A century-scale human-induced hydro-ecological evolution of wetlands of two large river basins in Australia (Murray) and China (Yangtze): Development of an adaptive water resource management framework

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Abstract

Recently, the provision of food and water resources of two of the world's large river basins, the Murray and the Yangtze, has been significantly altered through widespread landscape modification. Long-term sedimentary archives, dating back for some centuries from wetlands of these river basins, reveal that rapid, basin-wide development has reduced the resilience of biological communities, resulting in considerable decline in ecosystem services, including water quality. Large-scale human disturbance to river systems, due to river regulation during the mid-20th century, has transformed the hydrology of rivers and wetlands, causing widespread disturbance to aquatic biological communities. Changes to cladoceran zooplankton (water fleas) were used to assess the historical hydrology and ecology of three Murray and Yangtze River wetlands over the past century. Subfossil assemblages of cladocerans retrieved from sediment cores (94 cm, 45 cm and 65 cm) of three wetlands: Kings Billabong (Murray), Zhangdu and Liangzi Lakes (Yangtze), showed strong responses to hydrological changes in the river after the mid-20th century. In particular, river regulation caused by construction of dams and weirs together with river channel modifications, has led to significant hydrological alterations. These hydrological disturbances were either: 1) a prolonged inundation of wetlands, or 2) reduced river flow, both of which caused variability in wetland depth. Inevitably, these phenomena have subsequently transformed the natural wetland habitats, leading to a switch in cladoceran assemblages to species preferring poor water quality, and in some cases to eutrophication. An adaptive water resource management framework for both of these river basins has been proposed to restore or optimize the conditions of wetland ecosystems impacted by these 20th century human disturbance and climate change.

1. Introduction

There has been a worldwide growing awareness of the value of healthy flow regimes (hydrology), as key 'drivers' of the ecology of large rivers and their associated floodplain wetlands (Bedford, 1996; Puckridge et al., 1998; Richter et al., 2003). Natural flows maintain ecological processes which include valuable biodiversity in the ecosystems of the river system and its associated floodplain wetlands. The river channels connecting to floodplain wetlands discharge water, mixed with rich sources of carbon, energy, and nutrients, from the river and its catchments, to the wetlands (Bunn and Arthington, 2002; Maddock et al., 2004). In addition, the allochthonous sources of organic matter deposited during flood pulses support reproduction and growth of biota (Junk et al., 1989; McGowan et al., 2011). Integration of local autochthonous production, including algae and inputs from the riparian zone during pulse events, further supports available energy for higher trophic levels (Thorp and Delong, 1994). As a result, large rivers and their associated floodplain wetlands are a potential source of ecosystem goods and services to humans; for example, flood attenuation, water purification, fisheries and other foods, and a range of marketable goods (Poff et al., 2003).

However, the flow regime of large rivers has been consistently modified to meet demands of water for mono-agriculture and hydroelectricity (Nilsson and Berggren, 2000; Davis et al., 2015). Many floodplain wetlands have been transformed into a new regime as a result of over-allocation of water to off-stream uses, or other alterations to the natural flow regimes of large river systems (Walker, 1985). The construction of dams and dykes obstruct migration pathways for fish between the river channels and wetlands, and the newly built reservoirs trap water-borne sediment. The diversion of water may lead to historical channels becoming permanently or intermittently dry. Subsequent inundation of upstream riparian zones increases soil anoxia, often extinguishing entire plant and animal populations and altering the riparian environment. Furthermore, downstream hydrological and geomorphological alterations can reduce groundwater recharge, and modify the pattern of sediment exchange between rivers and wetlands (Nilsson and Berggren, 2000).

Whilst it is recognised that widespread human disturbances have currently caused variation in biological and species diversity in many floodplain wetlands worldwide (Zhang et al., 1999; Maddock et al., 2004), the response of biological diversity to these disturbances is variable. Some floodplain wetlands have a reduced diversity index following the disturbance, while in other wetlands, the disturbance has paradoxically led to increased diversity index (Power et al., 1996). In either case, the nature of these disturbances over time and space have altered habitat stability, affecting species diversity and ecosystem functioning, and are potentially threatening the historical identity of these wetlands (Dumbrell et al., 2008; Biswas and Malik, 2010).

Further, the threats posed by widespread hydrological alterations to large rivers are often ignored or sidelined, with the demand for energy, irrigated food production, and industrial use for the projected growth of human population being, given a higher priority (Power et al., 1996). It is important, therefore, that while water allocation plans are being formulated to provide greater water security for immediate community use, it will be essential that understanding of the considerable socioeconomic benefits provided by healthy floodplain wetland ecosystems associated with these large rivers are not lost, and that degraded ecosystems are restored for the benefit of future generations (Poff et al., 2003). Key socio-economic benefits, such as water purification, flood abatement and carbon sequestration, all of which are maintained by wetland biodiversity and ecosystem functioning, will thus not be impaired if care is given to the wetlands of large river basins to ensure that they are not lost or degraded (Zedler and Kercher, 2005).

Recent evidence suggests that a significant proportion of the national economy of Australia and China has been generated by two of their large river systems, the Murray and the Yangtze Rivers respectively. These rivers have contributed to a range of ecosystem services, including food, mineral, and water resources, to the communities living in the river basins (Palmer et al., 2008; Zhang et al., 2015). However, because water has been abstracted heavily for irrigation, hydroelectricity, and industrial development in both river basins, there has been widespread disruption in the hydrology of the rivers, for example the frequency, timing, and volume of flow in the main river and associated river channels linking to adjacent floodplain wetlands (Walker et al., 1995). This varying of natural flow regimes has interrupted natural flood pulses leading to changes in hydraulic residence time, wetland depth, nutrient inputs and sediment cycling, in addition to changing the structure, function, and species diversity of downstream floodplain ecosystems (Power et al., 1996; Kingsford, 2000; Chen et al., 2011; Kattel et al., 2015).

There are some parallels in the historical experience of these two river systems, which makes this simultaneous study more appropriate. Records show that following the arrival of Europeans in Australia in the early 1900s, the Murray River system began to be regulated for irrigation, hydroelectricity and navigation (Walker, 1985). The wetlands connected to the river were either inundated as water storage basins, or dehydrated due to upstream water extraction or diversion of connecting channels. Deforestation of the catchment became widespread during the expansion of agriculture. As a result, the majority of wetlands have been subjected to significant bank erosion and sedimentation (Gell et al., 2009). In China, similar contemporary pressure has been placed on the Yangtze River system. Similar large scale modifications of rivers and wetlands occurred during the 1950s–1970s. Riparian floodplain and wetland habitats across the Yangtze River Basin were extensively reclaimed for agriculture and rural development by the construction of dykes. This resulted in a significant loss of vegetation in the upper reaches of the Yangtze, followed by soil erosion and siltation of downstream wetlands (Yin and Li, 2001). The river sediment load of the Yangtze River between the 1960s and 1970s alone was more than 450 Mt/year (Yang et al., 2011a, b). Consequently, many lakes experienced reduced flood retention capacity due to disconnection from the main channel of the Yangtze River by construction of embankments and sluice gates in the river channels, which was subsequently followed by widespread eutrophication (Yu et al., 2009; Zhang et al., 2012). Because of alterations in natural flood pulses, ephemeral and temporary lakes tended to have fewer taxa than semi-permanent channels or terminal lake habitats (Sheldon et al., 2002). Excessive water abstraction or river-flow regulation in the Yangtze River disrupted natural variability in connectivity and hydrological regimes, consequently threatening ecological integrity, including the biodiversity of the floodplain system (Sheldon et al., 2002, Yang et al., 2006).

Studies show that the Murray and Yangtze River wetlands have lost significant density of submerged littoral macrophytes over the past century (Reid et al., 2007; Yang et al., 2008). For example, the subfossil assemblages of diatoms and cladocerans in the floodplain wetlands of the mid-reaches of the Murray River indicate a collapse of submerged vegetation coincident with the first appearance of the introduced conifer, *Pinus radiata* (Reid et al., 2007). Similarly, the multi-proxy responses, including diatoms and physico-chemistry of sediment of the Taibai Lake (lower Yangtze), show that after the 1990s, the lake shifted to hyper-eutrophic condition. This was thought to be due to increased dominance of algal biomass and a reduced density of submerged macrophytes (Liu et al., 2012). There has been a characteristic state shift in wetlands of both river systems due to the changes in the dynamics of submerged vegetation (Reid et al., 2007; Yang et al., 2008). The submerged vegetation in wetlands reduces phytoplankton by shading the substrate and competing for underwater light sources needed for photosynthesis, consequently improving the water quality by stabilising sediment resuspension (Jeppesen and Sammalkorpi, 2002; Folke et al., 2004). However, the characteristic alternative stable states of ecosystems, which are thought to be buffered by naturally occurring hydrology, nutrient enrichments and submerged vegetation dynamics in large river floodplain wetlands, (e.g. Scheffer et al., 1993) have been substantially disrupted in recent decades. Today, the prior, undisturbed ecological state of the Murray and Yangtze River wetlands has been difficult to understand, due to the effects of multiple stressors, including human disturbances and climate change. For instance, following river regulation (1950s), the wetlands of Yangtze have become eutrophic, even in the presence of submerged vegetation (Qin et al., 2009).

Understanding the effects of disruption in natural hydrological regimes of the Murray and Yangtze rivers on diversity and community structure of consumers, such as cladcoeran zooplankton (water fleas) in the adjacent floodplain wetlands, is crucial to assessing wetland ecosystem health. Both Australia and China have faced increasing challenges in addressing shortages of water and food supplies, resulting from reduced water flows in these catchments. A long term monitoring of wetlands exposed to hydrological disturbance is important to ensure maintenance of ecosystem services, by identifying the causes of degradation and using effective and adaptive restoration measures.

The subfossil cladocerans have responded to past climate change, eutrophication, and water pollution in many shallow lakes (Jeppesen et al., 2001). Some cladocerans are also significant indicators of locally associated hydrological factors, including the river flow, lake water depth, sediment properties, macrophyte cover, and biotic interactions (Nevalainen, 2011). Recently, Pawlowski et al. (2015) have documented cladoceran-inferred palaeo-hydrology, including the formation of meandering channels, hydraulic characteristics and water level change in the oxbow lake, of the Grabia River (central Poland) during the late Glacial and Holocene periods. Whereas the role of fossil cladocerans is becoming increasingly significant for understanding the past hydrology of large river basins elsewhere, understanding cladoceran response to long term hydrology and water level change of wetlands (eco-hydrology) in the Murray and Yangtze rivers currently is limited. In this paper, we aim to examine three sites: the Murray and Yangtze River floodplain wetlands, Kings Billabong (Murray), and the Zhangdu and Liangzi Lakes (Yangtze), each of which have been exposed to large scale human-induced hydrological disturbances during the 20th century, as inferred by subfossil assemblage and diversity of cladocerans.

