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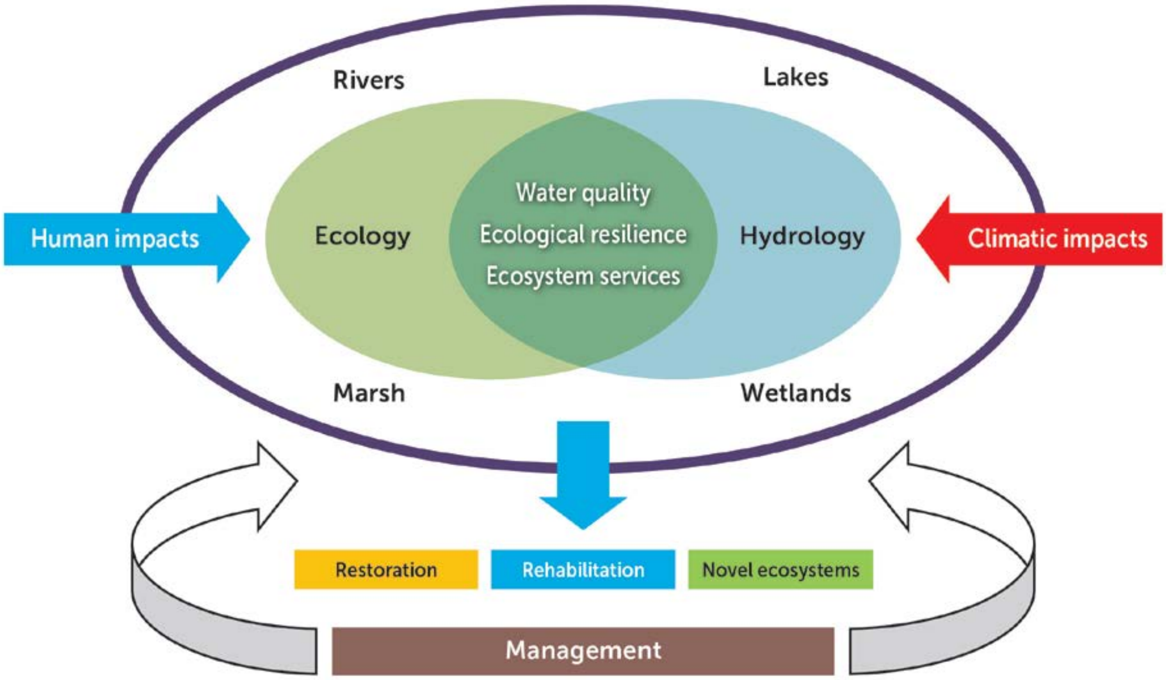

Conceptual framework of freshwater ecosystem resilience in the context of climate change and human impacts

The research program has an overlapping theme between ecology and hydrology of large river and wetland systems which are impacted jointly by human and climate change. The research focus of the symposium was to understand these issues and develop strategies for management and restoration of wetlands for promoting ecological resilience.

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Cover photos

Front cover: Barmah Lake, Victoria, Australia
Back cover left: Gunbower Island, Victoria, Australia
Back cover centre: Zhongmiao Temple, Chaohu Lake Anhui Province, China
Back cover right: Kings Billabong, Murray River, Victoria, Australia

All photos taken by Dr Giri Kattel

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Contents

Preface

This publication is a compilation of short papers presented at the Australia-China Wetland Network Research Partnership Symposium, held in China at the Nanjing International Conference Hotel, 24 March 2014. The symposium, jointly organised by the Collaborative Research Network (CRN) of Federation University Australia and the Nanjing Institute of Geography and Limnology Chinese Academy of Sciences (NIGLAS), brought together a range of scientists including the neo-ecologists, palaeoecologists and hydrologists from both Australia and China. More than 100 students and scientists from across China attended the symposium. A majority of papers presented at the symposium have overlapping themes between ecology and hydrology of the large river and wetland systems that are exposed to a range of impacts posed by humans and recent climate change. The research focus of this volume is around the topic highlighting "the conservation and management of degraded wetlands in Australia and China and the maintenance of a long term ecological resilience".

Each section represents an overarching theme. The first section focuses on baselines of wetlands, and wetland ecosystem services. Human domination in the biosphere is the main cause for decline in services provided by ecosystems due to alteration of global biogeochemical cycles. The papers outline the need for managing the natural capital of the human society in a sustainable way. Given the globally significant river systems in Australia and China, their maintenance is essential for future generations.

The second section focuses on the theme, ecosystem response to human disturbances, hydrological changes and reduction in ecological resilience. Recent industrial and cultural developments in Australia and China have modified attributes of wetland ecosystems. Modern farming practices have made significant implications for physical and hydrological features of floodplain wetlands, including the changes in water quality and sediment processes. The research papers within this theme provide insights into a large scale alteration of rivers in Australia and China for agricultural, hydropower and industrial development and its subsequent implications for conservation and management of ecosystems and biological diversity.

The third section focuses on ecosystem resilience and regime shifts of wetland ecosystems. Climate change has altered flood events, channel morphology, nutrient dynamics and growth and reproduction of wetland and riparian biota as well as increased nutrients and sediments transport into the wetland. The research within this theme is trying to unravel the direct and indirect mechanisms behind the impact of climate change on wetland ecosystems.

The fourth section focuses on the knowledge transfer and the use of management approaches for ecology and hydrology of the large river basins in Australia and China over a long period of time. The characteristic changes in hydrology and ecology, with respect to anthropogenic impacts and natural climate variability of water resources, are significant for future management of water resources. The research papers within this theme provide a better understanding of the sensitivity of energy and water resources to changes in atmospheric conditions, locally through connections with ENSO and Indian Monsoon variability, and what is expected by ground water resources through modelling.

The production of this volume would not be possible without the generous contributions from the participating scientists from the collaborating organisations of both countries. I am also

thankful to NIGLAS and CRN for organising this symposium jointly. Hayley Collins from the Centre for eResearch and Digital Innovation, Federation University Australia is acknowledged for design and typesetting management of this document.

Finally, with gratitude for the privilege to the Australia China Science and Research Fund (ACSRF) for awarding the Group Mission funding to Federation University Australia, and for supporting Australian scientists by enabling them to travel to China to present their research outcomes among the Chinese scholars.

Introduction

Some of the large river basins of Australia and China have a history of global significance. For example, two of the world's most significant river basins, the Murray and Yangtze Rivers, have been intensively developed for the provision of food and water resources. The Yangtze River System supports one-tenth of the world's population, supported by an economy dependent upon irrigation, hydropower and tourism, while the economy of the Murray River Basin, valued as high as $14 billion per annum, is supported by the River Murray for irrigation, hydropower and agricultural industries.

The impacts associated with the Murray River Basin following the arrival of Europeans, are mainly forest clearance for agriculture and subsequent development of water resources for irrigation through river regulations. While in the Yangtze River Basin, over recent decades the major impact has been associated with the transformation of the agrarian-based economy to an industrial-based economy. As a result of these impacts on natural habitats, the functioning of the larger river wetland ecosystems has become critical. Further, climate change has threatened the ecosystem functioning of the wetlands of both river systems.

Evidence suggests that the response of ecosystem structure and function of the Murray and Yangtze River Systems to various stressors is non-linear in nature. Nutrient and climate-driven complex ecosystem processes and associated feedback mechanisms have often led to a regime of thresholds, which can cross from one state to another. The nature of such changes through time is poorly known in wetlands of the large river systems of both countries. The regime change has impacts on biodiversity and ecosystem functions and subsequently, ecosystem services. Knowledge generated in the large river systems in Australia and China will be crucial for developing strategies to mitigate the ongoing pressures associated with intensive use of water resources and food production, and assist decision-makers in ensuring their sustainability.

Scientists from Federation University Australia (FedUni) and the Nanjing Institute of Geography and Limnology Chinese Academy of Sciences (NIGLAS) have long been aware of the condition of rapidly changing lakes, floodplain wetland environments, including the wetlands of the two large river basins, the Murray and Yangtze basins of Australia and China, and have made a call to unravel the historic changes in ecology and hydrology through exchange of knowledge and skills transfer. Australia China Wetland Network Research Partnership is an outcome of the call by FedUni and NIGLAS to foster a dialogue among Australian and Chinese Scientists in water resources management across Australia and China and to help develop well targeted collaborative research programs in lakes, rivers and wetlands of river basins with the aim to restore degraded ecosystems, which generates goods and services to people, and helps build a resilient society.

Section 1: Maintaining baselines and ecosystem services

Determining baselines in wetlands

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Abstract

The Ramsar Convention has formalised the concept of determining baselines in wetlands as a means of ascertaining change in the ecological character of wetlands. This has generally been based on the assignment of a baseline or reference condition at a point in time where suitable data is available. In some instances, this has been equated with a pre-disturbance state. At the same time, there have also been difficulties in describing the natural succession and variability within these states. This reflects a difficulty to come to grips with the ecological processes that shape and change wetlands in both short and longer time periods. The advent of global change, including climate change, has further complicated efforts to ascertain the ecological condition of wetlands and to determine when a wetland has changed adversely in response to human activities. In response, attempts are being made to predict the future condition of wetlands under climate change and to identify thresholds for what is generally seen as adverse change. A fundamental stumbling block has been the extent of information on which to base such assessments.

Keywords
Wetland ecosystem, ramsar convention, ecological character, baselines in wetlands, ecosystem services

1. Introduction

The Ramsar Convention on Wetlands is an intergovernmental environmental agreement with 168 member countries (June 2014). It was developed in the 1960s because of concerns by governments and non-governmental organisations over the conversion and loss of wetlands and the impact on people and nature. The text was signed by 18 countries in the Iranian city of Ramsar in February 1971. The member countries and observers meet every three years to discuss and review administrative matters, national reports on the implementation of the Convention, and technical guidance for the wise use of wetlands. These meetings have produced 289 formal decisions (some with multiple parts) to date, with the technical material derived being compiled in 21 handbooks for the wise use of wetlands (Secretariat of the Ramsar Convention 2011). The decisions and technical material provide a framework for national and international collaboration for the conservation and wise use of wetlands.

The framework is based around three topics, the wise use of all wetlands, designation and management of Wetlands of International Importance (Ramsar Sites), and international cooperation (Gardner and Davidson 2011), as shown in the mission statement, namely ‘the conservation and wise use of all wetlands through local and national actions and international cooperation, as a contribution towards achieving sustainable development throughout the world’ (www.ramsar.org, accessed 29 June 2014).

2. Wise use of wetlands – maintaining their ecological character

Wise use of wetlands has been at the centre of efforts by the Convention to manage and restore wetlands, and is now defined as 'the maintenance of their ecological character, achieved through the implementation of ecosystem approaches, within the context of sustainable development' (Finlayson et al. 2011). The technical and policy domain for making wise use of all wetlands has been outlined in the handbooks for the wise use of wetlands (Secretariat of the Ramsar Convention 2011). Efforts are underway to extend this domain with an increased focus on the interactions between policy and science issues (Finlayson 2014).

The essential objective of the wise use approach is to maintain the ecological character of all wetlands through a combination of integrated technical and social-ecological activities. Ecological character is "the combination of the ecosystem components, processes and ecosystem services that characterise the wetland at a given point in time." For the purposes of the Convention, maintaining the ecological character refers to the prevention of human-induced adverse alteration of any ecosystem component, process, and/or ecosystem benefit/service. This needs to take into account the natural variability and succession that occurs in wetlands (Finlayson 1996).

The starting point in making wise use of a wetland has been the provision of a baseline against which change can be measured, generally the current condition or a preferred historical condition (requiring restoration). To be effective, the ecological character should describe the natural variability in character, and provide a basis for determining limits of acceptable change in that character. This raises questions about the adequacy of the information base available for describing the wetland, in particular that needed for establishing the natural range of variability in character and whether this can be separated from human-induced change. For example, can the effects of climate variability on flow regimes be adequately separated from those caused by the effects of physical regulation of water in rivers/lakes through the construction of barrages or weirs?

Further, when a change has been detected is it significant or trivial? This introduces the concept of determining limits of acceptable change and whether a change in one parameter or indicator effectively reflects a change in other parts of the wetland. As wetlands usually comprise many species with different life cycles and responses to changes in ecological processes and human-induced changes, these issues may require more than trivial attention. There is an extensive literature on choosing indicators but hitherto there has been little application to determining the significance of change in the ecological character of wetlands, including changes in the ecosystem services.

