In the source region of the Yellow River, arthropods made up 85.4 % of the total density and 93.5 % of the total biomass. In the source region of the Yangtze River, arthropods comprised 86.2 % of the total density and 95.8 % of the total biomass.
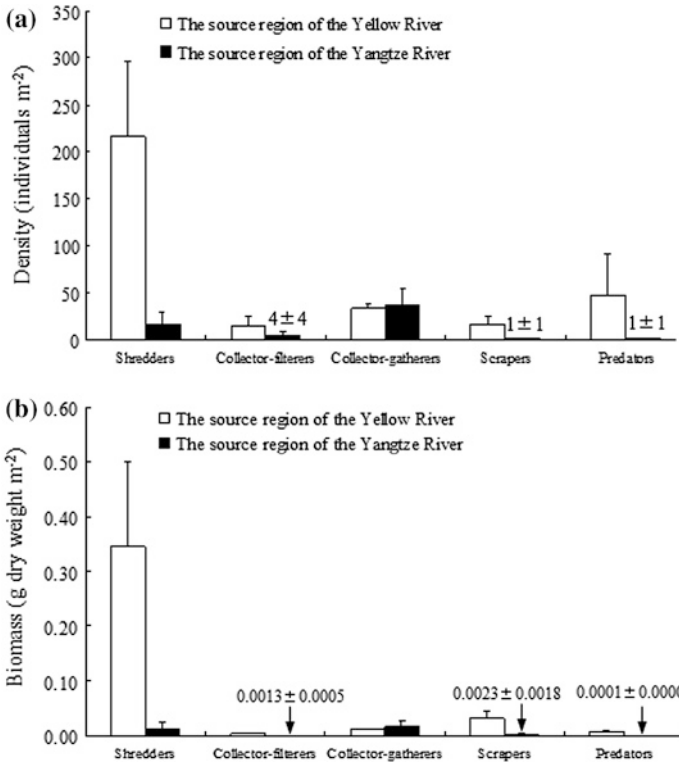
Fig. 9.16 Density (**a**) and biomass (**b**) of each functional feeding group of macroinvertebrates in the source regions of the Yellow River and the Yangtze River (Mean ± SE)

With regard to functional groups, shredders were the predominant group in the source region of the Yellow River, accounting for 65.7 % of the total density and 87.1 % of the total biomass (Fig. 9.16). In the source region of the Yangtze River, collector-gatherers and shredders were the dominant groups. Collector-gatherers made up 62.0 % of the total density and 50.6 % of the total biomass, while shredders made up 28.2 % of the total density and 40.7 % of the total biomass.

9.5.2.3 Discussion

Aquatic insects dominated macroinvertebrate assemblages in both source regions. Similar findings have been found for the mid-lower reaches of the trunk streams (Pan et al. 2011). Both potamophilic (water loving) and psychrophilic (cold-tolerant) species occurred in both source regions. The potamophilic taxa included *Ephemeroptera*, *Simuliidae*, *Rheotanytarsus*, *Stictochironomus*

Table 9.3 Comparisons of species richness, density and biomass of macroinvertebrates in a main channel with no macrophytes and a riparian wetland

Study site	Number of species	Density (individuals m^{-2})	Biomass (g dry weight m^{-2})
Mainstream of a tributary without macrophytes	7	11 ± 11	0.0051 ± 0.0051
Riparian wetland	14	232 ± 92	0.1374 ± 0.0651

and *Xenochironomus* and the psychrophilic species included *Stylaria lacustris*, *Limnodrilus grandisetosus* and *Limnodrilus profundicola*. The existence of psychrophilic species is consistent with the high altitude and low water temperature of the regions. In comparison with the source region of the Yangtze River, higher macroinvertebrate diversity in the source region of the Yellow River can be ascribed to more extensive wetlands which support more benthic biota, and lower altitudes where environmental conditions such as climate, oxygen concentration and atmospheric pressure are more favourable to biota.