Understanding the linkage between eco-hydrology and adaptive water resource management, or 'socio-hydrology', is becoming increasingly important in large river basins, since interaction between people and water systems is fundamental to long-term community and ecological health (Nilsson and Berggrern, 2000). However, until recently the use of palaeoecology (subfossil cladocerans) has been rarely examined in rapidly changing environments, nor has its role in socio-hydrology been fully exploited. A participatory approach of water resource management has been found to be successful in many regulated environments (Falkenmark, 2004), and such an approach appears to be sustainable in nature and to provide increased levels of integration between natural and social scientists, land and water users, land and water managers, planners and policy makers across spatial scales (Macleod et al., 2007). This type of integrated platform is crucial for learning and exchange of knowledge among stakeholders for successful management outcomes (Pahl-Wostl, 2009). Based on scientific evidence of ecological and hydrological transitions responded to by cladocerans, we have proposed in this paper an adaptive water resource management framework for the Murray and Yangtze River wetlands. Such management framework is expected to potentially contribute to the resolution of critical issues of the management of the wetlands of both river basins.

2 Study areas

2.1 Kings Billabong (Murray River)

Kings Billabong (34° 14' S & 142° 13' E) is a shallow (~1.8 m deep) wetland (210 ha), located along the River Murray near Mildura (northwest Victoria), Australia (Fig. 1). Kings Billabong was once an important source of food and water for the *Nyeri Nyeri* Aboriginal Community. The intensification of agriculture around Kings Billabong by early European settlers began in 1891 and continued until 1923. Initially in 1896, Kings Billabong was used as a pumping station and was converted to water storage basin (Lloyd, 2012). Modification of the landscapes around the billabong and construction of dams, including the series of locks and weirs for upstream water storages, have significantly altered the natural flow regime of the River Murray which feeds Kings Billabong (Gippel and Blackham, 2002). The hydrology and, in particular, the variability of flows which include duration and water retention time in the river, have substantially influenced the volume of water in Kings Billabong (Lloyd, 2012). Since formal regulation of the River Murray began in 1927, with construction of Lock 11 at Mildura and Lock 15 at Euston in 1937, downstream river flows and naturally occurring flood pulses have altered in many wetlands, including Kings Billabong (Gippel and Blackham, 2002). The artificial flooding linking Kings Billabong to the weir pool of Lock 11 has led this wetland becoming permanently inundated.

The first sign of impact due to river regulation on Kings Billabong was observed as widespread dieback of River Red Gum (RRG) forests and the establishment of fringing Cumbungi (*Typha* sp.) vegetation (Parks Victoria, 2008). Logging of RRG forests was intensified in the region until the 1950s, with the timber used to fuel steam-operated pumps and paddleboats along the River (Parks Victoria, 2008). The life cycle of native aquatic biota in the wetlands around the lower Murray has thus become disrupted due to the variation in natural wet-dry events caused by river regulation (Ellis and Meredith, 2005). Increased distribution range of exotic fish and weeds were also observed following regulation. For example, in a survey of native and exotic fish in Kings Billabong, *Gambusia* (an exotic species), comprised 35% of the total species collected (Ellis and Meredith, 2005).

Apart from human activity, climate change has also impacted the condition of Kings Billabong. Average water temperatures in the Southeast Australia have risen over the past 60 years and there has been a decrease of 40% in the total rainfall in the region (Cai and Cowan, 2008). This regional variability in climate change has led to significant changes in river flow, wetland volume, thermal structure and alteration of catchment inputs, all of which are influenced by a marked increase in frequency and intensity of extreme events such as droughts and floods (Lake et al., 2000).

2.2 Zhangdu Lake (Yangtze River)

Zhangdu Lake (30° 39' N & 114° 42' E) is a floodplain wetland (1.2 m deep) of the Yangtze River system, which is located in Hubei Province, central China (Fig. 2). During high river flows, Zhangdu Lake previously received flood pulses from the Yangtze River. However, the lake was disconnected from the Yangtze River in the 1950s, due to the construction of dams and widespread land reclamation across the catchment. By the 1980s, the shoreline of Zhangdu Lake had been significantly modified as a result of the increased reclamation activity and construction of water conservancy infrastructure, which commenced in the 1970s. In 2005, after the reclamation of 50 square km of shoreline, funding from the World Wildlife Fund enabled Zhangdu Lake to be seasonally reconnected with Yangtze River for the purpose of habitat restoration. This lake now has an area of 35.2 km^2, with an average depth of 1.2 m and a maximum depth of 2.3 m. The watershed lies within the northern subtropical monsoon zone, with a mean annual temperature of 16.3°C, mean annual rainfall of 1150 mm and evaporation of 1525.4 mm. The terrain slopes gently with an elevation of 16 to 21 m. The main inflows of Zhangdu Lake are from the Daoshui River in the west and the Jushui River in the east. Water drains from the lake into the Yangtze River via an artificial channel in the south-eastern corner. Historically, Zhangdu Lake has interacted not only with the Yangtze River when the water level is high, but it has also connected with surrounding lakes, Qi Lake and Tao Lake, during flood events (Zhang et al., 2013). However, due to the construction of dams, dykes and land reclamation, it became disconnected from the river in the 1950s. Water conservancy and reclamation construction reached a peak in the 1970s, attaining its current finished and formed shape during the 1980s. Following the mid-20th century reclamation phase, the rate of carbon accumulation in Zhangdu Lake has increased, possibly due to an increase in shallow marginal areas favouring the growth of carbon rich macrophytes (Dong et al., 2012). However, the ecological impacts of disconnection from the river in Zhangdu Lake have become severe. Wild fishery production has reduced from 95% in 1949 to less than 5% in 2002, and fish diversity has decreased, from 80 species in 1950s to 52 species at present (Wang et al., 2005). To address this decline, funding from the World Wildlife Fund (WWF) in 2005 reconnected Zhangdu Lake with the Yangtze River.

2.3 Liangzi Lake (Yangtze River)

Liangzi Lake (30°3' N, 114°26' E) is a shallow wetland (3-5 m deep), located in southeast region of Hubei province on the southern bank of the middle reaches of the Yangtze River. The lake area is 304.3 km^2 with a drainage area of 3,265 km^2. The lake has an elevation of 20 meters and is 31.7 km in length with a mean width of 9.6 km (Fig. 2). The lake connects to

Yangtze River via a 43.3 km river canal (Xie et al., 2001). Since 1992, the western part of the lake, approximately 6000 ha in area with mean depth of 4.2 m, has been separated from the main lake by a 2000 m nylon screen (mesh size 20 mm) for the purpose of aquaculture. Water exchange occurs easily between the two parts of the lake. Intensive stocking with commercial fish, including grass carp *Ctenopharyngodon idella* (Val.), bighead carp *Aristichthys nobilis* (Richardson) and silver carp *Hypopthalmichthys molitrix* (*Cuvier* and *Valenciennes*), is common in the western part of the Liangzi Lake (Xie et al., 2001). Because of grass carp stocking, macrophytes were completely eliminated from the western part of the lake. However, areas of less intensive aquaculture still maintain an abundant density of submersed macrophytes, with *Potamogeton maackianus* (A. Bennet) as the dominant species (Xie et al., 2001). Apart from fisheries, Liangzi Lake provides significant services for drinking water, irrigation, transportation and recreation to the people living around the four large cities, Wuhan, Huangshi, Ezhou and Xianning Liangzi. Recently, one of the largest foreign investment projects to date in central and southern China, the Hubei Liangzi Lake International Golf Club, has opened a training centre at the edge of the lake.

3 Frameworks for changes in hydrology of Murray and Yangtze River wetlands

Figure 3 presents hydrological frameworks for both Murray and Yangtze River systems. This diagram shows the deviation of baseline flows of the two rivers and associated wetlands before and after regulation. Construction of weirs in the lower Murray River during the 1920s and 1930s, and construction of dams in the Yangtze River during the 1950s to the 1970s, significantly altered peak flows and downstream wetland hydrology (Lloyd, 2012; Yang et al., 2011a, b).

Naturally occurring spring flood patterns in the River Murray, experienced prior to the construction of Lock 11 in 1927, have been altered by regulation, and as a result, the amount of water released to meet peak irrigation demands has changed (Lloyd, 2012). Increased demand for water has resulted in the flow of the Lower Murray River falling below the historical baseline (Fig. 3 A-i). Regulation for wetland permanency has led to the depth of Kings Billabong being above the historical baseline level (Fig. 3 A-ii).

In Zhangdu Lake, water levels were maintained through inflows from two rivers, the Daoshui River from the west and the Jushui River from the east, and outflow to the Yangtze River via by an artificial channel from the southeast corner of the lake. The water level was maintained by permanent connectivity between the Zhangdu Lake and the Yangtze River channels prior to the 1950s, but became disrupted by regulation (Fig. 3 B-i). The decline in annual discharge of the Yangtze River (−11%) after the 1950s (Yang et al., 2011a,b), has led to a reduction of the historical baseline flow of the river, subsequently reducing the baseline water level in Zhangdu Lake (Fig. 3 B-ii). The South-to-North Water Diversion Projects, in addition to wetland reclamation and construction of new dams, particularly after the 1970s-80s, has further altered the hydrology of Zhangdu Lake (Qin et al, 2009; Yang et al., 2010). However, the project initiated in 2005 by the World Wildlife Fund for Nature has recharged the channel hydrology and increased water level of Zhangdu Lake (Fig. 3 B ii).

4 Methods

4.1 Assessment of diversity and ecosystems of Murray and Yangtze River wetlands

The diversity and ecological conditions of the three floodplain wetlands, Kings Billabong, Zhangdu Lake and Liangzi Lake associated with two large river systems, were assessed using subfossil cladoceran zooplankton remains retrieved from lake sediments deposited over the past century. A high resolution subsampling of a 94 cm long core, collected from Kings Billabong, was carried out at 1 cm intervals.

In the case of Zhangdu Lake, a subsampling of a 45 cm long core was carried out at 1 cm intervals for up to 27 cm, and at 2 cm intervals for up to 45 cm respectively. For Liangzi Lake, the subsampling of 65 cm core was carried out at 2 cm intervals. Subsamples from all three lakes, weighing approximately 3-4 g each as wet sediment, were treated with 100 mL of 10% KOH solution, and heated at $60^{o}C$ on a hotplate for at least 45 minutes. Sieving of the sub-sample mixture was carried out through a 38 μm mesh. More than 200 identifiable cladoceran remains were enumerated at 400 x magnification from each subsample. Numbers were converted to individuals per g dry weight of sediment, followed by the calculation of relative proportion of the remains present in the sample (Kattel et al., 2008). Cladoceran taxa were identified following the procedures suggested by Frey (1986), Shiel and Dickson (1995), Zhu et al. (2005) and Szeroczyńska and Sarmaja-Korjonen (2007).