3. Global environmental change and baselines

The impact of global environmental change on wetlands has been well documented through assessments such as the Millennium Ecosystem Assessment (MEA 2005 a,b). The MEA has reported unprecedented levels of change in ecosystems with significant and largely irreversible changes to species diversity with humans having increased the species extinction rate by as much as 1,000 times over background rates, with 10–30% of mammal, bird, and amphibian species currently threatened with extinction.

The Assessment also reported unprecedented change in global drivers of change, such as the biogeochemical cycles, whereby the flow of biologically available nitrogen in terrestrial ecosystems doubled and flows of phosphorus tripled, and >50% of all the synthetic nitrogen fertilizer ever used has been used since 1985. There has also been unprecedented change in the extent of water storages and extraction of water for human uses.

The extent of loss and degradation of aquatic ecosystems, both inland and coastal/marine, has also increased with more than 50% of some wetland types having been lost, although extrapolation of this figure globally is speculative only given a lack of evidence about the extent and distribution for some types (e.g. ephemeral or intermittent) or regions. There is though, increased pressure on wetlands in Asia, Africa and southern America, and in small island states. Species indices also show that wetlands/rivers may be in faster declining rates than rainforests and savannah grasslands.

As a consequence of global change, many ecosystems could be or have already shifted away from historical references. Scientists and managers are also accepting the difficulty of returning wetlands to a ‘pristine’ state. At the same time, historical conditions remain the applied and conceptual cornerstone for restoring and managing human-modified systems. These issues have been receiving increased attention.

4. Emerging baseline concept

An Emerging Baseline (EB) concept has been proposed as an approach to recognise human-induced baselines for wetlands and provide a framework for assessing the importance of global change in wetlands (Kopf et al submitted). The approach recognises the value of historical records to predict the restoration potential of wetlands, but confronts the reality that many wetlands have moved far from historical references and that reinstating historic processes, such as the natural flow regime may not be sufficient to restore the wetland (e.g. removal of alien species, or the re-establishment of native species or ecological processes). Further, global environmental change has resulted in changes in species traits including: reduced body size-at-maturity of fish species due to size-selective harvesting; chemical resistance; changes in genotype, growth rate, morphology; altered migration pathways and the timing of reproduction.

EBs have one attribute in common, namely that contemporary human-modified systems or components of biodiversity functions differently than the historical baseline. The application of a historic reference, therefore, for assessing the extent of change may not accurately predict how the contemporary human-modified system will respond to management or restoration activities. It is emphasised though that not all ecosystems, or components, that have been affected by human-induced global change are classifiable as EBs.

Two criteria have so far been identified for classifying an EB, namely, there is a human limit on restoring a driver or process to a historic level which limits the restoration potential, and human-induced alteration of a functional response to a historic driver or process. It is recognised that emerging baselines will fluctuate over time and space and should be quantified at appropriate scales and provide estimates of variability – this could be done, for example, when describing the ecological character of a wetland as part of a management plan. The description will need to determine how and when the human-modified system or component of the system, changed from the historical reference. This in itself implies that there is sufficient understanding of the system, possibly through the use of a conceptual model of the system, and that any assessment and monitoring of the main drivers of change can identify where it differs or has diverged from the baseline.

Further investigations are underway to explore how and when to apply EBs. This includes taking steps to firstly identify and quantify, as far as possible, the causal processes or drivers which historically sustained a wetland or component of the biodiversity of the wetland, then ascertaining how different this is to the emerging baseline. The latter would preferably be done on a statistical basis which implies the existence of sufficient information and understanding of the casual-effect relationships between the natural and human drivers of change and the biodiversity. The final steps is to then ascertain whether or not the human drivers can be reduced or removed allowing the wetland to be restored fully to the historic, or least disturbed, level. Given the absence of information about wetlands in many parts of the world (MEA 2005b) this may prove to be extremely difficult – the development of conceptual models of the ecological functioning of wetlands and the impact of drivers of change may make this more achievable.

An underlying component of the EB concept is that the decision to manage or restore a wetland ecosystem towards a historic reference or an EB will ultimately be determined through a socio-political process that may or may not be fully informed by science-based information. This process could be guided by empirical evidence (the science) concerning the biodiversity and ecosystem services of the wetland as they existed in the past (the historical reference) and as they may exist in the future (the emerging baseline), but equally may be informed more by social factors. An as yet unanswered part of considering change and the importance of historical or emerging baselines is the time period being considered – the timescale being considered could shape both the usefulness of the existing ecological information about the wetland, as well as the direction of the social process used to manage or restore the wetland.

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Understanding the historical trend of ecosystem services for river restoration: a case study of the Yongding River, China

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Abstract

Freshwater ecosystems are changing rapidly worldwide, raising sustainability concern for rivers' health and for communities relying on their ecosystem services. Knowledge on historical trend of ecosystem services is key to formulate river management and restoration policies; however, there still lacks such knowledge for many rivers in China. We evaluate the ecosystem services of Yongding River (it was once called the "Mother River" of Beijing, but is suffering from serious dry-ups and water quality problems) through an intensive effort involving the local stakeholders. The assessment shows that the total values of the river ecosystem services have decreased by 40% over 1978-2009. Among all the services, water supply and cultural services have suffered from the sharpest declines, which have decreased by 94% and 54% respectively. We conclude that restoring culture-related services may be the most effective way to enhance the Yongding River ecosystems in the short run, but setting up a monitoring program to measure physical and biological parameters is also a priority to trace long-term changes of the river ecosystem services following restoration.

Keywords
Ecosystem services, river restoration, ecological degradation, Yongding River
Beijing

1. Introduction

Freshwater ecosystem is a part of human life providing a range of functions including industrial water supply and livelihood for food production, irrigation, electricity generation, climate regulation, navigation, culture and recreation (Constanza et al, 1997; Yu, 1990, Qinghua et al, 2003). However, due to both natural variability and human activities, the freshwater ecosystem, including the river ecosystem has degraded severely. As a result of rapid economic and social development, the world's rivers are influenced largely by land use activity, climate change, altered hydrological cycles and loss of biodiversity. The condition of the river ecosystem has been seen as dried-up with elevated temperatures, less hydrological connectivity, high water residence time, increased nutrient and sediment loads, exposed to new chemicals, simplified physical structure, exposed to invasive species, and lost biodiversity affecting structure and function, consequently reducing the ecosystem services (Sbaer, 2010; Bullock et al. 2011).

The rapid change of the freshwater ecosystems worldwide raises the sustainability concern for rivers' health and for communities relying on their ecosystem services. The ecological restoration of degraded rivers becomes ensuing focus of each nation. In order to enhance the environmental conditions of fluvial systems, there are widespread river restoration projects worldwide (Campana et al .2014; Zheren, 2004). Some developed countries have focused on the “chemical, physical and biological management” approach of degraded rivers, which include pollution control, conservation of catchment and designing natural river channels to restore ecosystems. In the US, the river ecosystem restoration management program is highlighted on the basis of the classification of the river, evaluation and monitoring. The “river basins method” has been proposed as the best practice for restoration through stressing the “connection of upstream to downstream and the surface water to ground water” system. In the 1980s, Germany and Switzerland proposed the concept of “re-naturalising” approach aiming to restore the degraded rivers. Similarly, the UK adopts the “Near Natural” technique, prioritising the greater ecological functions of rivers. The Netherland stresses the synthesis of river restoration and flood control, proposing the concept of “Room for the River” to give back to rivers, the space used for economic development. In Japan, the “Naturally Diverse River Works” concept is highlighted for a healthy ecosystem with high biodiversity and pristine landscape. China has also increased emphasis on water restoration both in concepts and practices, focusing on maintenance of river’s ecological flow (Zheren, 2004; Yannan et al., 2004).

The river restoration itself is “the process of assisting the recovery of an ecosystem that has been degraded, damaged or destroyed” (SER 2004). No matter what kind of concepts and methods adopted, the restoration should respect historical conditions, either as the basis for explicit objectives or to reset ecological processes to defined pre-disturbance conditions (Harris et al. 2006; Juren, 2003). Knowledge on historical trend of ecosystem services is a key to formulate river management and restoration policies. How to pursue the success of river restoration by tracing the historical trend of river’s ecological services is the major question the river managers are facing before having the restoration actions. In order to address this question, we aim to conduct a case study on ecosystem services in the Yongding River, near Beijing, by working closely with the local stakeholders.

2. Methods and materials

2.1. Study area

Yongding River is located in the northeast China (Longitude 112°～117°45′, Latitude39°～41°20′) with total area of 47016 km2. It originated from Inner Mongolia, flows through Shanxi province, Hebei province, Beijing and Tianjin City. As one of four key flood-control rivers in China, it plays important roles in flood control, water supply and water source conservation for the development of Beijing, the capital city of China, so that called as Beijing’s “Mother River”. Yongding River flows along Beijing from Youzhou to Liangge Zhuang, covering the five districts of Beijing, namely Mentougou, Shijingshan, Fengtai, Daxing and Fangshan. The length of main channel of Yongding River in Beijing is around 170m, with the area of 3168 km2. The Yongding River in Beijing could be divided into 3 streams based on the terrain, of which the upstream is the gorge from Guanting Reservoir to Sanjiadian, the midstream is from to Lugou Qiao and the downstream is from Lugou Qiao to Lianggezhuang (Figure 1).

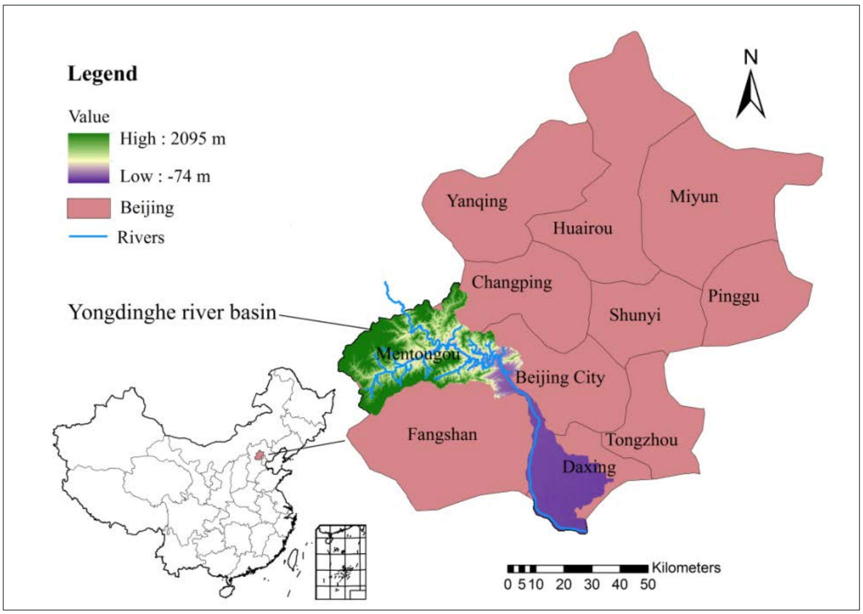

Figure 1: The map of Yongding River

Since the 1980s, the demand for water is increasing rapidly as a result of rapid socio-economic development in the region. Meanwhile, the industrial and residential pollutants pesticides, chemical fertilisers and poultry wastes are, drained directly into the river. As a result, the water quality of the river has deteriorated, drying-up has occurred and has increased the deposition of sandstorm. Both limited water supply and degrading ecological condition across the river catchments, have threatened the agricultural productivity and livelihood of the people life.

2.2. The classification of ecosystem services for the Yongding River

The Millennium Ecosystem Assessment (MEA) defines the ecosystem services is the "benefits that people obtain from the ecosystems" and classifies these services into four categories: "provisioning services such as food and water, regulating services such as regulation of floods, drought, land degradation, and disease, supporting services such as soil formation and nutrient cycling, and cultural services such as recreational, spiritual, religious and other non-material benefits" (MEA, 2005).

Based on the MEA classification on ecosystem services, we classified the ecosystem services of Yongding River into four categories: provisioning, regulating, cultural and supporting services. Ecosystem services data were collected based on onsite survey, questionnaire, interview and discussion workshops. Suitable indicators were established to evaluate the ecosystem services.

2.2.1 Provisioning services

The provisioning services of river ecosystem refers to the capacity of river itself and its ecosystem to provide products that could be used for market exchange and brought humans for direct benefits. These products are the direct services provided by the ecosystem maintaining human's production and life. Yongding River delivers a wide range of services that contribute to the development of people around, such as water supply for livelihood, agriculture, industry and environment, fishery production and hydropower generation.