In areas of limited human disturbance, macroinvertebrate assemblages are influenced primarily by habitat conditions such as water velocity (Brooks et al. 2005), substrate size (Quinn and Hickey 1990), bank morphology and other geomorphological considerations. Variance of macroinvertebrate assemblages in the Upper Yellow and Yangtze rivers is considered to reflect different habitat conditions. Among all measured parameters in the two river source regions, the difference of sediment concentration was particularly prominent. The sediment concentration of the source region of the Yangtze River was six times that of the source region of the Yellow River. High-sediment concentration limits the light needed for algal and macrophyte growth, and sand pellets may damage the cell walls of phytoplankton (Pan et al. 2009; Wang et al. 2006). These effects not only constrain the food available to macroinvertebrates, but macrophytes also create significant horizontal and vertical heterogeneities that provide a physical template for distinct niches (Rooke 1984) and serve as a refuge against predators (Pan et al. 2011). Moreover, macrophytes can serve as a site for oviposition (Scheffer 2004) and provide chances for snails to crawl on the air–water interface (particularly for pulmonates) (Wang et al. 2006).

Table 9.4 Comparisons of species richness, density and biomass of macroinvertebrates upstream and downstream of Jiangrang power station

Study site	Number of species	Density (individuals m^{-2})	Biomass (g dry weight m^{-2})
The upstream of Jiangrang hydroelectric power station	10	59 ± 30	0.0100 ± 0.0053
The downstream of Jiangrang hydroelectric power station	8	43 ± 31	0.0075 ± 0.0050

Further evidence of the role played by macrophytes in supporting large and diverse macroinvertebrate assemblages comes from a comparison between assemblages in riparian wetland sites and in the main channel of a tributary with no macrophytes (Table 9.3). The density and biomass of macroinvertebrates in the riparian wetland were 21 and 27 times higher than in a tributary channel with no aquatic vegetation.

9.5.3 Impacts of Human Activities on Macroinvertebrate Communities

Flow regulation impacts in the two river source regions are restricted to a few run-off hydroelectric power stations. To measure any effect of these dams on aquatic ecosystems, macroinvertebrate assemblages were compared upstream and downstream of the Jiangrang run-off hydroelectric power station in Maqin County. This dam has a much smaller water storage capacity than reservoir hydroelectric dams, raising the water level by less than 1 m. Only small differences were seen between species richness, density and biomass of macroinvertebrates upstream and downstream of the dam (Table 9.4). The limited impact noted in this instance reflects the fact that run-off hydroelectric power stations have a less adverse impact on the natural flow regime than reservoir power stations.

Human-induced wetland degradation presents a significant threat to aquatic ecosystems in the river source regions, as wetlands are able to support a far greater abundance of benthic individuals than habitats without vegetation (e.g. Sharma and Rawat 2009; Tarr et al. 2005; see Table 9.3). Furthermore, grassland degradation and erosion in recent years have induced serious soil loss, increasing sediment concentration in rivers (Luo and Tang 2003), which in turn decreases the abundance and diversity of aquatic biota. Measures to counteract these trends include conservative grazing and vegetation management.

9.5.4 Summary

Taxa number, density and biomass of macroinvertebrates in the source region of the Yellow River were, respectively, 1.7, 5.6 and 12.9 times higher than in the source region of the Yangtze River. Compared with the source region of the Yellow River, lower benthic animal resources in the source region of the Yangtze River were ascribed to higher altitude, higher sediment concentration and wetland degradation. Vegetation management and wetland restoration aiming to reduce soil erosion and maintain water supply are key considerations in the ecological protection of these source regions, which are critical to maintaining healthy ecological conditions throughout the entire basins.