4.2 Dating

The age chronology was based on the standard ^{210}Pb dating for all sites (Appleby, 2001). For Kings Billabong, radionuclide activity was detected at 51 cm, while the radionuclide activities for Zhangdu and Liangzi Lakes were detected at 45 cm and 65 cm respectively. The age modelling of Kings Billabong can be found in detail in Kattel et al. (2015).

The sediment samples from Zhangdu and Liangzi Lakes were dated using ^{210}Pb and ^{137}Cs by non-destructive gamma spectrometry laboratory at the State Key Laboratory of Lake Science and Environment, NIGLAS. The activities of ^{210}Pb, ^{226}Ra and ^{137}Cs in samples were determined by counting with an Ortec HPGe GWL series well-type coaxial low background intrinsic germanium detector. The ^{137}Cs was used to identify the peak that indicated use of the 1963 nuclear bomb. This evidence was then used for developing a constant rate of supply (CRS) model to calculate ^{210}Pb chronology for the core. The important dates relevant to hydrological changes were indicated in the stratigraphy.

4.3 Numerical analyses

Dendrograms for subfossil cladoceran samples were produced in the TILIA Graph following the constrained incremental sums of squares (CONISS) analysis. Zonation of samples in the diagram was based on the chord-distance dissimilarity coefficients obtained in CONISS (Grimm, 1987). Indirect ordination techniques, such as detrended correspondence analysis (DCA) were used for identifying species alignments with samples over time (Hill and Gauch, 1980). DCA was run for sub-fossil cladoceran samples meeting 200 counts in each sample followed by running CA or PCA as per the gradient length of the first DCA axis (ter Braak, 1995). The CA and PCA sample scores were incorporated in the stratigraphy diagrams.

5 Results

5.1 Diversity of subfossil cladocerans (water fleas) in Murray and Yangtze River wetlands

The species richness (species count) of subfossil cladocerans was higher in the Murray River wetland than in the Yangtze River wetlands. More than 40 species of subfossil cladoceran were recorded from Kings Billabong, while core samples from Zhangdu Lake and Liangzi Lake had only 36 and 20 species, respectively. The most commonly recorded cladoceran taxa in Kings Billabong were *Bosmina meridionalis*, *Chydorus sphaericus*, *Biapertura setigera*, *Dunhevedia crassa, Biapertura affinis* and *Alona guttata* (Fig. 4) while the most commonly recorded taxa in Zhangdu Lake were, *Bosmina*, *Chydorus sphaericus* and *Sida crystallina*, and in the Liangzi Lake, *Bosmina*, *Acroperus harpae*, *Alona guttata*, *Alona rectangula* and *Chydrorus sphaericus* (Figs. 5 & 6).

The species diversity test of cladoceran counts over time by using the Hill's N2 diversity index reflected a small change in both river systems. The Hill's N2 diversity index assumes that the number of species in an ecosystem is uniformly distributed (Hill, 1973). In Kings Billabong, the N2 diversity index was low during the 1900s. However, prior to human disturbance of the river (c. 1870s), as well as in c. 1960s, the N2 diversity index was relatively high (Figure 5).

In Zhangdu Lake, the N2 diversity index prior to the construction of the dam (c. 1881-1954) was low compared to the post-dam construction period, during which time the taxa preferring disturbed environment increased (Fig. 5). Similarly, the N2 diversity index of Liangzi Lake during the earlier period (c. 1900-1930) was lower than post dam construction period in the Yangtze River (Fig. 6). Differences in responses of cladoceran diversity to regulation in Murray and Yangtze rivers as shown by the N2 diversity index suggest some degree of variations in disturbances between the Murray and Yangtze River systems. Unlike the occurrence of more severe and frequent disturbances in Kings Billabong following the arrival of early European immigrants, gradual and intermediate frequencies of disturbance in Yangtze River wetlands may have resulted in the increased species diversity of cladocerans following regulation similar to the condition described by the intermediate disturbance hypothesis model (Townsend and Scarsbrook, 1997).

5.2 Cladoceran responses to ecological and hydrological changes of Murray and Yangtze River wetlands

5.2.1 Kings Billabong

The subfossil assemblage of cladocerans in Kings Billabong showed four distinct changes in ecosystem. Until the 1890s, (Zone I) Littoral cladocerans such as *Dunhevedia crassa*, *Alona guttata*, *Chydorus sphaericus* and *Graptoleberis testudinaria* were the dominant species (Zone I). This period experienced a relatively low abundance of the planktonic species *Bosmina meridionalis* (Fig. 4). However, total littoral cladocerans gradually declined, while small littoral species such as *Alona guttata* became abundant during the period 1890 to 1950 (Zone II). During this time, an increasing density of planktonic *B. meridionalis* contributed to total planktonic cladocerans. Some *Daphnia* records (1950s-1970s) were also retrieved, and coincided with the timing of the 1956 flood in the River Murray (Zone III) (Fig. 4). Although total littoral cladocerans declined, some littoral species such as *Alona guttata* and *A. quadrangularis* were still abundant during this time. However, in the 1970s-2000s, planktonic *B. meridionalis* and littoral *A. guttata*, *Biapertura longispina*, *A. quadrangularis* and *Chydorus sphaericus* dominated, while the littoral *D. crassa* declined significantly. In the meantime, the frequency and density of cladoceran resting eggs also increased in the sediment (Fig. 4).

In Kings Billabong, the L:P ratios of cladocerans began to decline rapidly from about 75 cm depth (c.1930s) (Fig. 4). The subfossil assemblages of littoral and planktonic cladocerans responded to hydrological changes of the Murray River, together with subsequent changes of water level of Kings Billabong. The construction of Lock 11 in the Murray River near Mildura led to permanent inundation of Kings Billabong during the

1920s-1930s, the time of major hydrological shift (Fig. 4). Because of the expansion of the pelagic habitat in Kings Billabong, the assemblage of subfossil *Bosmina* increased (Fig. 4). Although the billabong was inundated, there was sustained increase in the abundance of some littoral cladocerans including *Alona guttata*, *Alona quadrangularis* and *Biapertura longispina*. Following the hydrological shift, Kings Billabong began to respond to this change with declining water quality. For example, littoral cladocerans such as *A. guttata* and *A. quadrangularis*, which prefer poor water conditions, were sustained together with *B. meridionalis*. However, the assemblage of the dominant littoral cladoceran, *Dunhevedia crassa,* which prefers clean water conditions, significantly declined following the hydrological shift, from pre-regulated, variable water levels to post-regulated, constant inundation, in Kings Billabong, due to the imposition of river regulation in 1927 (Fig. 4).

5.2.2 Zhangdu Lake

Three distinct ecosystem changes were observed in Zhangdu Lake, based on the subfossil assemblage of cladocerans from lake sediment. Planktonic cladocerans dominated the period c. 1880s-1960s (Zone I), when the planktonic *Bosmina* sp. was the most dominant species. During this time, the abundance of total littoral cladocerans declined, when only a few species, including those that characteristically occupy both littoral and planktonic habitats, such as *Chydorus sphaericus,* were present (Fig. 5). However, the major hydrological shift occurred during the c. 1960s-1980s (Zone II). Following the construction of dams across the Yangtze River channels (c. 1950s), sediments deposited in the dam contained increasing numbers of remains of the littoral cladocerans, where by some of the common species of cladocerans such as *Acroperus harpae*, *Alona guttata*, *Alona rectangula*, *Chydorus sphaericus*, *Graptoleberis testudinaria* and *Sida crystallina* were gradually becoming dominant (Fig. 5). The abundance of littoral cladoceran species such as *A. harpae*, *Alona intermedia*, *Alona affinis*, *Kurzia lattissima*, *Leydigia leydigi*, *A. guttata*, *Camptocercus rectirostris* and *Disparalona rostrata* increased further during the c. 1990s-2000s (Zone III) indicating a significant change in the system. In addition, the concentration of the cladoceran resting eggs increased during this time (Fig. 5).

In the Zhangdu Lake, increased diversion of the water from the Yangtze River, during the 1960s-70s because of the construction of dams, led to significant decline in water level. This resulted in a decrease of water depth around the lake margins, consequently providing suitable conditions for the increased growth of littoral vegetation and associated habitat for cladocerans. In response, the abundance of littoral cladocerans, including *Alona affinis*, *Alona guttata*, *Alona intermedia*, *Camptocercus rectirostris*, *Kurzia latissima* and *Leydigia leydigi*, increased with high L:P ratios (Fig. 5). Smaller *Alona* such as *A. guttata*, *A. rectangula* and *A. intermedia* showed a distinct presence during this time (Fig. 5).

5.2.3 Liangzi Lake

Four distinct ecosystem changes were observed in Liangzi Lake, based on the subfossil assemblage of cladocerans retrieved from lake sediments. Prior to 1900 (Zone I), the total abundance of planktonic *Bosmina* was high. In the c. 1900s-1920s (Zone II), the relative abundance of *Bosmina* began to decline, while the abundance of littoral species increased. The dominant species during this time were *Acroperus harpae*, *Alona rectangula*, *Camptocercus rectirostris* and *Dunhevedia crassa* (Fig. 6). During the c. 1930s-1950s (Zone III), the relative abundance of *Bosmina* was relatively constant, but the abundance of littoral species continued to increase. Four dominant species were found in this community; *Alona rectangula*, *Chydrorus sphaericus*, *Dunhevedia crassa* and *Graptoleberis testudinaria*.

During the c. 1960s-2000s, the period of major dam construction in the Yangtze, the total abundance of *Bosmina* increased, particularly in the early 2000s, and four species of littoral species, *Alona guttata*, *Alona intermedia*, *Chydorus sphaericus* and *Sida crystallina* also became dominant throughout this period (Fig. 6).

6 Discussion

6.1 Shifts in hydrology and its implications for ecosystem functioning of wetlands within the Murray and Yangtze River wetlands

Over the past century, impacts on the Murray and Yangtze Rivers include the construction of irrigation dams, hydroelectric power plants, regulation works for navigation, land reclamation projects, and large-scale flood control measures (Maheshwari et al., 1995; Sun et al., 2012). As a result, vast areas of floodplain wetlands of both river systems have been drained and disconnected from the river. In some areas, this reduced hydrological connectivity has resulted in a flushing of organic matter and nutrients from the floodplains only during extreme floods, when the river retention capacity is the lowest. Therefore, organic matter from the floodplain system is not accessible to wetland organisms. With the loss of dynamically connected floodplains, the biogeochemical budget of the Murray and Yangtze River wetlands has changed significantly. Previous evidence strongly suggests that the climatic cycles of drought and flood have become extreme, triggering unusual responses of floodplain wetlands to the disturbance regime of these rivers (Zhang et al., 2012).