Water supply

Rivers play key roles in storage and retention of freshwater, which is the origin of life. Yongding River is the important water source, supplying freshwaters to local people. Located at the upstream of the Yongding River, the Guanting Reservoir is one of the largest water sources for Beijing City. Recent years, due to the severe drought, drying-ups are a regular occurrence of the Yongding River. As the total water supply of the Guanting Reservoir is from the Yongding River, reduction in water in the reservoir and its declining supplies to the industries reflect the characteristics and changing trend of water supply of the Yongding River. The total water supply of Guanting Reservoir has declined significantly from 1251.04million m^3 in 1980, to 60 m^3 in 2009. From the year 2006, the water storage of Guanting Reservoir decreased so dramatically, that it could only serve the industrial water use, and the supply of water for the other three sectors was nil.

Fisheries production

The river ecosystem also provides the necessary conditions for fisheries production. The fisheries production not only contributes to local food supply but also supports recreational activities. The fisheries of the Yongding River are significant around the Mentougou area and Guanting Reservoir, where the fish production accounts for the total of the Yongding River. With the exacerbated water scarcity of Beijing City, the areas for fisheries have progressively declined, reflecting the shrinkage of fish production. During 1978-1986, the fisheries production of the Yongding River was relatively low. As a result of better policy, in 1993, fisheries and the production increased to its peak in 1994. In the following four years from 1995 to 1998, the production slowed down, but there was rising trend again from 1998 to 2009.

Hydropower generation

The Guanting Hydropower station of the Yongding River was built in 1954. It was the first automatic power station designed and built by China, with the total capacity of 350000KW, depending on the water storage capacity of 2.27 billion m^3 of the Guanting Reservoir. In 1959, the power generation reached the peak of 160.2 million kWh. Following the Guanting Hydropower station, the water storage of Guanting Reservoir declined significantly. In 2008, the water storage of Guanting Reservoir dropped as low as 163 million m^3. The power generation also decreased to 0.9 million kWh.

2.2.2 Regulating services

The regulating services are the services the natural environment the river ecosystem provides benefits to humans. Five regulating services of the Yongding River have been mapped, including water storage and retention, flood regulation, sediment transport, purification of water and air and climate regulation.

Water storage and retention

Surface and ground waters of the Yongding River were analysed for the water storage and retention services. The amount of surface water fluctuated around 50~80 million m^3 during 2000 to 2009, with an exception of 322 million m^3 in 2008, as a result of frequent and heavy rainfalls. The amount of ground water during this period was around 210 million m^3 ~280 million m^3, with an exception of 352 million m^3 in 2008.

Flood regulation

One of the Yongding River's flood control mechanisms, the Guanting Reservoir which was built in the 1950s, covers the total flooding area of 43400 km^2, with the total capacity of floodwater around 2 billion m^3r. The second flood control reservoir contains three sub-reservoirs, Daning, Daotian and Machang, with the total capacity of 44 million m^3 floodwaters.

Sediment transport

During 1956 to 1979, the annual average of sediment transport of Yongding River was ranged in largest amount of around 28.62t/km^2. With the yearly lowering water level, the amount of sediment transported by the river was also decreased, and the annual average reached its lowest point 18.03 t/km^2, during 1980 to 2000.

Purification

The Yongding River also performs the function in purification for water and air. The forest land in the gorge area of the upstream Yongding River is the natural carbon sink, with the functions of carbon storage and oxygen release, as well as absorbing other poisoning gases. The shelter belt along the midstream and the agriculture system in parts of downstream, perform the functions of absorbing SO_2 and reducing the noise.

2.2.3 Cultural services

As Beijing's "Mother River", the Yongding River owns abundant cultural resources such as religion, geology, mountain, folk custom, and ancient villages. The cultural services of the Yongding River include recreation, aesthetic, cultural heritage and education, of which the recreation and cultural heritage take the most important position (Peng et al, 2010). The increasing ecological deterioration of the Yongding River exerts great negative impacts on the culture and the landscape.

2.2.4 Supporting services

The Yongding River provides habitat for biodiversity. Based on the onsite investigation and statistics, the Wetland Natural Reserve in Mentougou district is affluent in all kinds of species. There are 527 types of vascular plant, 520 types of angiosperm, 4 types of fern and 4 types of gymnosperm. There are 11 types of state protected wildlife, including the 2 types of grade 1 protected species of *Ciconia nigra* and *Otis tarda* and 9 types of grade 2 protected species like *Cygnus cygnus*, *Aix galericulata* and *Grus grus*.

2.3 Valuation of ecosystem services

According to classification of the services and designation of indicator system for Yongding River, we used different approaches for valuation of these services (Table 1).

Table 1: Valuation of ecosystem services of Yongding River

Categories	***Ecosystem service related good***		***Quantification indicator***	***Valuation method***
Provisioning	Water supply	Surface water	Agricultural water	Market value method
			Industrial water	
		Ground water	Water for environment	
			Water for life	
	Fishery production		Aquatic products	Market value method

	Hydropower		Power generation of Guanting hydropower station	Market value method
			Power generation of Xiamaling hydropower station	
			Power generation of Xiaweidian hydropower station	
Regulating	Water storage and retention		Surface water	Shadow engineering method
			Ground water	
	Purification	Water purification	Load capacity for nitrogen	Shadow engineering method
			Load capacity for COD	
		Air purification	Amount of anion increased	Shadow engineering method
			Amount of dust absorbed	
	Flood regulation		Damage area/ storage capacity	Shadow project method
	Sediment transport		Amount of sediment transported	Shadow project method
	Carbon sequestration and oxygen release		Carbon fixation	Afforestation cost method
			Releasing oxygen	Industrial oxygen-producing method
Cultural	Recreation		Tourism	Direct market method
			Recreation	Travel cost method
	Aesthetic		Real estate appreciation	Willingness to pay method
	Cultural heritage values		Cultural values	Willingness to pay method
Supporting	Biodiversity		The grade 1 protected species	Prevention cost method
			The grade 1 protected species	

3. Results

3.1 Ecosystem services values in 2009

Based on the indicator system, data collection and different valuation methods, we worked out the total value of ecosystem services of Yongding River in 2009 as follows (Table 2 and Figure 2).

Table 2: The value of services of Yongding River in 2009

<table>
<tr><th>Categories</th><th colspan="2">Ecosystem service related</th><th>Value (CNY'00000000)</th><th>Sub-total value (CNY'0000000)</th><th>Total value (CNY'00000000)</th></tr>
<tr><td rowspan="3">Provisioning</td><td colspan="2">Water supply</td><td>3.36</td><td rowspan="3">3.85</td><td rowspan="14">415.74</td></tr>
<tr><td colspan="2">Fishery production</td><td>0.45</td></tr>
<tr><td colspan="2">Hydropower</td><td>0.04</td></tr>
<tr><td rowspan="5">Regulating</td><td colspan="2">Water storage and retention</td><td>18.11</td><td rowspan="5">227.75</td></tr>
<tr><td colspan="2">Purification</td><td>0.28</td></tr>
<tr><td colspan="2">Flood regulation</td><td>206.11</td></tr>
<tr><td colspan="2">Sediment transport</td><td>0</td></tr>
<tr><td colspan="2">Carbon sequestration and oxygen release</td><td>3.25</td></tr>
<tr><td rowspan="3">Cultural</td><td colspan="2">Recreation</td><td>12.66</td><td rowspan="3">169.64</td></tr>
<tr><td colspan="2">Aesthetic</td><td>0</td></tr>
<tr><td colspan="2">Cultural heritage values</td><td>156.98</td></tr>
<tr><td rowspan="2">Supporting</td><td rowspan="2">Biodiversity</td><td>The grade 1 protected species</td><td>10</td><td rowspan="2">14.5</td></tr>
<tr><td>The grade 2 protected species</td><td>4.5</td></tr>
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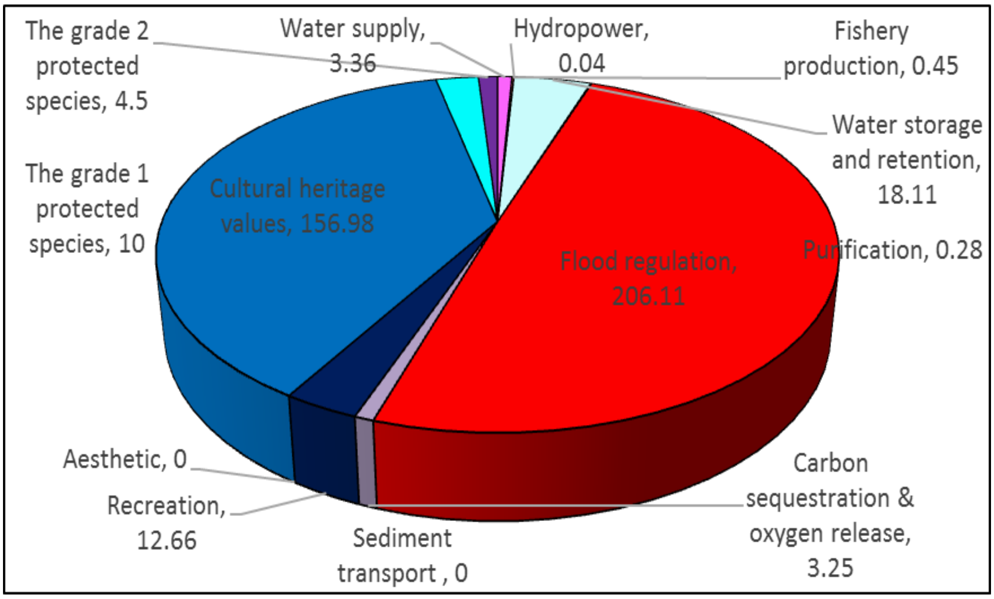

Figure 2: The value of services of Yongding River (Unit: 100 million CNY)

From the calculation, the total value of services of the Yongding River in 2009 was 41574 million Yuan, and it equals 32% of the total GDP for the five districts of Beijing City. The regulating and cultural services contribute to a big proportion, accounting for 96% of the total value. The value of flood regulation and cultural heritage are ranked in the first place, accounting for 50% and 38% of the total value respectively. As mentioned before, the Yongding River was the “Mother River” of Beijing City with unique cultural characteristics in its history. The valuation result shows that compared with other services, the Yongding River bears great advantages in services of flood regulation.

3.2 Historical ecosystem services values

Historical records reflect the development trend for the future. The value of services of the Yongding river is not stable but in changing trend. Based on the analysis of the available data for the period of 1978 to 2009, a progressively decreasing trend for the service value of Yongding River is observed (Figure 3).

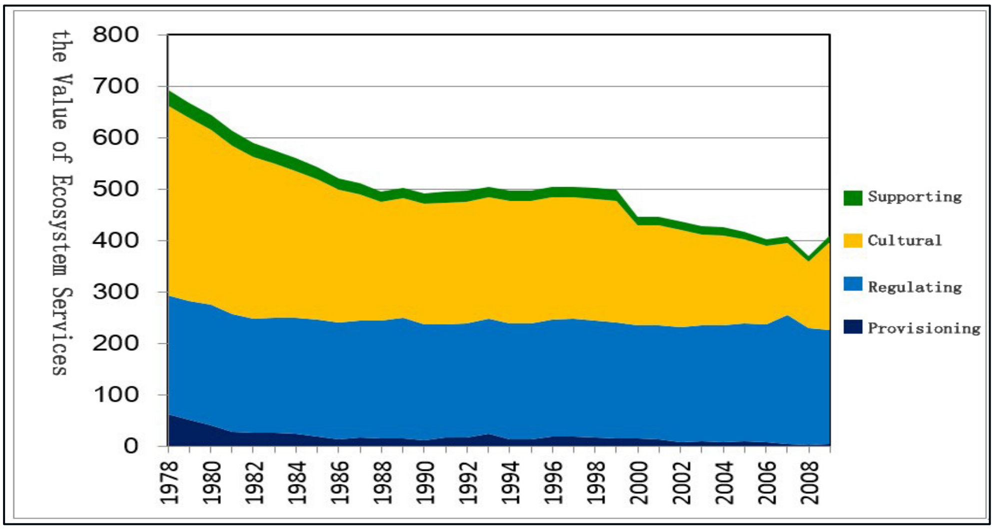

Figure 3: Historical ecosystem service value of Yongding River (Unit: 100 million CNY)

From 1978 to 2009, the total value of ecosystem services for the Yongding River decreased by 40%, from around 70 billion Yuan to 41.6 billion Yuan. The decrease in value is highly relevant to increasing loss of biodiversity and ecosystem. This period was over-exploited by human for economic development, amount of water resources, river areas had shrunk significantly and the land use patterns had changed to a great extent.