9.6 Conclusions

This chapter has outlined concerns for the degradation and protection of the diverse wetlands and aquatic ecosystems in the Sanjiangyuan region. Degradation of the Ruoergai peatlands is causing environmental suffering (i.e. desertification, grassland loss and water shortage), impacting upon local herdsmen. Moreover, it is affecting the supply of water to the Upper Yellow River. Beyond question, wetland formation and evolution warrant further investigation in relation to global climate warming. The use of macroinvertebrates as an indicator species for the status of aquatic ecosystems in this region has indicated the range of conditions in the Upper Yellow and Yangtze Rivers. Undoubtedly, human activities such as overstocking, road construction, hydropower development, peat mining and tourism are impacting upon local aquatic ecosystems. These pressures accentuate concerns raised by the large-scale development strategy in West China. Further work is required to assess the influences of human activities on wetland and aquatic ecology in order to improve future prospects for wetland areas.

References

Baidu Encyclopedia. (2014). Chinese army across the grassland. http://baike.baidu.com/subview/180650/11248854.htm?fr=aladdin (in Chinese).

Barbier, E. B., Acreman, M., & Knowler, D. (1997). *Economic valuation of wetlands: A guide for policy makers and planners*. Gland, Switzerland: Ramsar Convention Bureau.

Blue, B., Brierley, G., & Yu, G. (2013). Geodiversity in the Yellow River Source Zone. *Journal of Geographical Science, 23*, 775–792.

Brierley, G. J., Yu, G. A., & Li, Z. W. (2016). Geomorphic diversity of rivers in the Upper Yellow River Basin. In G. J. Brierley, X. Li, C. Cullum, et al. (Eds.), *Landscape and ecosystem diversity, dynamics and management in the Yellow River Source Zone*. (pp. 59–78). Berlin: Springer.

Brinson, M., & Malvarez, A. I. (2002). Temperate freshwater wetlands: Types, status and threats. *Environment Conservation, 29*, 115–133.

Brooks, A. J., Haeusler, T. I. M., Reinfelds, I., et al. (2005). Hydraulic microhabitats and the distribution of macroinvertebrate assemblages in riffles. *Freshwater Biology, 50*, 331–344.

Cao, L., & Fox, A. D. (2009). Birds and people both depend on China's wetlands. *Nature, 460*, 173.

Feng, Y. Z., Yang, G. H., Wang, D. X., et al. (2009). Ecological stress in grassland ecosystems in source regions of Yangtze, Yellow and Lancang Rivers over last 40 years. *Acta Ecologia Sinica, 29*, 492–498. (in Chinese).

Gao, J. G. (2016). Wetland and its degradation in the Yellow River Source Zone. In G. J. Brierley, X. Li, C. Cullum, et al. (Eds.), *Landscape and ecosystem diversity, dynamics and management in the Yellow River Source Zone*. (pp. 209–232). Berlin: Springer.

Gao, J., Li, X., Brierley, G., et al. (2013). Geomorphic-centered classification of wetlands on the Qinghai-Tibet Plateau, Western China. *Journal of Mountain Science, 10*, 632–642.

Guo, L. Y., Wu, R., Wang, Q. C., et al. (2008). Influence of climate change on grassland productivity in Xinghai County in the source regions of Yangtze River. *Chinese Journal of Grassland, 32*, 5–10. (in Chinese).

Huang, H. Q., Liu, X. F., Brierley, G. J., et al. (2016). Hydrology of the Yellow River Source Zone. In G. J. Brierley, X. Li, C. Cullum, et al. (Eds.), *Landscape and ecosystem diversity, dynamics and management in the Yellow River Source Zone*. (pp. 79–100). Berlin: Springer.

Krebs, C. J. (1978). *Ecology: The experimental analysis of distribution and abundance* (2nd ed.). New York: Harper and Row.

Li, L., Li, F. X., Guo, A. H., et al. (2006). Study on the climate change trend and its catastrophe over "Sanjiangyuan" region in recent 43 years. *Journal of Natural Resources, 21*, 79–85. (in Chinese).