Wetlands losing hydrological connections with the river result in divergence of aquatic micro- and macro-invertebrate assemblages (Qin et al., 2009). The disruptions in the natural variability and connectivity of hydrological regimes, due to river-flow regulation, have consequently reduced ecological integrity, resulting in reduced invertebrate diversity (Sheldon et al., 2002). The downstream impacts of low flows in the River Murray were visible mainly following the construction of Hume Dam in 1936, but at present, average monthly and annual flows are still considerably lower than those of natural conditions in the past (Maheshwari et al., 1995). The study of natural flow regimes in the Murray River suggests that the strength of average annual floods (annual exceedance probability 50%) has reduced by over 50% at all stations. The effects of large floods with an average recurrence interval of 20 years or more, are, however, relatively low (Maheshwari et al., 1995). The number of low flows defined by a given annual non-exceedance probability, are higher under regulated conditions than under natural conditions (Maheshwari et al., 1995). The implications of these changes are not only for communities of native plants and animals in both riverine and floodplain environments, but also for the long-term use of the riverine resources by humans (Maheshwari et al., 1995). Rivers and their associated wetlands exchange particulate and dissolved organic matter, including suspended sediments, nutrients, and algal biomasses (Tockner et al., 1999). These nutrients are fundamental for the support of ecosystem structure and function in riverine food webs (Bunn and Arthington, 2002). The current flow regimes also determine which physical habitats are available for all aquatic species that have evolved life history strategies primarily in direct response to natural flow regimes (Bedford, 1996).

Permanent inundation of wetlands occurred in many areas across the Lower Murray River in response to the 1914 Commonwealth Act. This legislation enforced a requirement to manage the Murray River's water by the construction of locks, weirs, and water storage areas. Construction of the Mildura Weir (Lock 11), which began in August 1923, resulted in an increased water level in Kings Billabong by the time construction was completed in 1927. These long periods of water storage in Kings Billabong are thought to have increased stagnation, nutrient levels, and primary productivity, subsequently impacting the higher trophic levels around the billabong (Kattel et al., 2015). Some have argued that the high nutrient input in the river system, combined with relatively long water residence times in water storages, supports phytoplankton growth and a tendency towards eutrophication (e.g. Tockner et al., 1999; Chaparro et al., 2015).

In the Yangtze River, construction of many dams and water impoundments has significantly altered downstream hydrological regimes, which have directly affected the relationship between the Yangtze River and its river channels and floodplain wetlands, including the Zhangdu Lake (e.g. Yang et al., 2011a, b). The construction of dams throughout this catchment has caused changes in channel morphology and sedimentology, with a concomitant drastic decline in sediment transportation and severe channel erosion in connections to lakes. From the monitoring of stream cross-sections, changes to river channels are evident, including the reduction of water level within wetlands (Yang et al., 2011a, b). These have inevitably induced alterations in inundation patterns of the wetlands, resulting in changes to ecosystem structure and function, which in turn have disturbed the habitats of biota (Maheshwari et al., 1995; Sun et al., 2012). As a consequence of a rapid expansion of human activity in the watershed during the 1960s, significant changes at the base of the food web in Zhangdu Lake have been observed in the subfossil composition of testate amoeba communities. For instance, the characteristic oligotrophic, lake-dwelling species (e.g. *Difflugia biwae*) have been replaced by eutrophic species (e.g. *Difflugia oblonga*) (Qin et al., 2009).

6.2 Cladoceran-inferred responses to hydrological shifts in Murray and Yangtze River wetlands

Cladoceran assemblages of three floodplain wetlands, Kings Billabong, Zhangdu Lake, and Liangzi Lake all have shown strong responses to human-mediated hydrological alterations in the Murray and Yangtze Rivers over the past century. Although the N2 diversity index did not show a strong response to disturbance, the impact of river regulation and permanent inundation of Kings Billabong in the 1920s nonetheless revealed a decline in the density of littoral species.

The Hill's N2 diversity index assumes that the number of species in an ecosystem is uniformly distributed (Hill, 1973). Following this advice, we assumed that the distribution of cladoceran species along the temporal scale of Murray and Yangtze River wetlands should also have been uniform. However, the N2 diversity index of cladocerans in Kings Billabong and Yangtze River wetlands was found to be non-uniform across our measurement period, and, in addition, they showed different trends. Following similar regulation and construction of dams in the two sites, the N2 diversity index decreased in Kings Billabong, whereas the N2 index in Yangtze River wetlands increased. We argue that the observed disturbances in each site were due to quite different impacts of regulation. In Kings Billabong, the disturbance appeared to be severe following the arrival of Europeans, whereas the disturbance in Yangtze River wetlands occurred over a longer time scale, and could be characterised as an intermediate frequencies of disturbance (Collins and Glenn, 1997). Indeed, records indicate that the early European immigrants in Australia transformed the landscapes quickly, which had severe impacts on Kings Billabong cladocerans. However, unlike Kings Billabong, the Yangtze River wetlands did not experience such a severe disturbance, and as the intermediate disturbance hypothesis model suggests, the diversity index increased following the disturbance (Townsend and Scarsbrook, 1997) indicating the intermediate frequencies of disturbance in cladoceran diversity of the Zhangdu and Liangzi lakes.

However, habitat stability determines the species and functional diversities of biota. In addition, the species diversity patterns are often context and system dependent (Biswas and Malik, 2010). For example, reduced water level, which results in increased light regime and higher growth of littoral vegetation, may provide stability of habitat for small *Alona* sp. in Yangtze River wetlands following the intermediate disturbance (c. 1960s), and consequently this leads to an increased N2 diversity index (Figs. 4 & 5).

The species such as *Dunhevedia crassa* and *Graptoleberis testudinaria*, are adapted to submerged vegetation and their decline in abundance indicates a reduction of suitable habitat, such as decreased water quality. The increase in the abundance of lentic species, such as *Bosmina meridionalis,* demonstrates a switch from the prior ephemeral state to one of more or less constant inundation. Although drought had little or no impact on the water nutrient levels in Kings Billabong following regulation, by contrast, large-scale flood events such as in 1956, may have significantly increased nutrient input in the water column. The apparent result was to increase the population of *Bosmina*, as well as littoral species (e.g. *A. guttata*) that prefer enriched nutrient environments (Hofmann, 1996). Turbidity from suspended sediment during flood events also limits growth of submerged vegetation, due to a reduction of light penetration. By the early 2000s, planktonic *B. meridionalis* and littoral *A. guttata* and *Biapertura longispina* were the dominant species. The high density of cladoceran ephippia retrieved from the wetland sediment also indicates "stress" among the cladoceran community during the prevailing conditions of post- regulation period in the Murray River system (Nevalainen et al., 2011). The low abundance of *D. crassa* following river regulation reflects the impact of river regulation on the aquatic ecosystem, with degraded water quality and reduced resilience in the wetland community. In shallow lakes, a consequence of human-induced actions is the tendency towards a regime shift, followed by poor ecological resilience

(Folke et al. 2004). The loss of functional group species and consequent reduced species diversity may lead to a loss of whole trophic levels or ‘top-down effects’ (Folke et al., 2004).

The Zhangdu Lake aquatic community responded to downstream water shortages in the river channel connecting to the lake, as revealed by low lake levels following the construction of dams and reservoirs for water conservation in the 1950s-1970s. Subsequent to river regulation during the 1950s, hydrological alterations of the river channel and changes to the water level of Zhangdu Lake, increased the growth of littoral plants. This also resulted in increased abundance of littoral cladoceran species, such as *Acroperus harpae*, *Alona guttata*, *Alona rectangula*, *Chydorus sphaericus*, *Graptoleberis testudinaria* and *Sida crystallina* (Fig. 5). Although the abundance of littoral species in the lake indicated increased growth of submerged vegetation, the condition of the wetland ecosystem following regulation was poor. The clear water regime, present prior to regulation, gradually transformed to a eutrophic state following the construction of dams. Many small cladocerans recorded in Zhangdu Lake following the work of the 1950s, are typically associated with still (lotic) water, eutrophic and poor water quality conditions, and have been found in similar disturbed habitats elsewhere. For example, in Europe, cladoceran species such as *A. harpae*, *C. sphaericus* and *S. crystallina* have a characteristic affiliation with lotic environments (Nevalainen, 2011). In addition, in Tibet, *Chydorus sphaericus* has been found to be adapted to wide range of environmental gradients, while *Alona affinis* and *Acroperus harpae* colonize dense aquatic macrophytes, and *Graptolebris testudinaria* and *Eurycercus lamellatus* are adapted to shallow littoral environments, with a preference for debris-rich substrates (Liping et al., 2005).

Eutrophication in Zhangdu Lake, due to hydrological changes of the wetland, was also indicated by the presence of testate amoeba (Qin et al., 2009). Our results strongly suggest that hydrological alterations of rivers and wetlands can result in eutrophication and lead to an increased abundance of smaller size littoral cladocerans. The low level of floods could reduce water level, increase telematic plant growth, and decrease the redox condition of the wetland resulting in the variation in growth, metabolism and reproduction of such cladocerans (Pawlowski et al., 2015). The shallow littoral environment provides habitats for different fish species, and may increase the predator-prey interactions (Pawlowski et al., 2015). Following regulation, the large number of cladocearn ephippia recorded in the sediment in Zhangdu Lake (which is found in the lower Yangtze), also indicates the decline in lake levels and the loss of lentic habitats, which leads to reduced feeding habitats and reproductive output or an increased ecological stress among the cladoceran community, particularly during the c. 1990s-2000s. In Europe, increases in sedimentary resting eggs of cladocerans are reported to be associated with major environmental transitions; for example, climate change (e.g. Pleistocene-early Holocene), timing of strong predator-prey interactions (e.g. fish predation pressure), and increased human impact in the catchment (e.g. unprecedented release of chemicals) (e.g. Sarmaja-Korjonen, 2003; Nevalainen et al., 2011).

The response of the subfossil assemblage of cladocerans in Liangzi Lake to hydrological change in the Yangtze River during the 1950s was difficult to establish. This could be due to the permanent inflow to this lake from the Yangtze River. The higher abundance of *Bosmina* prior to 1900s indicate that the lake was kept at a certain water level, and much of the trophic materials contained in the surface water met the demands of planktonic cladocerans (e.g. Liping et al., 2005). However, the abundance of littoral species *Alona rectangula*, *Chydrorus sphaericus*, *Dunhevedia crassa* and *Graptoleberis testudinaria* during the 1950s are indicative of decreasing depth. During the 1990s to the 2000s, Liangzi Lake was impacted by intensive agriculture practices in the catchment and nutrient inputs into the wetland, as indicated by an increased abundance of planktonic *Bosmina* (Lipping et al., 2005). In 1992, the local government restricted aquaculture to the western part of the

Liangzi Lake, since this activity was affecting water quality throughout the entire lake (Xie et al., 2001). This problem had been detected from ecological stress responses of cladocerans, as revealed by an increased density of resting eggs in the sediment, as well as an increased abundance of *Bosmina* and the chydorid species such as *Alona guttata*, *Alona intermedia*, *Chydorus sphaericus*, since these are all found in nutrient-rich environments (e.g. Sarmaja-Korjonen, 2003; Nevalainen et al., 2011).