4. Discussion

4.1 Integrating ecosystem services into river restoration

According to the evaluation, it could be concluded that during the period of 1978 to 2009, there was a sharp decreasing trend for the services of provisioning, supporting and cultural, especially the services of water supply and cultural heritage, which had dragged down the total value (Figure 4 and Figure 5). The results obtained in this study provided a significant benchmark for developing an effective restoration program for the Yongding River.

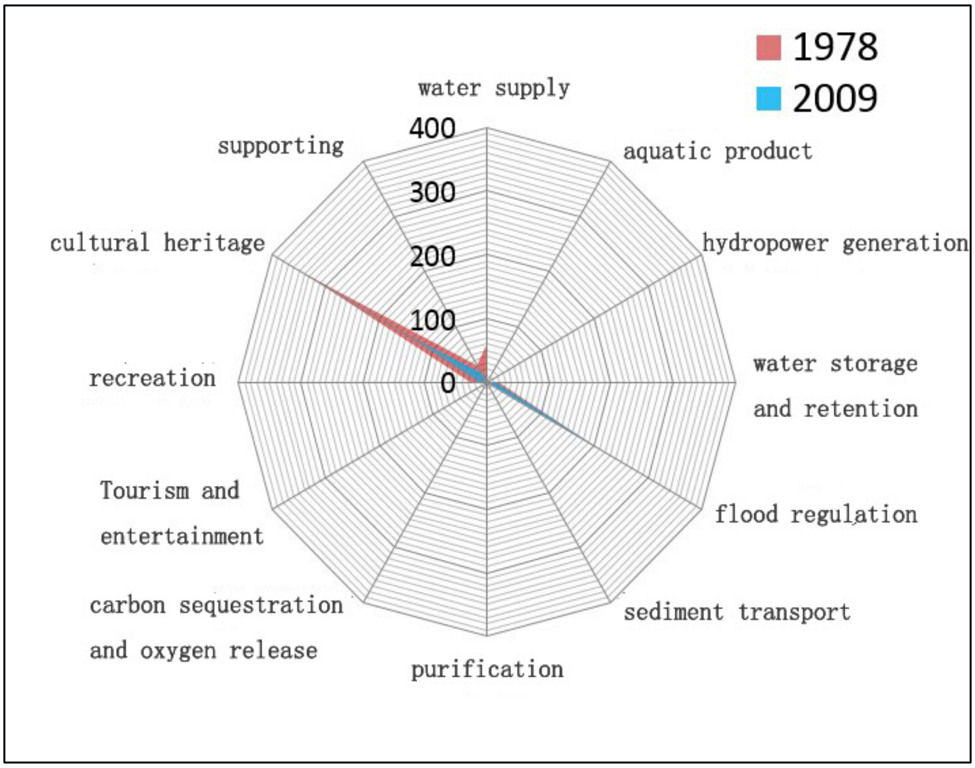

Figure 4: The historical value of services of Yongding River

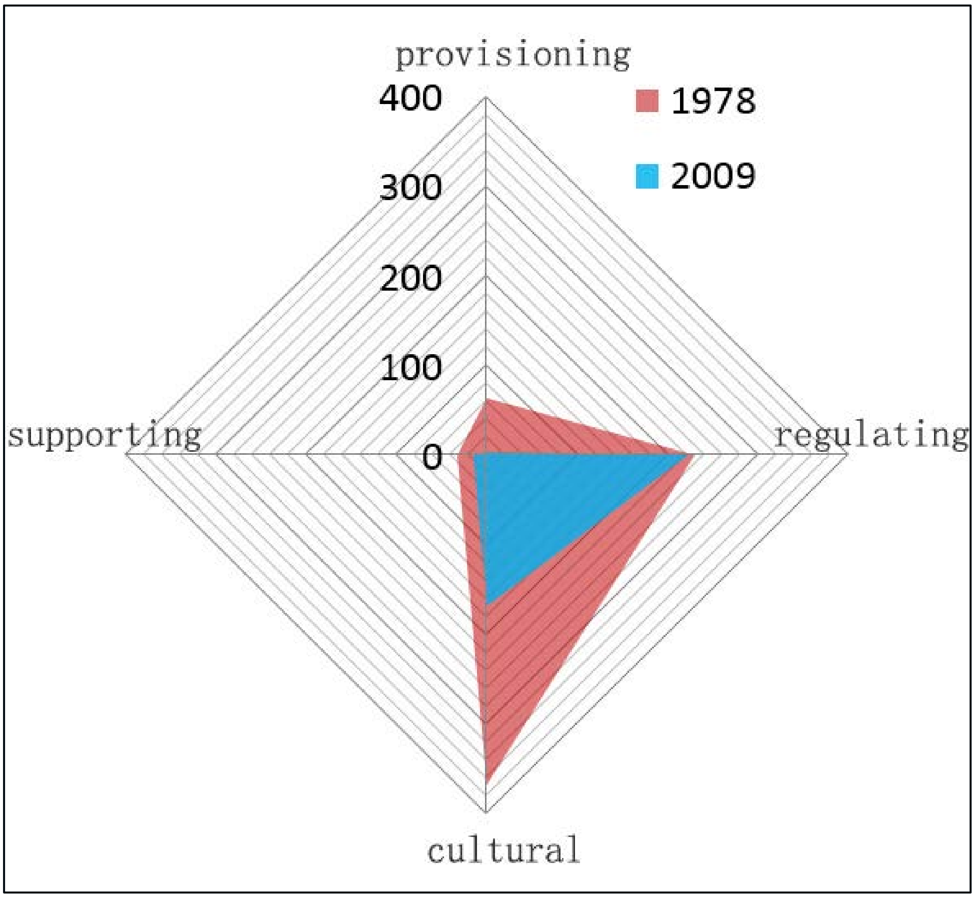

Figure 5: The historical value of four types of services of Yongding River

The output of water supply depends on both conditions of the natural variability and human activities. The Yongding River is located in the north and is prone to drought episodes as a result of its geographical location. The water that is controlled by the upstream reservoirs also block the water supply, therefore, it's very difficult to strengthen the capacity of water supply by current restoration techniques. Even though efforts are exerted, it is less likely to achieve improvement of the overall services. In comparison to many services, it could be more feasible and influential to improve the capacity of cultural heritages by using a range of river restoration approaches (Figure 6).

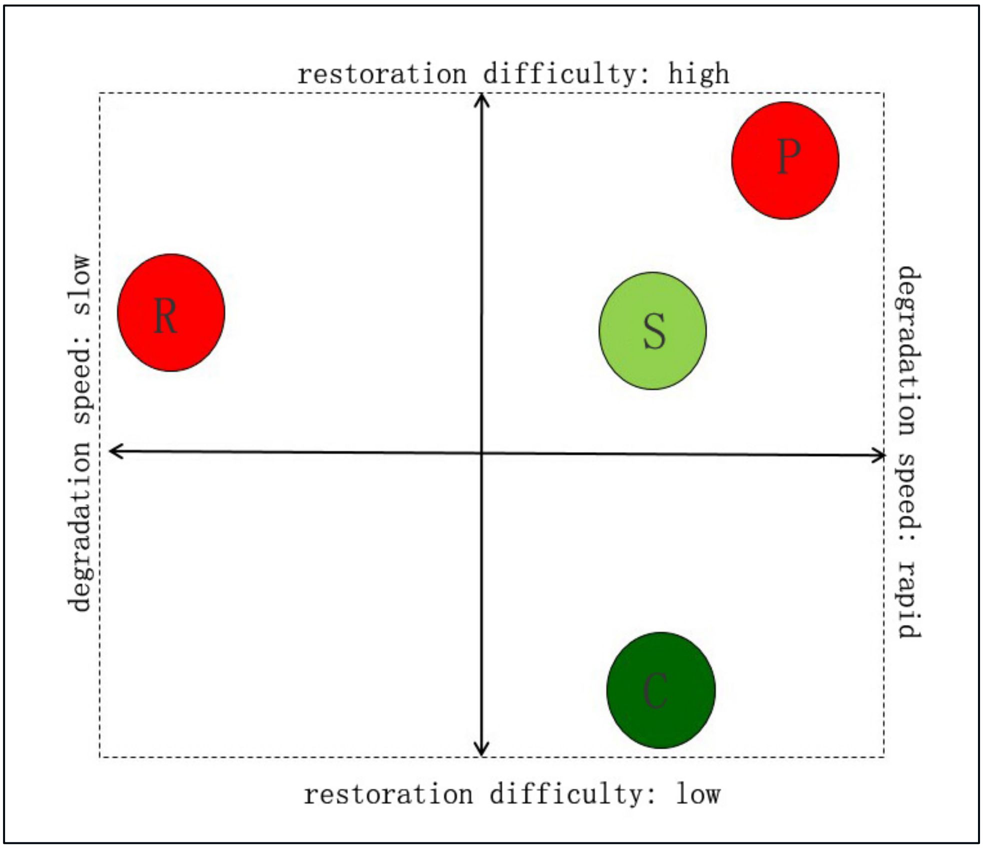

Figure 6: The comparison of restoration for different services

4.2 Stakeholders involvement in River restoration for the Yongding River

As a "Mother River" of the Beijing City, the restoration of the Yongding River was considered significant. A range of restoration ideas were collated by stakeholders. Actions were based on what specific benefits would generate from those ideas in the restoration process. The stakeholders' participation in the Yongdong River restoration program was classified into three categories: local residents' participation, experts' knowledge, and the actions of decision makers' (e.g Beijing Municipal Government, Mistry of Environment Protection). These stakeholders' participations contributed significantly to the every aspect of this study and their ideas were fully incorporated in the study process (Fig. 7).

4.3 River restoration program for the Yongding River

In recent years, the China's Central Government and the Beijing City's relevant Administrative Departments have taken a series of measures for the restoration of the Yongding River. As a result, this has improved the river's ecosystem significantly. The "Yongding River Ecological Corridor Construction Planning" was launched in 2010 aiming to construct a 1500 km^2 area ecological corridor by the end of 2014 with an increase of surface water areas of 1000 hectares and green areas of 9000 hectares. This is expected to improve the Yongding River's ecosystem services including the water supply, flood control and cultural and aesthetic values. Up until now, the first restoration project of the planning has been finished for water restoration. For example, the four dried-up or cut-off lakes of the river namely, Mencheng Lake, Lianshi Lake, Xiaoyue Lake and Wanping Lake have been restored to the total water area of 150 hectare and greening area of 120 hectare.

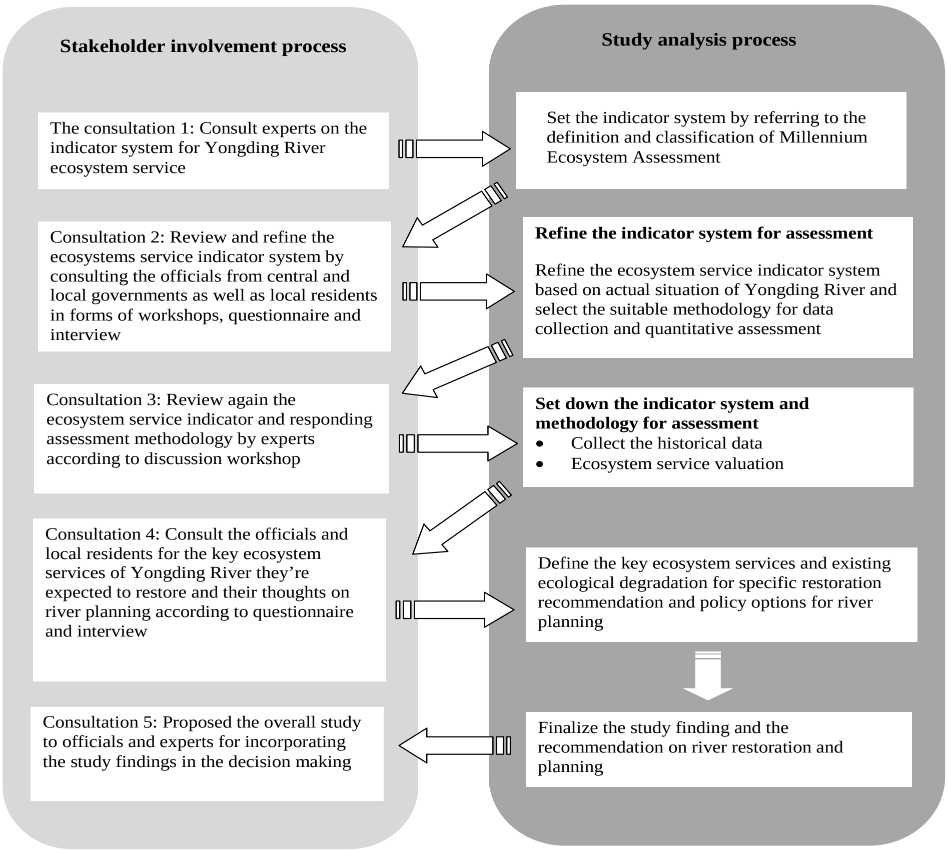

Figure 7: Stakeholders involvement in the study on Yongding River restoration

5. Conclusions

The Yongding River restoration has become the key for the development of the environment of the Greater Beijing City. To efficiently set the restoration goal and carry out the restoration project, it is essential to evaluate the Yongding River's key ecosystem services, based on which component would clarify the phased restoration measures and promote the successful restoration.