Li, Y., Yan, Z., Gao, S., et al. (2011). The peatland area change in past 20 years in the Ruoergai Basin, eastern Tibetan Plateau. *Frontier Earth Science, 5*, 271–275.

Li, Z.-W., Wang, Z.-Y., Brierley, G., et al. (2015). Shrinkage of the Ruoergai Swamp and changes to landscape connectivity, Qinghai-Tibet Plateau. *Catena, 126*, 155–163.

Liu, S. Y., Zhang, Y., Zhang, Y. S., et al. (2009). Estimation of glacier runoff and future trends in the Yangtze River source region, China. *Journal of Glaciology, 55*, 353–362.

Luo, X. Y., & Tang, W. J. (2003). Eco-environmental problems at source area of the Yangtze River and countermeasures for their prevention and treatment. *Journal of Yangtze River Scientific Research Institute, 20*, 47–49. (in Chinese).

Morse, J. C., Yang, L. F., & Tian, L. X. (1994). *Aquatic insects of China useful for monitoring water quality*. Nanjing: Hohai University Press.

Nicoll, T., Brierley, G., & Yu, G. (2013). A broad overview of landscape diversity of the Yellow River Source Zone. *Journal of Geographical Science, 23*, 793–816.

Ning, H., Li, G., & Wang, J. (2011). Change features of pan evaporation in Maqu area in the upper reaches of Yellow River in recent 40 years. *Journal of Arid Land Resources and Environment, 25*, 113–117. (in Chinese).

Niu, Z. G., Zhang, H. Y., & Gong, P. (2011). More protection for China's wetlands. *Nature, 471*, 305.

Niu, Z. G., Zhang, H. Y., Wang, X. W., et al. (2012). Mapping wetland changes in China between 1978 and 2008. *Chinese Science Bulletin, 57*, 2813–2823.

Pan, B. Z., Wang, H. J., Liang, X. M., et al. (2009). Factors influencing chlorophyll *a* concentration in the Yangtze-connected lakes. *Fresenius Environmental Bulletin, 18*, 1894–1900.

Pan, B. Z., Wang, H. J., Liang, X. M., et al. (2011). Macrozoobenthos in Yangtze floodplain lakes: Patterns of density, biomass and production in relation to river connectivity. *Journal of the North American Benthological Society, 30*, 589–602.

Pan, B. Z., Wang, Z. Y., & Xu, M. Z. (2012). Macroinvertebrates in abandoned channels: Assemblage characteristics and their indications for channel management. *River Research and Applications, 28*, 1149–1160.

Qiu, P. F., Wu, N., Luo, P., et al. (2009). Analysis of dynamics and driving factors of wetland landscape in Zoige, eastern Qinghai-Tibetan Plateau. *Journal of Mountain Science, 6*, 42–55.

Quinn, J. M., & Hickey, C. W. (1990). Magnitude of effects of substrate particle size, recent flooding, and catchment development on benthic invertebrates in 88 New Zealand rivers. *New Zealand Journal of Marine and Freshwater Research, 24*, 411–427.

Rooke, B. J. (1984). The invertebrate fauna of four macrophytes in a lotic system. *Freshwater Biology, 14*, 507–513.

Safran, E. B., Bierman, P. R., Aalto, R., et al. (2005). Erosion rates driven by channel network incision in the Bolivian Andes. *Earth Surface Processes and Landforms, 30*, 1007–1024.

Scheffer, M. (2004). *Ecology of Shallow Lakes*. Kluwer Dordrecht: Academic Publishers.

Schumm, S. A., Michael, D. H., & Chester, C. W. (1984). *Incised channels: Morphology, dynamics, and control*. Littleton, Colorado: Water Resources Publications.

Sharma, R. C., & Rawat, J. S. (2009). Monitoring of aquatic macroinvertebrates as bioindicator for assessing the health of wetlands: A case study in the Central Himalayas, India. *Ecological Indicators, 9*, 118–128.