All three of these wetlands appear to exhibit characteristic traits of hydrologically triggered ecosystem changes, as revealed by subfossil cladoceran assemblages, since each has tended to undergo regime shifts during recent decades. Furthermore, species richness in each is indicative of reduced water quality. Hydrology strongly drives the community composition of phyto- and zooplankton, relevant nutritional resources, and habitat characteristics, mainly via input of N and P from the eutrophic main channels during flood events (Van den et al., 1994; Nevalainen, 2011). The phenomena observed in the dynamics of physical and biological assemblages, and the diversity of cladoceran zooplankton, in Kings Billabong and Zhangdu Lake, for example, have shown tendency of existing in alternative stable states resulting from switching of ecosystems, irrespective of inundation (Kings Billabong) or dehydration (Zhangdu Lake).

The alternative 'stable states phenomena' in shallow lakes and wetlands have been widely viewed as indicative of changes to resilience of ecosystems (Scheffer and Jeppesen, 2007). Such phenomena have shown the condition of wetlands to vary from a relatively good water quality, vegetation-rich state to a poor, turbid water state, which is usually less desirable to society (Folke et al., 2004). Positive feedback associated with the condition of increased water quality, species richness and population dynamics of *D. crassa* in Kings Billabong prior to 1900 is characteristic of a resilient ecosystem (e.g. Suding et al., 2004). By contrast, an open water habitat, which may be characteristic of a longer flood duration following regulation, leads to negative feedback, which is turbid and less resilient (e.g. Suding et al., 2004). Similarly, in Zhangdu and Liangzi Lakes, an increased abundance of smaller, mud-dwelling cladoceran species such as small *Alona* sp. and *Leydigia leydigi*, as well as presence of other meso-eutrophic species, *Chydorus* and *Bosmina* following regulation, is indicative of increased eutrophication (Hofmann, 1996) caused by alteration of flow regime and dehydration of wetlands.

Long term persistent human disturbances alter species diversity and have functional consequences in ecosystem processes (MacDougall et al., 2013), which may be observed via impact on ecological traits (Chapin III, 2000). The components of species diversity expressing certain traits include the number of species present (species richness), their relative abundances (species evenness), the particular species present (species composition), the interactions among species (non-additive effects), and the temporal and spatial variation in these properties. The consequence to the environment as a result of cladoceran diversity change in the Murray and Yangtze River wetlands is difficult to predict, but in the longer term, poor functioning of the ecosystem due to reduction in diversity in Kings Billabong is expected. In the Yangtze River wetlands, the dominant species richness trait, for instance abundance of the small *Alona* sp. Group, can also lead to poor ecosystem functioning (e.g. Chapin III, 2000). This evidence strongly reflects the reduction in resilience and the limited capacity of these wetlands to support ecosystem services for the society in these increasingly regulated river basins. Further decline in eco-hydrological conditions including the water quality, water quantity, fishery resources, and recreational amenities, due to cumulative stressors can lead to the collapse of ecosystem services, in which case society will no longer be benefitted (Falkenmark, 2003).

The ecosystems of both Murray and Yangtze rivers are affected by a range of drivers. The cumulative stressors upon these wetlands are nutrient enrichments from agricultural catchments, heavy metal release from industries (mainly in Yangtze wetlands) and climate change (flooding and drought episodes). Increased nitrogen deposition has been reported to have a great effect on diversity and ecosystem functioning of wetlands, leading to collapse of food chain and ecosystems (Hooper et al., 2012). This collapse may lead to crises to higher trophic levels including the humans, with conflicting demands placed on natural resources and increasingly poor public health of the local community (Kattel et al., 2013). The participatory approach of river basin management can help increase resilience of wetland ecosystems and goods and services to society (Vörösmarty et al., 2010). Joint action by various stakeholders including ecologists, resource managers and decision makers can be useful to achieve management goals for natural resources (Biswas, 2004; Carpenter et al., 2009, Liu et al., 2014). Such an adaptive management approach for water resources is increasingly appropriate for maintaining ecosystem services of large river basins (e.g. Richter et al., 2003).

6.3 Development of an adaptive water resource management framework for Murray and Yangtze River wetlands

Water problems in large river basins are increasingly interconnected with multi-sector developments such as agriculture, energy, industry, transportation and communication. Several authors (Walker et al., 1995; Kingsford et al., 2000; Fu et al., 2003) suggest that maintaining ecosystem health of wetlands associated with large river basins, requires a new paradigm in water management. Today, the wetlands of both the Murray and Yangtze River basins have faced greater challenges from hydrological modification, water shortage and eutrophication than at any time before (Yang et al., 2006; Shen, 2010; Gell and Reid, 2014). There are growing concerns about the uncertainties of climate change and socio-economic impacts on these river basins (Palmer et al., 2000). For example, due to rapid decline in water quality, biodiversity and ecological characters of the lower Yangtze River, this region has already been declared as the ecosystem of "lost resilience" (Zhang et al., 2015). A comprehensive synthesis by Varis and Vakkilainen (2001) suggests that following the 1970s, China's environmental pressures have surpassed the carrying capacity of the ecosystem, resulting in greater challenges for water resource management in the Yangtze and many other river basins. Similarly, a rapidly declining trend of biological diversity and ecosystem states of the Murray River basin has also been widely reported following the 1950s (Kingsford et al., 2000). For example, more than 80% of wetlands in the Lower Murray River reaches (Australia) have undergone a significant decline in flow regimes and ecosystem health, due to rapid rates of sedimentation, turbidity and loss of macrophytes (e.g. Mosley et al., 2012; Gell and Reid, 2014). Additionally, the wetlands of both large river basins have experienced substantial loss of ecosystem services, and increased river regulation during the 20th century. With increasing demand for water, food, fibre, minerals, and energy in the 21st century, these pressures have degraded conditions of these natural resources even further (e.g. Davis et al., 2015). Solutions for water issues are not possible without a joint effort by the various stakeholders involved in understanding the complexity of water management in large river basins (e.g. Biswas, 2004). It has been envisaged that the current management framework needs to be revitalized to resolve growing issues of wetland management and maintenance of associated ecosystem services, including the quantity and quality of water in both river basins.

Adoption of an Integrated Water Resource Management (IWRM) framework has been increasingly useful to resolve issues of quantity and quality of water worldwide. The IWRM promotes water management by maximizing relevant economic and social welfare in an equitable manner without compromising the sustainability of vital ecosystems (Biswas, 2004). Over the past decades, the IWRM approach has been constantly modified as per the societal needs of local water management. On this basis, we have proposed the development of an adaptive water resource management framework for wetlands of these two large, hydrologically-transformed river basins in Australia and China (Fig. 7). This consideration has been taken into account on the basis of eco-hydrological evolution of wetlands inferred by subfossil cladoceran assemblages and diversity (Figs. 4 & 5). These changes have been profoundly implicated by socio-economic developments in both river basins over the past century. The proposed adaptive water resource management framework (Fig. 7) is integrated and multi-disciplinary in nature. It is intended to improve management and to accommodate change by learning from the outcomes of management (restoration) policies and practices, as described by Holling, (1978) initially, and debated extensively by Jakeman and Letcher, (2003), Macleod et al. (2007) and Pahl-Wostl (2007). Such a management framework has been facilitated by dialogue between scientists, stakeholders and policy makers, and can be expected to result in highly positive outcomes in management (Falkenmark, 2004).

In the framework (Fig. 7), we consider that both the quantity and quality of water determines the resilience of the wetland ecosystems of the Murray and Yangtze Rivers. Prior to regulation, these wetlands were maintained by sustainable flow regimes with improved water quality and reasonably good ecological health at baseline conditions. The natural flood inundations maintained the amount of water, nutrients, carbon and salts in wetlands supporting biological diversity, ecosystem functioning and associated goods and services (Junk et al. 1989, Thorp and Delong 1994; Humphries et al. 1999; King et al. 2003). This evidence is also supported by various eco-hydrological models being developed and tested previously to measure flow regimes and ecosystems of the large river wetlands worldwide (Vannote et al. 1980; Naiman et al. 1987; Thoms and Sheldon, 2000).

The use of palaeoecological approach in our study provides the 1950s as a benchmark of change in flow regime and ecosystem of the Murray and Yangtze River wetlands (Fig. 7). Following river regulation (post 1950s), both the quantity and quality of water in the Murray and Yangtze river wetlands had been significantly altered, reaching a critically low level of flow and ecosystem health by the 2000s (Fig. 7). The condition of and changes in flow regime in the Murray River basin was reported by Maheshwari et al. (1995), where the average monthly and annual flows were considerably lower than those of natural conditions prior to regulation. We argue that the 2000s was the critical level of threshold for quality and quantity of water in wetlands of both river basins, and all available restoration measures should be adopted to avoid further decline in conditions in these wetlands.

In our adaptive water resource management framework (Fig. 7), we have proposed the role of three pillars: science, engineering and community engagement when restoring the degraded wetlands of these two large river basins of Australia and China. River regulation, including widespread infrastructure developments across the river basins, has consistently modified natural hydraulic residence time, leading to changes in diversity and associated ecosystem structure and function of wetlands. For example, construction of Hume Dam in the 1930s in Murray River, and several large dams, including the Three Gorges Dam (TGD) since the 1950s in Yangtze River, will have long-lasting effects on downstream flow regimes, as well as wetland ecosystem structure and function (Pittock and Finlayson, 2011; Wu et al., 2003). Whilst these infrastructures are already in place, strong scientific evidence including the understanding of the alteration of historical ecology and hydrology is potentially powerful tool to unravel the benchmark of the eco-hydrologic conditions of the Murray and Yangtze River wetlands over time. For example, the use of stable isotopes of carbon in subfossil cladocerans and chironomids in Kings Billabong indicated the shift in carbon energy source following the river regulation (Kattel et al., 2015). As this evidence is significant for understanding wetland ecology, the assessment of past moisture regimes based on various stable isotopes (e.g. oxygen) in water and organisms would be increasingly crucial to identify the source of water for wetlands and the condition of critical water shortages. Benchmarks are important for the development of predictive models on wetland restoration programs by understanding the change of quantity and quality of water over time. Such predictive models can also identify early warning signals of regime shift in wetlands (Wang et al., 2012). Resource managers can target restoration measures on the basis of benchmark conditions so that the investment will not be wasted on restoration of wetlands that would not result in improved values. Zweig and Kitchens (2009) suggest that the predictive hydrologic models can be the foundation for restoration programs of degraded wetlands, since these models can successfully identify the hydrologic effects on the state of transitioning ecosystems.