Acknowledgements

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Estimating visual quality, a component of culturally-associated ecosystem services in palaeo-lake environments

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Abstract

Evaluation of visual quality is essentially a multi-dimensional and multi-sensory experience of landscape assessment. Visual quality refers to the character, condition and quality of lakes/wetlands. It involves perceiving, preferring and valuing the visual quality by the public. Visual quality is an outcome of the perceptual, cognitive and emotional processes in response to visual stimuli of a lake environment. Visual quality therefore is dependent upon the perceptual and structural aspects of perceived scenes of wetlands.

Visual assessment, an evaluating process of gaining non-material or intangible benefits by people from ecosystems, through spiritual enrichment, cognitive development, self-reflection, recreation, and aesthetic experiences, has now become one of significant research areas under cultural components of ecosystem services. Public perception in such studies is composed from both the objective and subjective elements of human–landscape interactions. However, it is still a matter of debate whether subjective–objective realities are dichotomous or supplementary to enhancing the quality of human experiences in natural settings. In fact, much research considers them as inseparable and integral parts of landscape perception, despite the tendency for disintegrating landscapes into their constituent components. There is a fundamental theoretical divergence of opinions over the question whether a landscape has an intrinsic or 'objective' beauty, which may be in some ways measurable or comparable, or whether beauty is a value that can be only attributed subjectively to an area or a specific landscape.

Keywords

Visual quality, wetlands, lakes, palaeo-environment, expert opinion, recreation

1. Introduction

Estimation of visual quality of the wetland environment requires decomposing wetland landscapes into various biophysical components. A range of techniques were employed to measure visual quality of wetlands including the use of photomontage or simulated virtual scenes. Use of Geographic Information Systems, coupled with virtual reality is also being widely used as a technique of visual quality assessment. The evaluations of visual quality undertaken by either the public, experts or both assessment approaches have become increasingly useful in study of ecosystem services mainly the cultural services (Cheetri, 2006).

2. Methods

The visual quality, a component of culturally-associated ecosystem services of palaeo-lake environments was estimated. Using the paleo-data collected in two popular lakes, Dianshan and Liangzi in the Yangtze River Catchment in China, this paper develops a methodological framework to estimate the levels of visual quality of wetland environments (see Figure 1). Visual quality refers to the character, condition and quality of lakes/wetlands. It involves perceiving, preferring and valuing the visual quality by the public. Visual quality is an outcome of the perceptual, cognitive and emotional processes in response to visual stimuli of a lake environment. Visual quality therefore is dependent upon the perceptual and structural aspects of perceived scenes of wetlands.

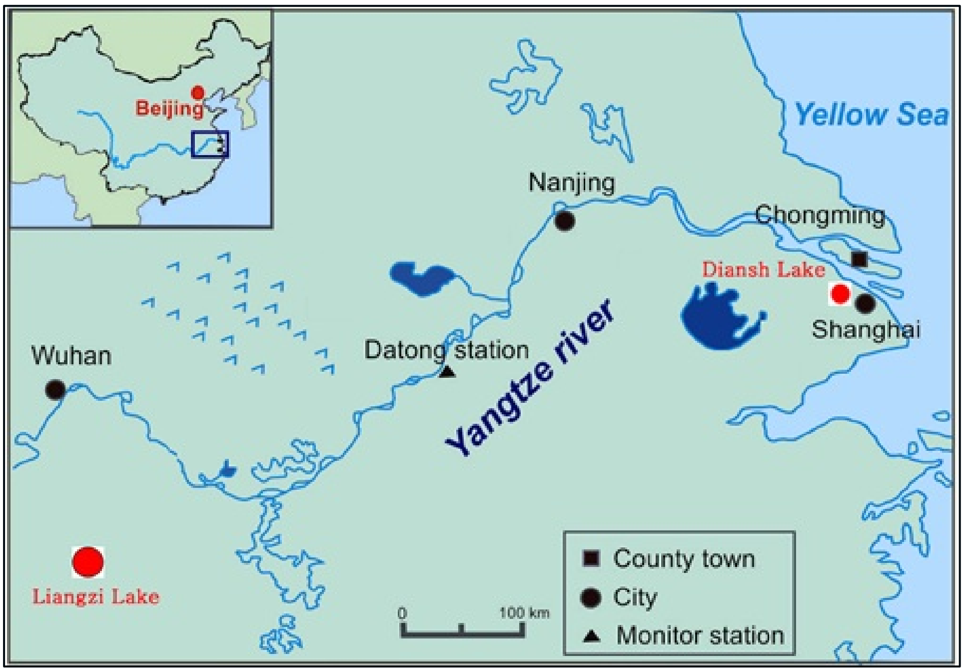

Figure 1: Location of Liangzi and Dianshan lakes

In this research, visual quality is measured using the biophysical properties of the wetland environment. The raw data contains estimated biophysical properties of wetlands such as sedimentary total nitrogen, carbon and total phosphorous, lake depth and aluminium, from 1896 through to 2008. The evaluation of visual quality is undertaken either by the public, experts or both. Expert judgements are used to subjectively categorise data into ordinal scale. Indicators, representing the cultural ecosystem services, are developed to represent visual quality of wetlands. These bio-physical properties are then reclassified and converted into a visual quality scale, ranging from 1 to 5, least attractive to most attractive. A set of surrogate measures (e.g. diatom) are generated to represent key visual quality indicators such as levels of biodiversity, water clarity, turbulence, contamination, odour; abundance of lacustrine species and water level. The scores allocated to each of the surrogate measures are aggregated to create a range of visual quality indices. These indices are then computed for different periods of time, to allow capturing and monitoring changes in the visual quality of Dianshan and Liangzi wetlands.

In this research, a five-step research framework has been proposed to estimate visual quality of paleo-lake environment. The analysis begins with developing key dimensions of aesthetic quality, which are largely driven by human perception of lakes and other features of the biophysical landscape. As listed in Table 1, a set of indicators are ascertained to represent the attributes of visual quality of Lake Environment. Since there are no direct indicators of visual quality, as linked to the biological and chemical properties of paleo-data, the visual quality could only be estimated through surrogate measures. Weights can be assigned to those indicators to differentiate human perception of landscapes.

A range of indices can be created to reflect visual quality. Generally, an index is a quantitative or a qualitative measure derived from a series of observed facts that can reveal relative positions or importance of any phenomena or condition such as health or environmental condition. Indexing requires subjective and/or objective data integration, which includes data standardisation, weighting and aggregation of various variables (Chhetri and Arrowsmith 2008). Composite index is one kind of linear aggregation method and it can be defined as an aggregation of the indicators' values which collectively convey information about the quality of some complex aspects or components of a condition. This is a method that combines a series of indicators in an appropriate manner. The weighted average technique is the most widely used technique of composite index. This technique can be used when the indicators have different weights in terms of importance in assessing performance. The final stage of the proposed framework generates estimates of visual quality of paleo-lake environment (Figure 2).

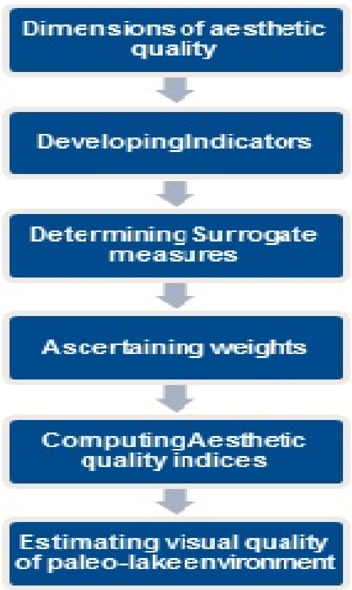

Figure 2: Proposed research framework for estimating visual quality of paleo-lakes.

Table 1: Suggested dimensions, indicators and surrogate measures of visual quality derived from paleodata

Dimensions	***Indicators***	***Surrogate Measures***
Complexity/Diverisity	Species assemblage of cladocerans	Total littoral cladocerans
	Species assemblage of diatoms	Total littoral diatoms

	Species diversity of cladocerans	Number of species
	Species diversity of diatoms	Number of species
	Species diversity of birds	Macrophyte density
Abundance/Diversity	Water level	Littoral: planktonic ratio of cladocerans
	Water level	Littoral: planktonic ratio of diatoms
Abundance/Diversity	Water clarity	Total Daphnia Eggs only
	Water clarity	Total diatoms (specific sps)
Visual clarity	Water clarity	Sediment Accumulation Rates
	Geomorphology	Grain size
	Geomorphology	Magnesium
	Water pollution	Total Phosphorous, Potassium
Disturbance	Water pollution	Lead concentration
	Water pollution	Iron (Fe) concentration
Smell	Odour	Total Organic Carbon/Loss-on-Ignition

3. Results and discussion

The application developed within this model has significant use for landscape planners, and managers of national parks and recreation management. This research will be the first attempt to compute visual quality of palaeo-lake environments. It will provide a systematic framework for monitoring changes in visual characteristics of wetlands, which in turn will enable the landscape managers in devising visual quality management plan to protect and conserve the wetland environment for future recreational use and estimating the cultural values of ecosystem services at a catchment level.

Acknowledgements

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Section 2: Ecological and hydrological impacts of humans

Assessing change in floodplain wetland condition in the Murray Darling Basin

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Abstract

Lowland Australian rivers and their floodplains have been affected by the progressive introduction of agriculture, flow regulation and invasive exotic species for more than a century. In the context of this complex suite of stressors, our capacity to understand and mitigate the causes of ecosystem change is limited by the lack of historical records of the condition of ecosystems over the past 200 to 300 years. However, records of change over this critical time period can be established through analysis of sedimentary records. Such records can be used to provide benchmarks of the range of natural conditions prior to European settlement and, by providing a long time series of conditions, enhanced capacity to detect trends and trajectories of change. Over the past two decades, more than 50 sediment records from billabongs, lagoons and waterholes throughout the Murray-Darling Basin have been subject to palaeoecological analysis. The picture that emerges from these studies is of ecosystems that have undergone substantial ecological change in response to human activities; however, there are also intriguing differences in the timing and nature of change experienced by aquatic ecosystems in different parts of the Murray-Darling Basin. These patterns of ecosystem response appear to reflect underlying differences in the resilience of these ecosystems in relation to different anthropogenic stressors, which, in turn, may result in contrasting hydrologic, geomorphologic and climatic contexts. This paper presents an attempt to systematically compile and summarise the palaeoecological evidence of change in the aquatic ecosystems of the Murray-Darling Basin and, in so doing, shed light on what the principal drivers of change are in floodplain wetlands across the basin, and hence provide guidance as to how these systems can be best preserved and restored.

Keywords
Wetlands, Murray River, salinity, regime shift, sediments

1. Introduction

The Murray-Darling Basin is Australia's largest spanning 1.06 x 106 km^2. It extends from sub-tropical zones in southern Queensland to temperate climates in the south. Its main watershed is the elevated, alpine to sub-alpine zones in the south-east associated with the Great Dividing Range. To the west, the effect of continentality ensures a drier climate. Here, low in the catchment, the main river channels pass through semi-arid, and even arid, climate zones. So, the main runoff is driven by cool season rainfall in the south-east, followed by snow melt, yet the northern parts of the catchment can receive warm season rainfall. Ultimately, river flow is impacted by high evaporation rates in the dry, western zone limiting the volumes that pass through the outlet to the sea in South Australia.