Shen, S., Wang, J., & Yang, M. (2003). Principal factors in retrogression of the Zoige Plateau marsh wetland. *Journal of Sichuan Geology, 23*, 123–125. (in Chinese).

Sheng, H. Y., Yang, G. H., Wang, D. X., et al. (2007). Current status of grassland resources utilization and the corresponding solutions in a sustainable development context in the regions of three river source. *Journal of Northwest Sci-Tech University of Agriculture and Forest, 35*, 147–153. (in Chinese).

Shi, X. H., Qin, N. S., Xu, W. J., et al. (2007). The variations characteristic of the runoff in the source regions of the Yangtze River from 1956 to 2004. *Journal of Mountain Science, 25*, 513–523. (in Chinese).

Tane, H., Li, X. L., & Chen, G. (2016). Ecogenesis of the Huang He (Yellow River) headwaters. In G. J. Brierley, X. Li, C. Cullum, et al. (Eds.), *Landscape and ecosystem diversity, dynamics and management in the Yellow River Source Zone*. (pp. 275–330). Berlin: Springer.

Tarr, T. L., Baber, M. J., & Babbitt, K. J. (2005). Macroinvertebrate community structure across a wetland hydroperiod gradient in southern New Hampshire, USA. *Wetlands Ecology and Management, 13*, 321–334.

Tooth, S., McCarthy, T. S., Brandt, D., et al. (2002). Geological controls on the formation of alluvial meanders and floodplain wetlands: The example of the Klip River, Eastern Free State, South Africa. *Earth Surface Processes and Landforms, 27*, 797–815.

Wang, H. J., Pan, B. Z., Liang, X. M., et al. (2006). Gastropods on submerged macrophytes in Yangtze Lakes: Community characteristics and empirical modelling. *International Review of Hydrobiology, 91*, 521–538.

Wang, G. X., Li, N., & Hu, H. C. (2009). Hydrologic effect of ecosystem responses to climatic change in the source regions of Yangtze River and Yellow River. *Advances in Climate Change Research, 5*, 202–208. (in Chinese).

Wang, Z. Y., Lee, J. H. W., Cheng, D. S., et al. (2008). Benthic invertebrates investigation in the East River and habitat restoration strategies. *Journal of Hydro-environment Research, 2*, 19–27.

Yan, Y. J., & Liang, Y. L. (1999). A study of dry-to-wet weight ratio of aquatic macroinvertebrates. *Journal of Huazhong University of Science & Technology (Natural Science Edition), 27*, 61–63. (in Chinese).

Yan, Z., & Wu, N. (2005). Rangeland privatization and its impacts on the Ruoergai wetlands on the Eastern Tibetan Plateau. *Journal of Mountain Science, 2*, 105–115.

Yang, Y. (1999). Ecological environment deterioration, mire degeneration and their formation mechanism in the Zoige Plateau. *Journal of Mountain Science, 17*, 318–323. (in Chinese).

Yang, Z. P., Gao, J. X., Zhou, C. P., et al. (2011). Spatio-temporal changes of NDVI and its relation with climatic variables in the source regions of the Yangtze and Yellow rivers. *Journal of Geographical Sciences, 21*, 979–993.

Zhang, S. F., Hua, D., Meng, X. J., et al. (2011). Climate change and its driving effect on the runoff in the "Three-River Headwaters" region. *Journal of Geographical Sciences, 21*, 963–978.

Zhang, X.Y., Lu, X.G. (2010). Multiple criteria evaluation of ecosystem services for the Ruoergai Plateau marshes in southwest China. *Ecological Economics*, *69* (7), 1463–1470.

Zhou, D. M., Gong, H. L., Wang, Y. Y., et al. (2009). Driving forces for the marsh wetland degradation in the Honghe National Nature Reserve in Sanjiang Plain, Northeast China. *Environmetal Modelling & Assessment, 14*, 101–111.