Secondly, innovative and environmentally-friendly infrastructure development and operation have been proposed in water restoration programs, and there is an increased demand of efficient infrastructure development for the wetlands of Murray and Yangtze River basins (e.g. Fu et al., 2010). One of fundamental issues of the integrated water resource management program is to meet balanced water allocations between industry and environment (Poff et al., 2003; Biswas, 2004). Due to the overwhelming industrial demand for water in recent decades, economists have developed efficient environmental water allocation schemes for various river basins including the Murray and Yangtze (e.g. Lee and Ancev, 2009; Jiang, 2009). The proposed adaptive water resource management framework (Fig. 7) highlights the role of institutional capacities and development of efficient water allocation infrastructures (e.g. Yu et al., 2009). Consideration of efficient infrastructures for consumptive water uses and environmental water allocation for ecosystem function of large river basins is crucial for wetland restoration measures and sustainability of ecosystem services (e.g. Grafton et al., 2013).

Finally, the need for strong linkages between scientific community and management stakeholders is essential in order to achieve the goal of wetland ecosystem management and restoration (e.g. Pittock and Finlayson, 2011, Liu et al., 2014). Any decision making should be based on the need of the local community and mutual understanding among scientists, resource managers and community leaders (Poff et al., 2003). The successful outcomes of water resource management in river basins would be possible if the community is engaged with all aspects of environmental hydrology, ecology and water resource management programs including both structural (e.g. hydropower dams) and non-structural infrastructure developments (e.g. awareness in adaptation to change), as well as water saving (e.g. Shen, 2010). The proposed adaptive water resource management framework (Fig. 7) is expected to enhance wetland resilience by improving both water quality and quantity, including ecosystem function, consequently assisting the basin-wide management of food and water security issues through extensive community participation. For example, the WWF-supported partnership program, together with government agencies and local communities, was highly successful for improving water resources, both quantitatively and qualitatively, in the Yangtze River Basin. Under this type of management program and in partnership with local people, the three Yangtze lakes (Zhangdu, Hong and Tian-e-zhou), which were disconnected from the main channel during the 1950s-1970s, have now been recharged by opening of sluice gates (Yu et al., 2009). The recharging of Zhangdu Lake has not only enhanced resilience of the lake environment to climate change and but also livelihoods of the local people (Yu et al., 2009). Recently, the role of community participation in water resource management has also been reported significant in some wetlands of the Murray Darling Basin. For example, the living Murray project initiated by the Murray Darling Basin Authority with the view of increased indigenous community engagement has led to improvements in the ecological health of the Barmah–Millewa floodplain wetlands supporting large bird breeding events (MDBA, 2014). This kind of success has also been revealed by the coupled socio-hydrologic models showing strong association between the trajectory of human-water co-evolution and associated goods and services in the Murrumbidgee River basin (one of sub-basins of the River Murray) (Kandasamy et al., 2014).

7 Conclusions

Evidence from subfossil assemblages of cladocerans over the past few decades from all three wetlands, Kings Billabong, Zhangdu Lake and Liangzi Lake, suggest that river regulation by humans in the Murray (Australia) and Yangtze (China) rivers have significantly altered natural flows, including the hydrology and ecology of these wetlands. The response of subfossil cladoceran assemblages was evident via both prolonged flooding (inundation) and dehydration (abstraction) of water in the Murray and Yangtze Rivers, respectively. Other factors, such as land use, socio-economic developments, and rapid climate change, particularly over the past 30-40 years, may have exacerbated the hydrological and ecological processes further. The conditions of wetlands following the large-scale disturbances, such as widespread river regulation, and construction of dams and reservoirs, have shown a tendency to trigger wetland ecosystem switch, and highlights the urgent need for restoration measures to improve ecosystem services, through better management of quantity and quality of water. The proposed adaptive water resource management framework, based on science, engineering, and community participation, is expected to enhance resilience of the Murray and Yangtze River wetlands and help manage the basin-wide water and food security issues.

Acknowledgements

This project was supported by a number of grants awarded to GK including the Australian Institute of Nuclear Science and Engineering (AINSE) #AINSEGRA11087, Australia-China Science and Research Fund (ACSRF), Chinese Academy of Sciences (CAS) and National Science Foundation of China (Grant No. 41472314, 41102105). Australian Nuclear Science and Technology Organisation (ANSTO); Collaborative Research Network (CRN), and the Faculty of Science and Technology of Federation University Australia (FedUni); State Key Laboratory of the Nanjing Institute of Geography and Limnology Chinese Academy of Sciences (NIGLAS) assisted for collection of samples from the field and analyses at the respective laboratories. This paper was presented in Australia-China Wetland Network Research Partnership Symposium (March 24. 2014), Nanjing, China. I would like to thank the HESS Editor, Giuliano Di Baldassarre, and the other two anonymous reviewers for making a critical review of the manuscript and Jim Sillitoe and Sandra Weller for editorial supports.

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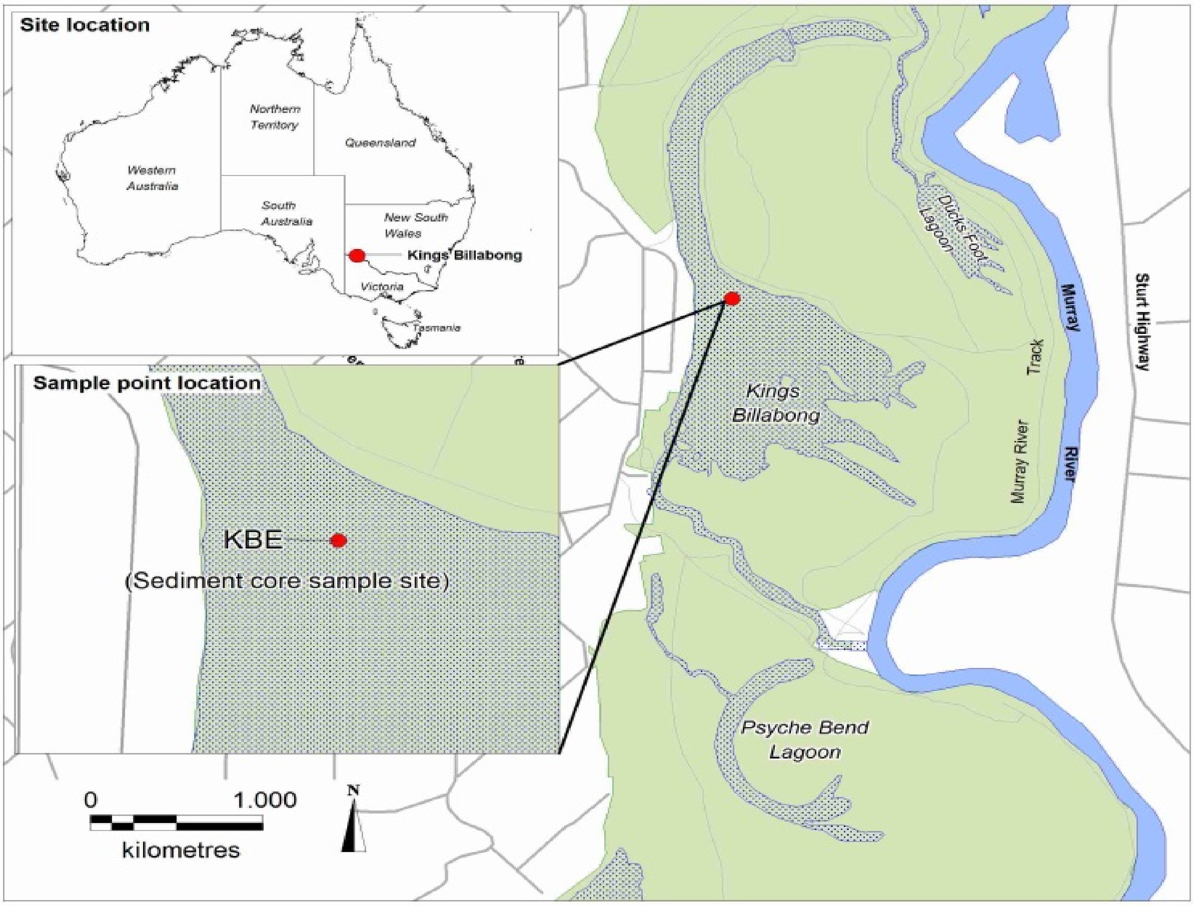

Figure 1. Kings Billabong, one of the wetland complexes of the River Murray system in Southeast Australia. KBE was the deepest point of the lake, where a sediment core for this study was taken.