While the junction of the Murray and Darling Rivers hosted a relatively large population of indigenous Australians (Pardoe, 1998), the impact of human populations increased substantially after the arrival of European settlers from the 18th century. The catchment was extensively used for sheep grazing from early in settlement and the main rivers were a focus for travel and navigation owing to the difficulty of traversing the land. For navigation, trees were cut for firing engines and channels were cleared of woody debris. Early on stocking rates were high and impacts intense, particularly along stock routes where animals were driven to markets. The river waters were sought for irrigation agriculture from as early as 1888. When variable river flows brought calls for river regulation to ensure passage, the commissioning of weirs from 1922 most advantaged the irrigators, with river transport having largely succumbed to that on land. By 1936, much of the lower River Murray was regulated and significant additions to the reservoir system were marked by the commissioning of the Hume Dam (3000 GL) in 1936 and Dartmouth Dam (3800 GL) in 1979.

River regulation and a very high water reserve were responses to a highly variable climate, and the most variable runoff on Earth (McMahon & Finlayson, 1992). The region is subject to the cycles of dry and wet, associated with the El Nino Southern Oscillation which can bring significant drought phases and extensive floods. Memory focuses on the Federation Drought (1907), World War II drought and the recent Millennium Drought (1997-2009), considered the longest and deepest dry phase in European history (Gergis et al., 2012). The region is also impacted by multi-decadal climate variability with significant wet phases (1845-1898; 1946-1976), deemed flood-dominated regimes, and extended dry phases (1898-1946; 1997 - 2009) under drought dominated regimes (Warner, 1987). Notably, La Nina ensured that the early 20th century drought dominated regime was interrupted by the 1917 flood. One could speculate that, similarly, the 2010-11 floods were mere interruptions of an extended drought dominated regime that commenced in 1997, particularly given rainfall deficits across the region since 2011.

The condition of the waterways of the Murray-Darling Basin is recognised as being in a degraded condition (Norris et al., 2002). Only the remote, unregulated streams of the arid north-west are considered in good ecological condition with the remaining, intensively used systems considered degraded or even severely degraded. This state is closely associated with the level of abstraction and diversion of river flow, largely for irrigation agriculture. Much infrastructure was funded to significantly increase water allocations through the flood dominated regime of the post WW II period. This has left water users, and the environment, highly vulnerable to the recent drought, and in particular, a drought dominated regime should be the 'drought' persisting beyond the recent La Nina phase. In response to this water stress, the local authority has implemented a highly contested Murray Basin Plan that dictates that 3200 GL of water will be returned to the environment to restore its health. Depending on whether this water is redeemed through the purchase of water rights, or through water efficiency infrastructure, the provision of this environmental allocation will come at a cost of between $5.5B and $27.5B (Wittwer & Dixon, 2013).

However, the provision of water is likely. With the degraded condition of the Basin's aquatic systems, with the increased flux of salt, sediments and nutrients, only one of the drivers strongly implicated in the changing waterway condition. Little is known about the nature of the change in these drivers of change, with water quality monitoring programs commencing from the mid-1900s. This instrumental data is pre-dated by the clearance of vegetation from the catchment, the initiation of intensive cropping, the widespread application of irrigation water, the regulation of flow and the running of many million head of stock. Regional water tables and extensive erosion was noted by the 1930s. Therefore, reference to the available data will not reveal the impact of these early drivers of waterway change. Evidence for these changes is evident, however, in the natural archives of the biological elements of wetlands and rivers that are preserved in continuous sediment sequences. These records of change have been a focus of recent research and the condition of many wetlands, relative to their unimpacted state, has now been revealed (e.g. Thoms et al., 1999; Gell et al., 2005 a, 2005b; Fluin et al., 2007, 2010; Reid et al., 2007). The synthesis of a large number of such records can reveal changes at a sub-catchment scale and so the timing and magnitude of the influence of these drivers of change can be documented.

2. Sediment records

Long term records of change exist where the biological remains of organisms are buried with sediments in still water environments. Continuous sediment records that contain this evidence are more likely in sites which are perennially covered in water. The climatic diversity of the Basin dictates that some areas are more humid than others, and the high climatic and runoff variability ensures that basins in many areas are not always filled with water. Therefore, sites which provide continuous records are focussed in the humid upper catchments and along the main channels where overbank flows regularly refill wetlands. Critically too, many intermittent floodplain wetlands have become permanent, their water level linked to the height of the weir pool behind a nearby lock, and so provide sediment records mostly over the last century.

Central to the reconstruction of a long term history from sediment is the establishment of chronology. As dead carbon can be exposed to air, degraded or metabolised radiocarbon techniques are not straightforward, particularly where there are geological sources of old carbon. Also, ^{210}Pb approaches to date sediments are best where the proportion of aerial input of fine sediment is highest. Clearly, floodplain systems naturally receive river born sediments, episodically, creating challenges in developing depth-age models in sediment sequences (Gell et al., 2005b). Further, luminescence dating relies on a known dosimetry (Gell et al., 2007) and this varies if a lake is wet or dry. So, the establishment of time lines, especially at around the time of known first impact by European people, is challenging and few confidently dated sequences exist. Nevertheless, the use of a suite of dating techniques, including exotic pollen, provides for generalised chronologies which enable attribution of human impact; albeit not so precisely in time.

Traditionally, in Australia at least, fossil pollen was the main biological indicator extracted from sediments. This focus was to examine vegetation responses to long term climate cycles and was infrequently applied to reconstructing the impact of industrialised people, particularly along rivers where records were considered too shallow to be of great interest. Over the last 25 years, Australian palaeoecology has increasingly utilised fossil diatoms as an indicator. Diatoms reflect water quality and so have better enabled the provision of evidence of human impact on aquatic systems. In concert with radiometric dating of sedimentation rates and other preserved indicators such as Daphnia and plant macrofossils, changes in the salinity, nutrient status, turbidity, acidity and sedimentation of wetlands can be gauged.

Table 1: Summary of changes deduced from fossil diatom assemblages in records from Murray Basin wetlands (S = salinity; N = nutrients; T = turbidity; SR = sedimentation rates; A = acidity). *Cores were taken from 15 sites across each Coorong lagoon.

SITE	***Section***	***Core depth (cm)***	***Estimated Age Yrs***	***Change***	***Source***
Ajax Achilles	Lwr Murray	250	2200	T	Khanum unpubl.
Alexandrina nth	Murray estuary	85	7000	N, T	Gell et al., 2005; Fluin et al., 2007
Alexandrina sth	Murray estuary	500	7000	N, T	Fluin et al., 2007
Balranald Weir	Murrumbidgee	100	~100	S, N	Gell & Little, 2006
Berry Jerry	Murrumbidgee	150	>200	S, N, SR	Gell & Little, 2006
Billabong 21	Ovens				
Billabong 23	Ovens				
Billabong 38	Upper Murray				Reid et al., 2002
Boman	Murrumbidgee	85	<100	S, N, SR	Gell & Little, 2006
Bora	Macquarie	40	< 100	T	Yu et al. subm.
Brenda Park	Lwr Murray	140	>100	N, T	Fluin & Gell, in prep
Callamondah 1	Goulburn	125	>3000	N, alkalinity	Thoms et al., 1999; Reid et al., 2002
Callamondah 2	Goulburn	210			Thoms et al., 1999
Coonoocoocabil	Murrumbidgee	115	~100	S, N, SR	Gell & Little, 2006
Coorong nth*	Murray Estuary	87-594	7000	S, N, T, SR	Fluin et al., 2007; Gell in press.
Coorong sth*	Murray Estuary	66-800	2000	S, N, T, SR	Fluin et al., 2007; Gell in press.
Cullulleraine	Sunraysia, Vic	35	< 100	N, T	Reid et al., 2002; Fluin et al., 2010
Goolwa Barrage	Murray estuary	210	800	fresher	Fluin & Gell, in prep
Gooragool	Murrumbidgee	70	>200	S, N	Gell & Little, 2006
Gurra Nth	Riverland, SA	50	<80	S, N, T	Fluin & Gell, in prep
Gurra Sth	Riverland, SA	40	<80	T	Fluin & Gell, in prep
Hogans	Upper Murray				Reid et al., 2002
Homestead	Murrumbidgee	35	~100	S, N, SR	Gell & Little, 2006

Hopcrofts	Murray d/s Murrumbidgee	240	1000	SR	Gell et al., 2005b
Iona 2	Upper Murray				
Junction Park	Murrumbidgee			N, SR	
Kings	Murray u/s Darling, Vic	130	<150	T	Kattel unpubl.
Longmore	Gunbower, Vic				Grundell unpubl
Loveday	Riverland, SA	125	800	S, N, SR, A	Gell et al., 2007
Luna	Riverland SA	45	250	S, N, SR	Gell et al., 2007; Gell, 2010
Martin’s Bend	Riverland, SA	60	<100	S, A	Gell, 2010
McKenna’s	Murrumbidgee	40	~100	S, N, SR	Gell & Little, 2006
Moorna	Lwr Murray, NSW	60	< 100	S, N, T, SR	Heinitz unpubl.
Mundic	Riverland, SA	300	3000	T, SR	Gell, 2012
Murroondi	Lwr Murray, SA	1400	5000	N, SR	Gell et al., 2005b; Fluin & Gell, in prep
Paringa	Riverland, SA	130	>100	N, T	Fluin & Gell, in prep
Pikes	Riverland, SA	200	>100	N, T, SR	Khanum unpubl.
Psyche Bend	Sunraysia, Vic	150	<150	S, N, A	Silva unpubl.
Ral Ral	Riverland, SA	81	~80	N, T, SR	Khanum unpubl.
Russell’s	Murrumbidgee				
Scott Ck	Lwr Murray, SA	300		SR	Fluin unpubl.
Sinclairs	Lwr Murray, SA	90	<100	N, T, SR	Grundell et al., 2013
Swanport	Lwr Murray, SA	700	>100	SR	Gell unpubl
Tanyaka	Riverland, SA	280	2000	T, SR	Fluin & Gell unpubl.
Tareena	Lwr Murray, NSW	460	5000	S, SR	Gell et al., 2005a, 2005b
Yanco Weir	Murrumbidgee	55	~ 100	S, N, T, SR	Gell & Little, 2006
Yanga	Murrumbidgee				Reid unpubl

4. Natural ecological character

Many natural resource management instruments attempt to identify a benchmark condition against which to assess contemporary condition. In Australia, the Sustainable Rivers Audit assessed the condition of rivers against expected, natural conditions. The Ramsar Convention on the protection of wetland of international significance, attempt to identify ‘natural ecological character’ which is often described at the time a wetland is added to the register of sites, and the Water Framework Directive aims to return European waterways to ’good ecological condition’ (Bennion & Battarbee, 2007). Where human impacts have a long history, and are multi-faceted, contemporary measures, and even recent comparisons, prove futile in assessing the condition of a wetland, before impacted by industrialised humanity. Further, particularly where climates are variable, benchmarks ought to report on historic range of variability, and so need to characterise the response of systems to variations in non-human drivers to gain insight into internal dynamics. Given the low frequency of high impact climate cycles, it could be argued that benchmarks require temporal context spanning millennia.

The record represented in the sediments of few floodplain sites span this time frame. Tareena Billabong commenced accumulating sediment ~ 5000 years ago and it was revealed to be a freshwater lagoon at the time (Gell at al. 2005b). It underwent relatively little variation in response to Holocene climate variability, after a brief phase of increased river connectivity ending ~3000 years ago. Even so, this record attests to the fact that a range of ‘natural’ conditions can be invoked depending upon which point in time is chosen. Muroondi wetland, towards the terminus of the river channel, holds a surprisingly deep 14 m of sediment record. The record here reveals a 5000 year, hydroseral evolution with planktonic forms, increasingly replaced by those of shallow waters and eventually epiphytic diatoms reflecting the evolution of a marshland in a shallow basin. This site attests to direction change which occurs as sediments naturally filled.

5. Human impact

The impact of indigenous people on Murray River floodplains wetlands is not well revealed in the sediment records. While direct burning of wetland vegetation is invoked from charcoal records of sites elsewhere (Head, 1988; Mooney et al., 2011; Mills et al., 2013), the routine analysis of charcoal from floodplain wetlands remains an important area of future endeavour. In the absence of charcoal records, inference for the impact of early humans on wetlands is left to ethnohistoric and archaeological evidence.