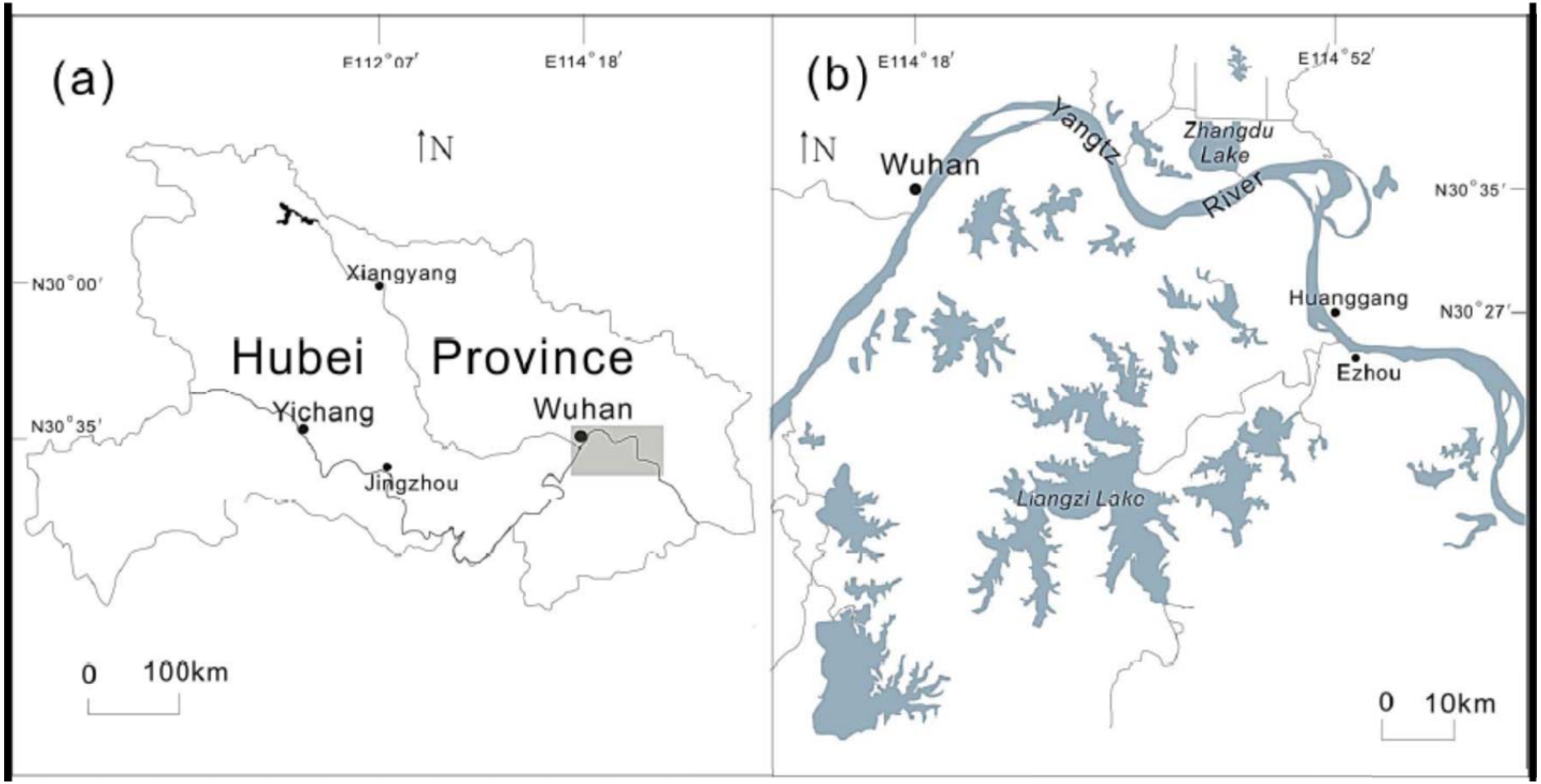

Figure 2. Zhangdu Lake and Liangzi Lake around the middle reaches of the Yangtze River in Hubei Province of China.

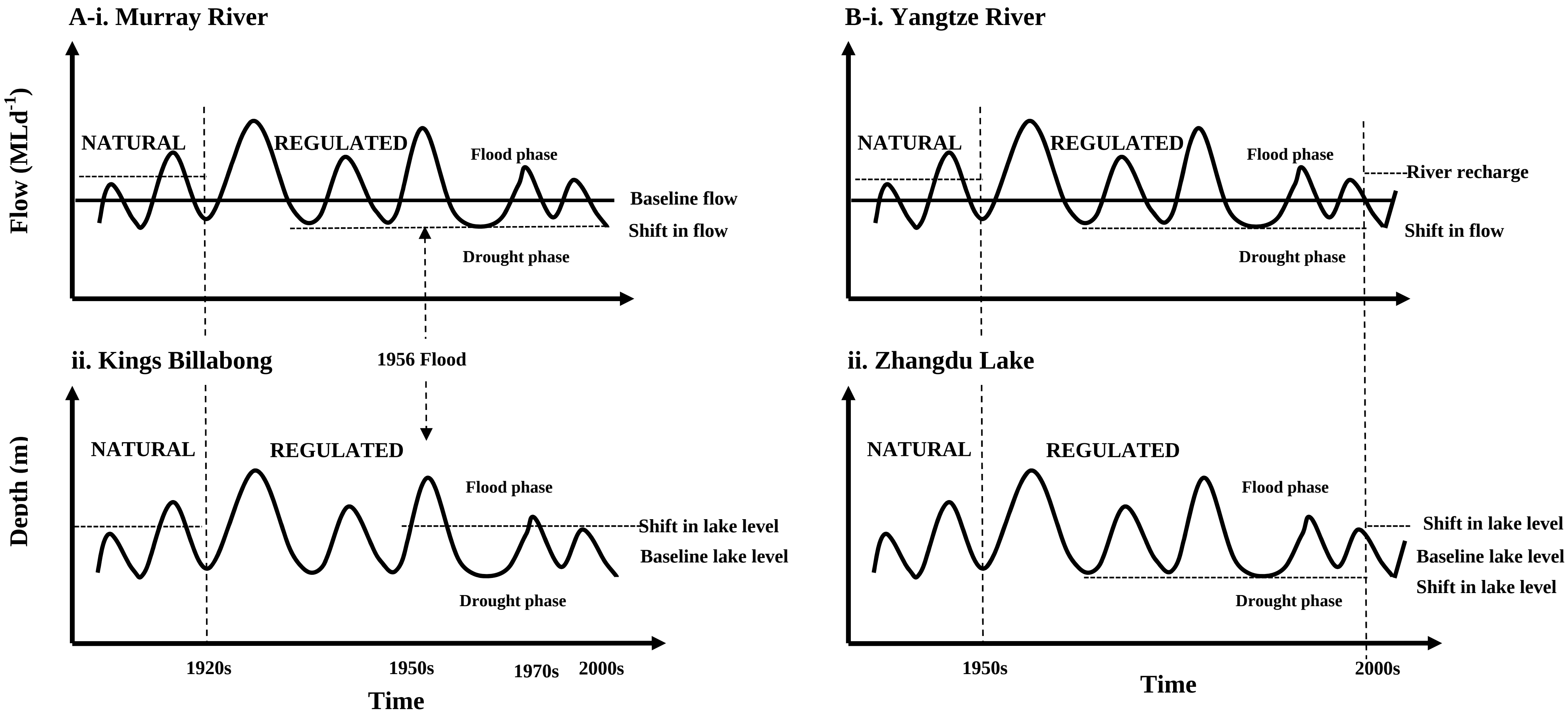

Figure 3. Hydrological frameworks of Murray and Yangtze rivers. A. i & ii. River Murray: regulation was imposed by humans in the 1920s AD, which resulted in low water volume in the down-stream river channels, but Kings Billabong's conversion to a water storage tank permanently led higher lake level, subsequently ceased natural dry-wet cycles; B. i & ii. Yangtze River: the first large scale human impact on the river was imposed during the c. 1950s, which ceased naturally occurring flood pulses in adjacent wetlands leading to a drying up of the river channel connecting to wetlands including low water volume in Zhangdu Lake.

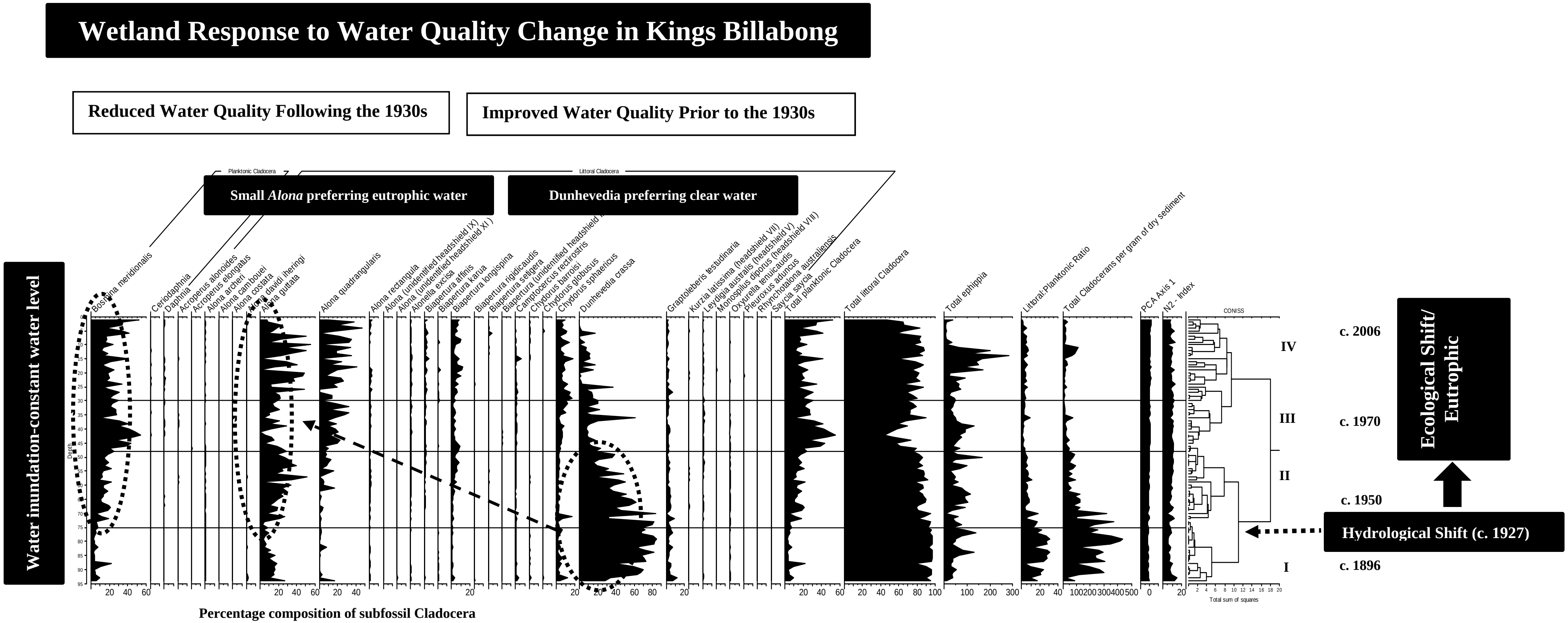

Figure 4. Percentage composition and N2 diversity index of subfossil cladocedans in Kings Billabong, their response to past hydrological and water quality change.