The sediment records do, however, strongly attest to the substantial and widespread impact of humans since settlement by Europeans. The most significant change attributable to a post-contact age is a sustained increased in diatom-inferred salinity, from ~ 1880 AD, at Tareena Billabong as indicated by substantial increases in the abundance of diatom species, *Amphora veneta*, *Gyrosigma apiculatum* and *Tryblionella hungarica*. This early impact is attributable to intense use of the site being a narrow passage for stock to be run between the wine distributory system and the dunefields. Salinisation, reflected by sustained increases in salt tolerant diatoms, is widely evident. Psyche Bend Lagoon naturally supported oligosaline taxa but became hypersaline by the early 21st century. Diatom-based salinity reconstruction shows that sites deemed to be in good condition, such as Loch Luna, have undergone ten-fold increases in salinity (Gell et al., 2007). A wide range of, usually cosmopolitan diatom species are indicative of elevated nutrient concentrations (Chessman et al., 2008). The increase, in sediment records across the basin, in these taxa reveal the chronic eutrophication of the system. Further, a suite of tychoplanktonic species (*Staurosirella* sp.; *Staurosira* spp., *Pseudostaurosirella* sp.), that are advantaged in low light, turbid, habitat-simplified systems have come to dominate, sometime abruptly, in a wide range of sites (Fluin et al., 2007; 2010; Grundell et al., 2012) again reflecting a chronic, widespread impact, this time in the form of fine sediments. This is also reflected by substantial and widespread increases in sedimentation rates from a baseline of 0.1 -1 $mmyr^{-1}$ to 10-40 $mmyr^{-1}$ (Gell et al., 2009). Such high rates of accretion, in such shallow wetlands, poses a great risk to the persistence of these depressions as wetland sites and represents a significant risk to the Basin's ecosystems (Laurance et al., 2010). Lastly, exposure of sulphidic sediments, through low water levels during the Millennium drought, has driven unprecedented acidification (Gell, 2010). The colonisation of the diatom *Haslea spicula* in two of these sites before acidification may mark this species as an early warning indicator of this risk.

5. Regime shifts

Wetlands are known to undergo abrupt changes in condition on account of the breakdown of stabilising forces that tend to direct the condition of wetlands to an alternative stable state, enabling a greater variation in condition that may approach thresholds of 'irreversible' change (Sheffer et al., 1993). One such threshold relates to the light regime whereby aquatic plants, attached to substrates, receive less energy as turbidity impacts upon the light regime and their capacity to photosynthesise. Abrupt changes to Murray Darling wetlands have been observed, sometimes delayed relative to the onset of likely drivers, suggesting a non-linear, threshold response may be at play. The most vulnerable of the Murray River floodplain wetlands are those which are large and deep, where high proportions of the benthic habitat of the wetland become energy limited once increased turbidity limits the penetration of light (Reid et al., 2007). More shallow sites in the upper catchment are thought to be resilient to reduction of light penetration and those in the lowlands able to reset after drying (Reid & Gell, 2011). Detailed analysis is essential to understand the light and trophic dynamics that may be controlling the changes recorded from the sediments, to enable mangers to most effectively intervene to recover wetland condition. Only the palaeoecological record has been able to reveal the changes that may come from changes in feedback, and more detailed analysis of the nature of these abrupt changes promises much in our understanding of threshold changes.

6. Conclusion

Paleolimnological approaches shine a light on the nature of human impact on the floodplain wetlands of the Murray Darling Basin. Many wetlands have been substantially changed on account of river regulation, abstraction, eutrophication, salinisation, acidification and declining light regime. Some changes occurred early in European settlement and, in many cases, changes were abrupt. The report card for the condition of the wetlands of the Murray Darling Basin, at least those studied paleolimnologically to date, is one of severe, widespread degradations. Sediment fluxes in particular risk the very existence of many wetland sites. The poor quality of the water of the rivers themselves suggests that the mere provision of water will not reinstate benchmarks conditions. Research to date has focussed on the southern basin and understanding of the Basin as a whole would benefit from the extension of these approaches to other sections.

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Damming-induced hydrological alternation hastened ecological regime shift in the Yangtze floodplain lakes

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Abstract

Shallow lakes in the middle and lower reaches of the Yangtze River floodplain are one of the largest groups in the world, providing essential ecological services to local communities. Unfortunately, most of them have undergone a substantial ecological degradation in past decades. This kind of widely-occurred ecological regime shift, particularly after the 1950s in this region, has caused serious environmental disasters but limited knowledge existed on how the regime shifts occurred in those ecosystems.

Diatoms are good indicators to investigate the ecological response to the natural and human modified hydrological and nutrient regimes in these shallow Yangtze lakes. In this study, four lakes (Zhangdu Lake, Taibai Lake, Chaohu Lake and Chihu Lake) were selected from the region, to reconstruct the long-term (~200 years) environmental and ecological changes, using fossil diatoms and other multi-proxy (geochemistry and grain size) based on $^{210}Pb/^{137}Cs$ dating of sediment cores. Results revealed that distinct diatom assemblage shifts in all the four lakes occurred in 1950-60s, majorly from planktonic species (e.g. *Aulacoseria granulata*) to non-planktonic species (periphytic taxa). In the most recent decades, the nutrient-tolerant species (e.g. planktonic *Cyclostephanos dubius* and facultative planktonic *Nitzschia palea*) occurred and dominated in Chaohu and Taibai Lakes. Correspondingly, other proxies also exhibited parallel trends with diatom assemblages.

The ecological regime shift after the ~1950 may have attributed to the lake ontogeny and the hydrological changes which were caused by the damming and land reclamation. This is also revealed by a variance partitioning analysis (VPA). VPA analysis on, take Taibai Lake as an example, over different periods 1860-2006AD, 1860-1980AD, 1960-2006AD revealed a significant contribution from hydrological dynamics since 1860s, with a stronger nutrient impact after 1960. However, the most recent ecological regime shift from a clear, mesotrophic, macrophyte-dominated condition to a eutrophic, phytoplankton-turbid condition in both Chaohu and Taibai should be a result of the high nutrient input and cultivation of planktivorous fish, as well as industrial development in the catchment. This strong eutrophication process in some lakes suggests that the system regime has probably exceeded the tipping point, and shifted to a “bad” stable state.

Keywords

Shallow lake, Yangtze floodplain, diatom, ecological change, regime shift, hydrological condition, nutrient

1. Introduction

Shallow lakes (more than 600 lakes with area >1 km^2) in the middle and lower reaches of the Yangtze floodplain (MLY) are one of the largest groups in the world (Wang and Dou 1998). They are inherently dynamic and important sites for biogeochemical cycling, biological habitats as well as human resources. Unfortunately, most of them have undergone a substantial ecological degradation, characterised by turbid water with algae blooming, eutrophication and aquatic biodiversity loss in the past decades (Yang *et al.* 2010). Undoubtedly, the human activities (e.g. fishery, agriculture and industry) derived pollution loading, coincidentally with the recent global warming, should be the important reason blamed for this ecological cascading consequence.

Given their nature of shallowness and floodplain-located, those lakes may suffer from significant hydrological impacts, not only via the inner lake processes (such as water level change, residence time, turbulence) but also via the catchment runoff (e.g. nutrient loading, soil erosion). For example, along the MLY, there were thousands of dams built during 1950-1970s (Yang *et al.* 2010). Those dams, originally for the purpose of preventing flooding, altered the hydrological condition in the lake and cut off the hydrological connection between lakes and Yangtze River. By far, the ecological and environmental response to this event has still remained unknown, partly due to lacking in long-term monitoring data.

Palaeolimnology provides a robust technique to reconstruct historical environmental changes, and evaluate the ecological and environmental consequence of such kinds of hydrological alternations (Battarbee *et al.* 2005; Smol 2010). This study chose four lakes from the Yangtze floodplain (Zhangdu Lake, Taibai Lake, Chaohu Lake and Chihu Lake, see Table 1), to investigate the long-term (~200 years) environmental changes responding to altered connectivity with Yangtze River, using high-resolution multi-proxy analysis on ^{210}Pb/^{137}Cs dating, diatom, geochemistry, and grain size.

Table 1: Basic feature of the four lakes

	Taibai Lake	***Zhangdu Lake***	***Chaohu Lake***	***Chihu Lake***
Coordinates	29^{0}56'-30^{0}00'N 115^{0}46'- 115^{0}50'E	30^{0}37'-30^{0}42'N; 114^{0}40'- 114^{0}48'E	31^{0}25'-31^{0}43'N 117^{0}16'- 117^{0}51'E	29^{0}44'-29^{0}50'N 115^{0}37'- 115^{0}44'E
Lake area/km2	25.1	35.2	770	80.4
Average depth/m	3.2	1.2	3.0	2.8
Water quality (2010s)	eutrophic	Mesotrophic	eutrophic	Mesotrophic
The time of damming	1947	1954	1962	1963
Catchment area/km2	960.0	514.0	9258	360

2. Results and Discussion

Distinct diatom assemblage shifts in all three lakes were exhibited since the dam construction (Figures 1-4). Prior to the dam construction, all the lakes were dominated by planktonic species *Aulacoseria granulata*, *Cyclotella bodinica*, indicating a relatively high water lever but with strong turbulence which may derive from frequent water exchange between the lake and the Yangtze River. Since the dams were constructed, diatom communities shifted to non-planktonic species dominated such as *Gyrosigma acuminatum*, *Achnanthes minutissima* and *Navicula* sp., although with an exception of a rapid shift of eutrophic *Cyclostephanos dubius* occurred in Chaohu Lake. The different trajectories of diatom assemblages and changing rates may attribute to the lake ontogeny. The establishment of dams baffled hydrological connectivity between the lake and the Yangtze

River, and slowed down the water exchange, which can facilitate nutrient accumulation under the background of increasing catchment nutrient loadings. In smaller and shallow lakes (e.g. Zhangdu Lake, Taibai Lake and Chihu Lake), aquatic plants were well developed with the moderate nutrient enrichment. In contrast, Chaohu Lake became eutrophicated due to the high nutrient input from the rapid agriculture and industrial development in the catchment.

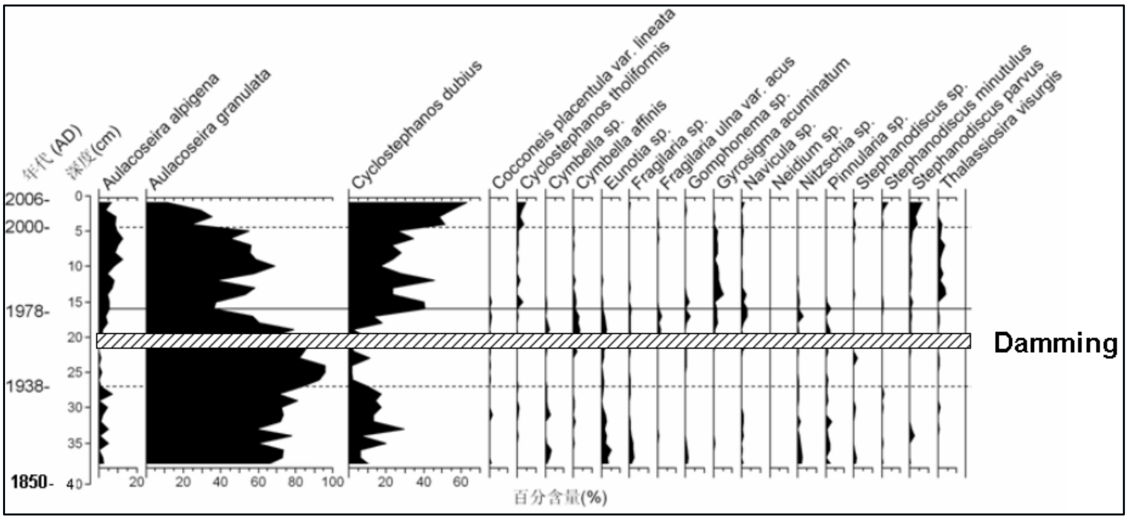

Figure 1: Diatom succession in Chaohu Lake over the past ~150 years. Only the major taxa are shown and the shadowed block indicated the event of dam construction.