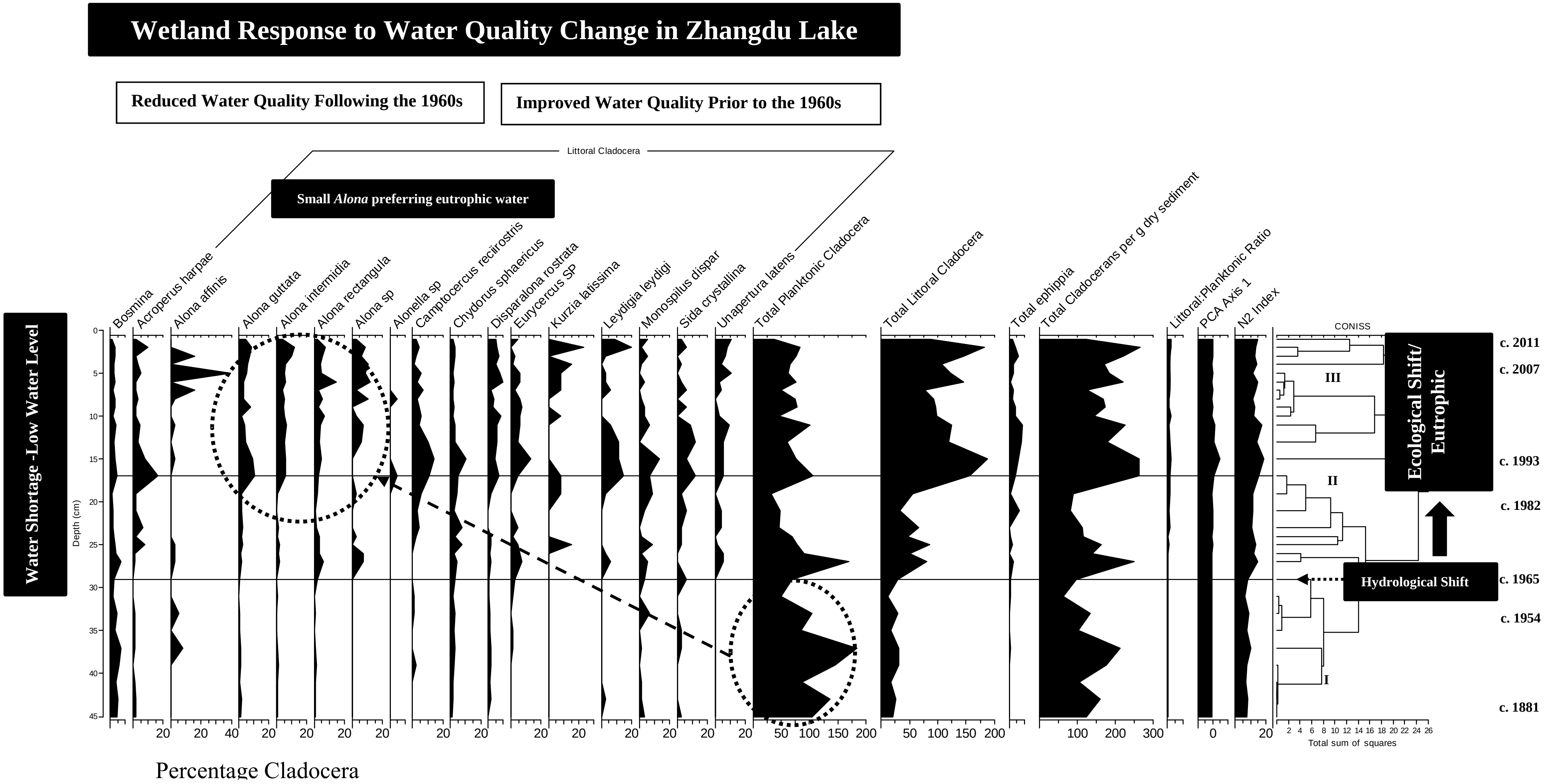

Figure 5. Composition (%) and N2 index of subfossil cladocedans in Zhangdu Lake, and their response to past hydrological and water quality change.

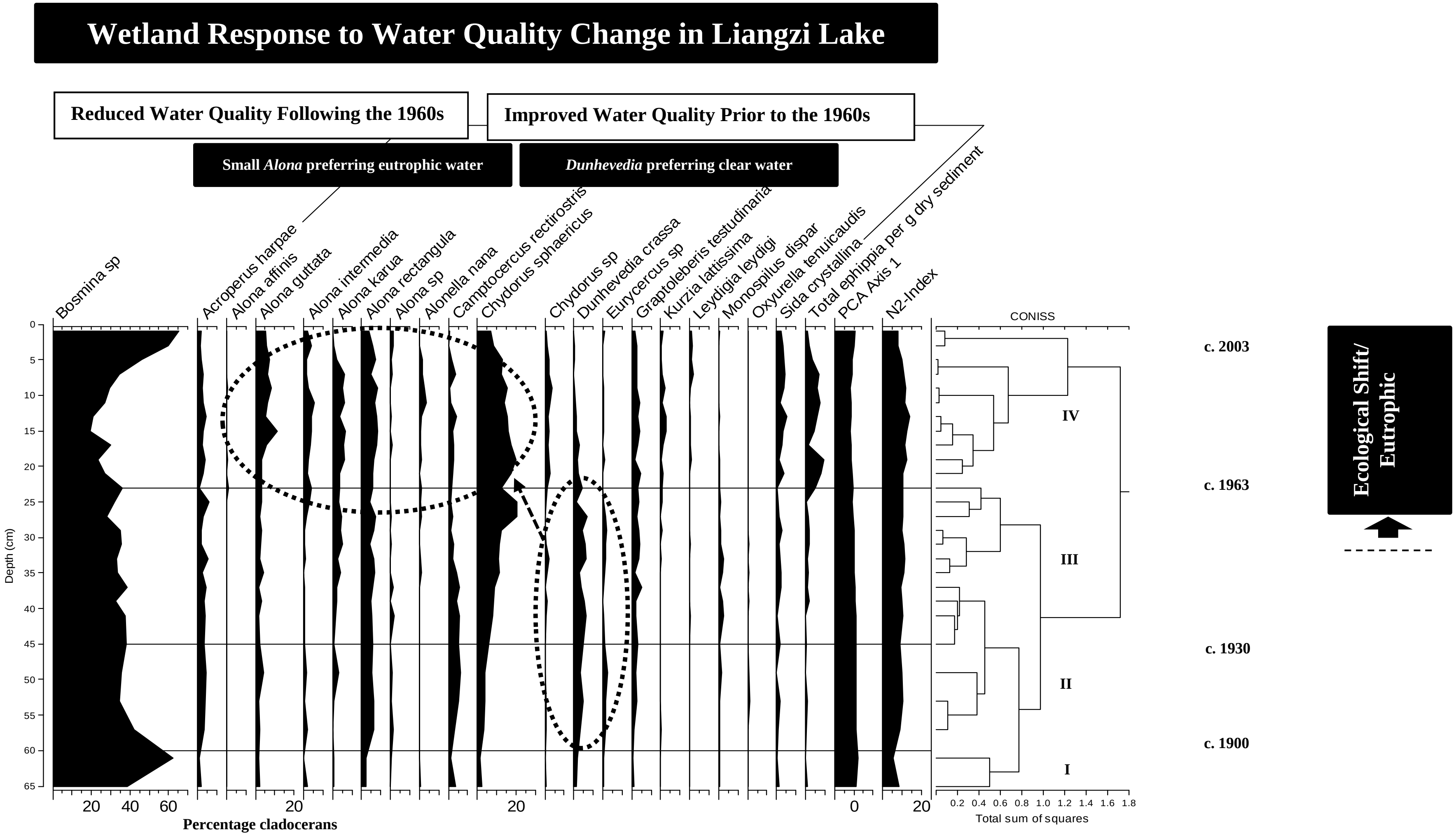

Figure 6. Composition (%) and N2 diversity index of subfossil cladocedans in Liangzi Lake, and their response to past water quality change.

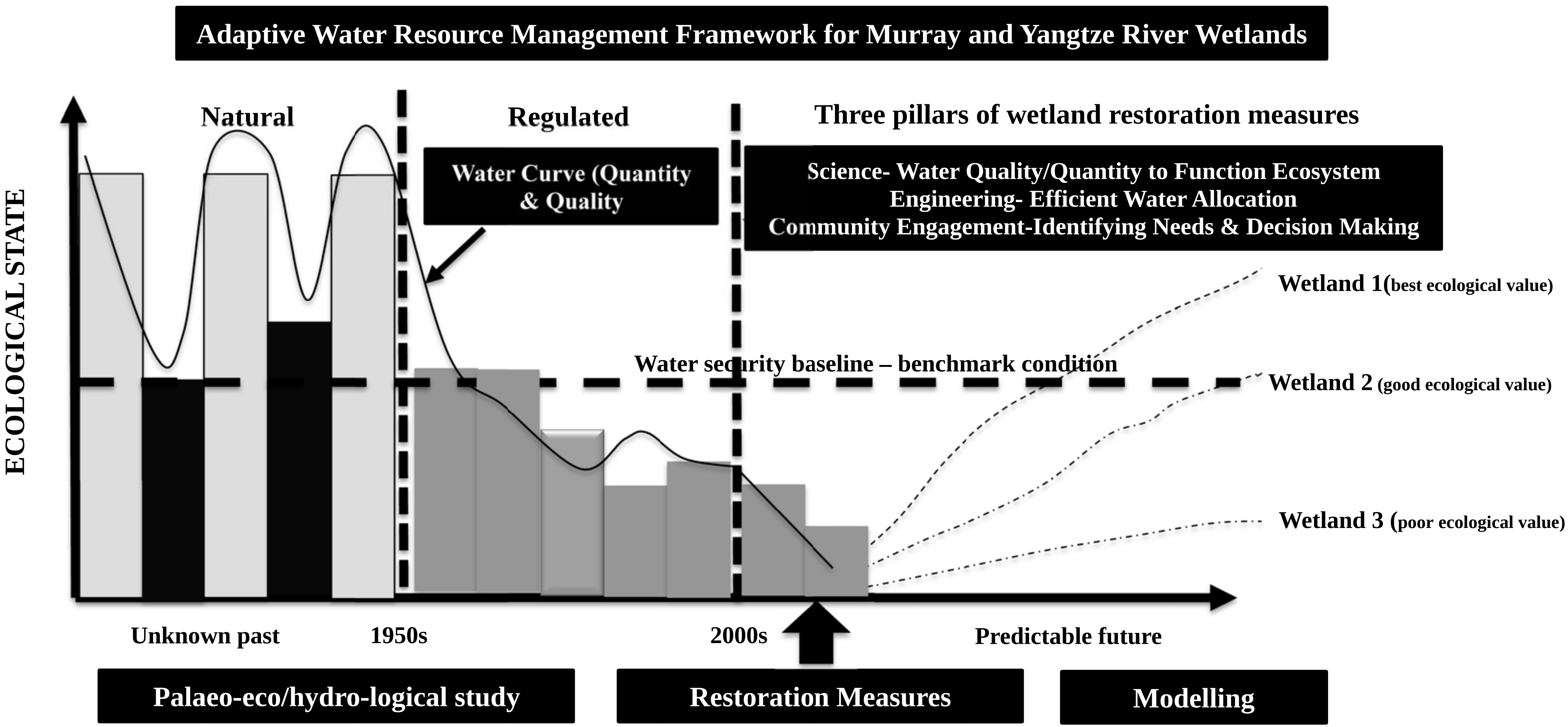

Figure 7. An adaptive water resource management framework based on palaeoecological study in Murray and Yangtze River wetlands: Prior to river regulation (c. 1930-50s), the quantity of water and wetland ecosystems was determined by natural flood pulses, when the water security curve was above the baseline and the state of ecosystem was natural. However, following the1950s, ecosystem responded to human impacts on both river systems showing a rapid downward movement of the water curve. By the early 2000s, the natural flood pulses reduced followed by deterioration of the condition of wetlands. The ecosystem structure and function were poor due to poor water quality and quantity and limited submerged vegetation. The restoration measures are proposed to bring the water quality and quantity back to the baseline condition by a joint effort from science, engineering and community participation. Scientific knowledge is enhanced by palaeoecological and hydrological monitoring and development of future prediction models in wetland ecosystem. However, not all wetlands can be restored to a baseline condition given their individual variability (detail is described in the text).