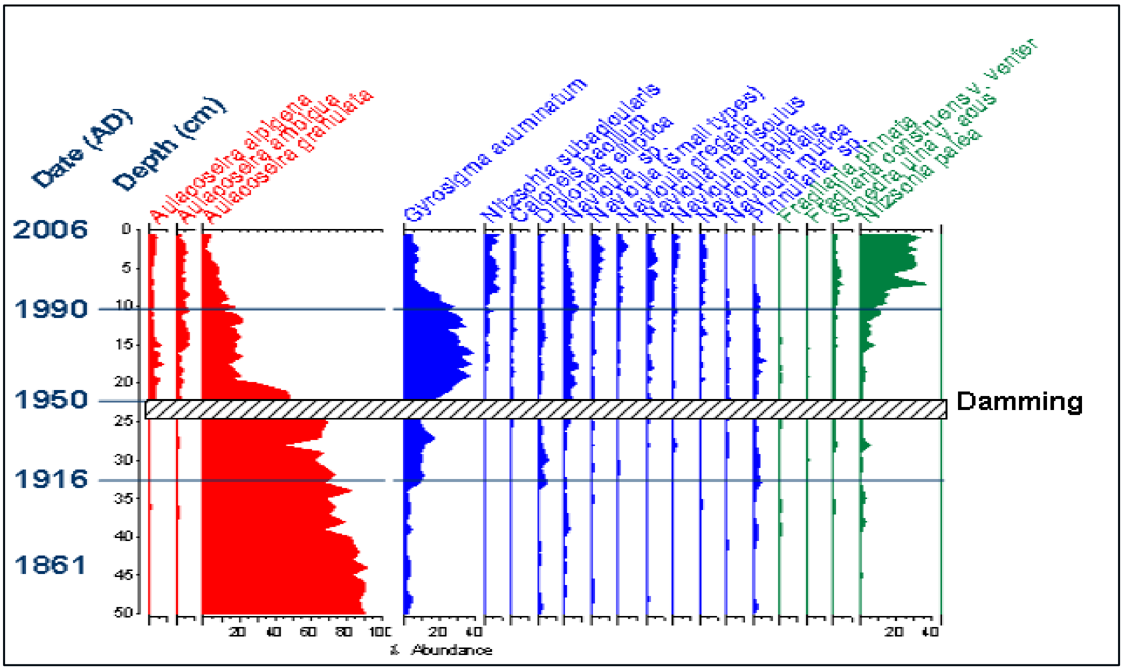

Figure 2: Diatom succession in Taibai Lake over the past ~150 years. Only the major taxa are shown and the shadowed block indicated the event of dam construction.

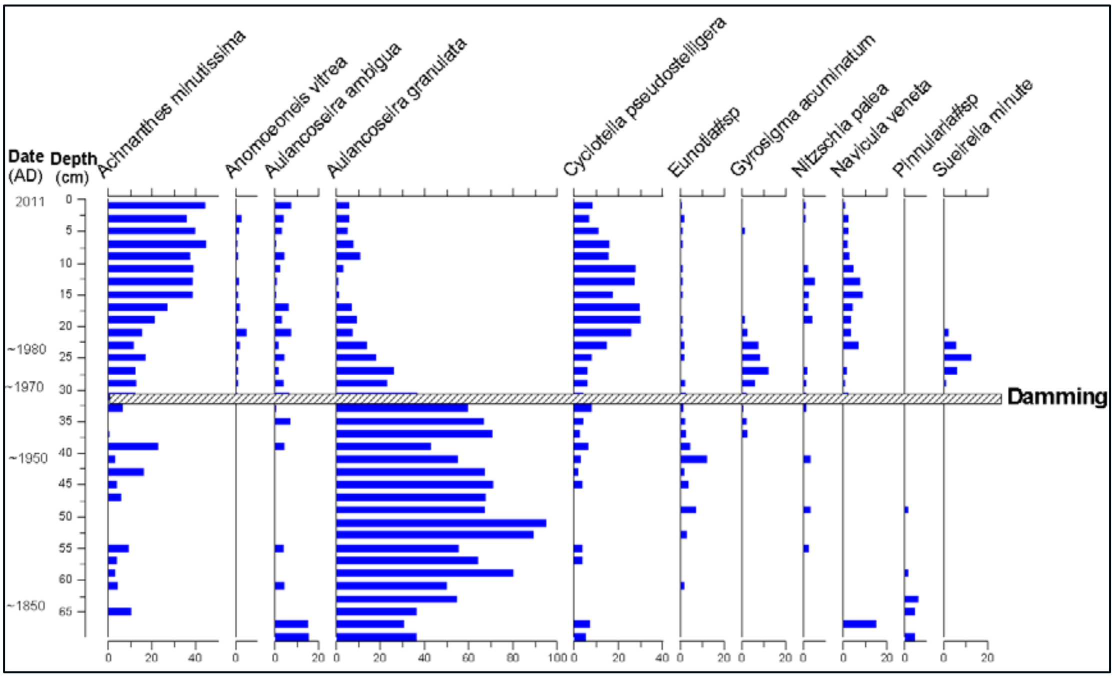

Figure 3: Diatom succession in Chihu Lake over the past ~150 years. Only the major taxa are shown and the shadowed block indicated the event of dam construction.

Given that the complex drivers (e.g. increasing nutrient loading, climate warming and hydrological alteration), they may coincidently have impacted on these aquatic ecosystems. As a result, a variance partitioning analysis (VPA) was conducted to explore the "pure" effect of hydrological condition on the ecosystems (Hall *et al.* 1999). VPA analysis on Taibai Lake over different periods 1860-2006AD, 1860-1980AD, 1960-2006AD revealed a significant contribution from hydrological dynamics since the 1860s, with a more stronger nutrient impact after 1960 (Figure 5). The VPA analysis on Chaohu Lake over periods post-1950, post-1970 and post-1990, also revealed a substantial contribution from hydrological dynamics before the 1970s (Figure 6 (see more details in (Chen *et al.* 2013)).

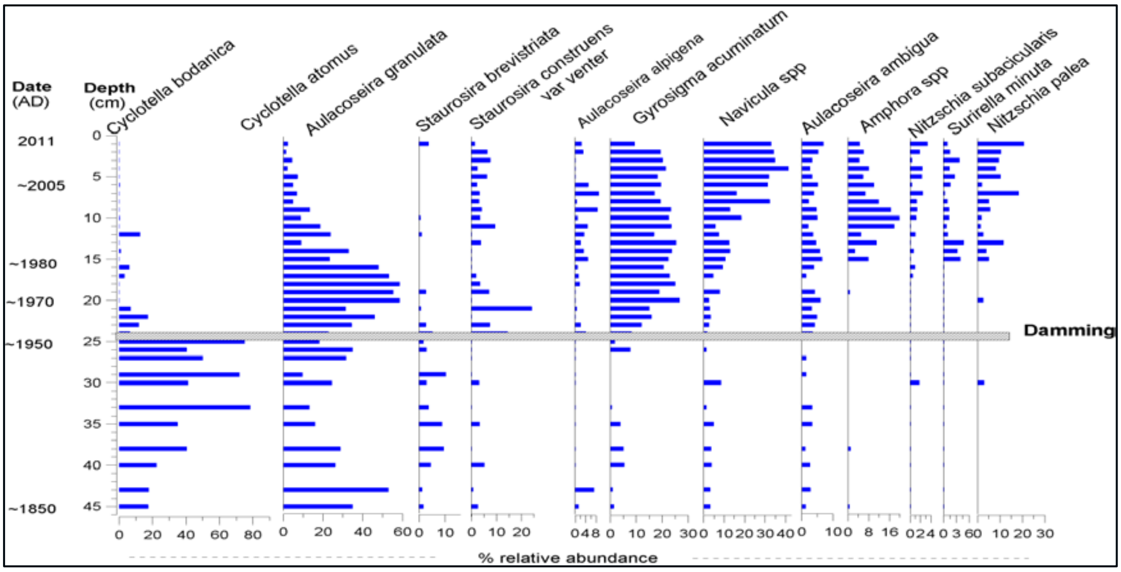

Figure 4: Diatom succession in Zhangdu Lake over the past ~150 years. Only the major taxa are shown and the shadowed block indicated the event of dam construction.

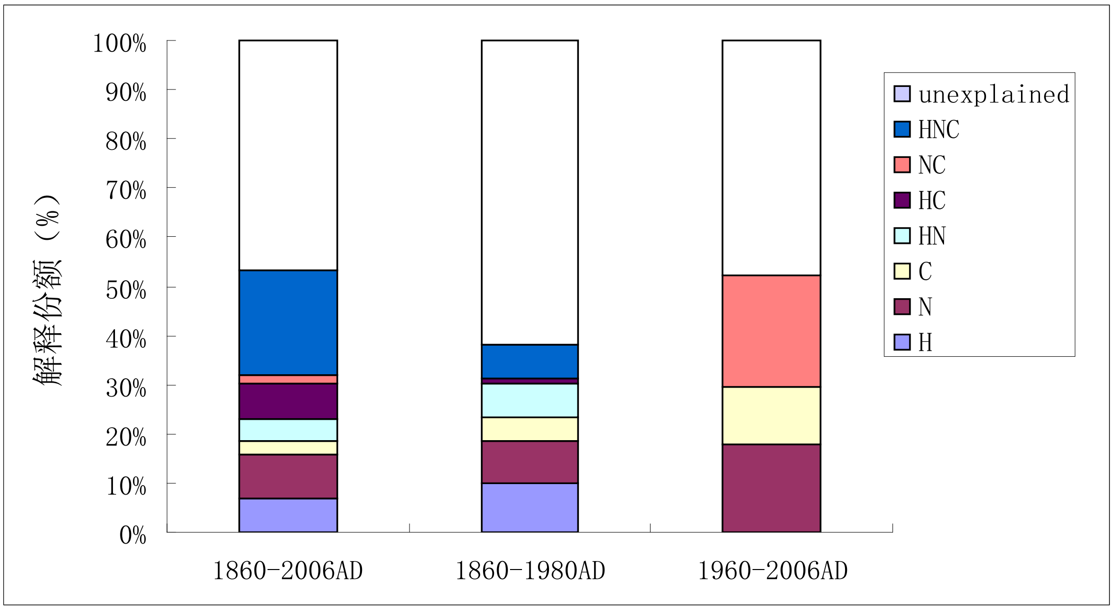

Figure 5: Effects of hydrology (H), nutrient (N), and climate (C) on fossil diatom assemblages from Taibai Lake determined using variation partition analyses with partial canonical ordination. Variation partition results for 1860-2006AD, 1860-1980AD, 1960-2006AD.

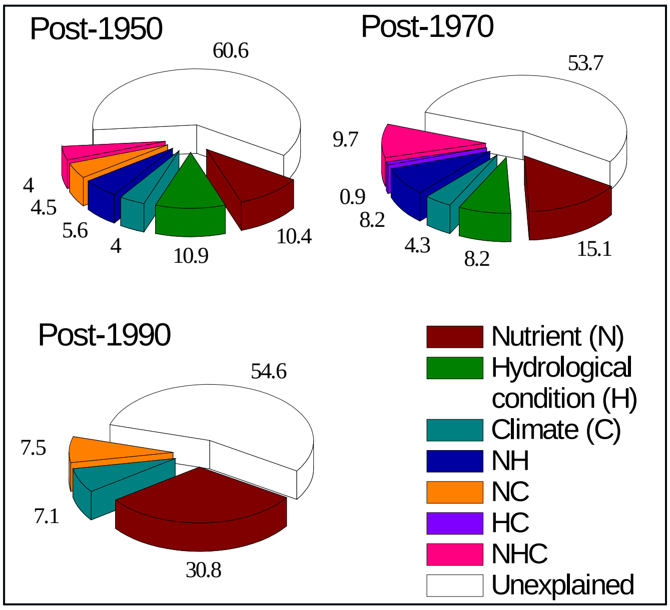

Figure 6: Effects of hydrology (H), nutrient (N), and climate (C) on fossil diatom assemblages from Taibai Lake determined using variation partition analyses with partial canonical ordination. Variation partition results for post-1950, post-1970, post-1990.

3. Conclusion

Long-term palaeolimnological records in the four Yangtze shallow lakes revealed that the altered hydrological conditions derived from dam construction played a key role in driving ecological change. With dam construction, nutrient accumulation was enhanced, thus triggered further ecological regime shifts, through changing the flow regime/lake retention time, nutrient dynamics and light climate.

Acknowledgements

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Cladoceran-inferred ecological and hydrological changes of two floodplain wetlands in two large river systems, the Murray (Australia) and Yangtze Rivers (China)

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Abstract

The landscapes of two of the world's large river basins, the Murray and Yangtze Rivers of Australia and China, have been intensively developed for the provision of food and water resources. Long term archives of change, reveal that man-made infrastructures in the river and catchment modifications for agricultural and industrial developments have reduced the resilience of wetlands ecosystem structure and functions in recent decades. The river regulations imposed during the 20th centuries in the Murray and Yangtze Rivers have transformed hydrology and ecology of the river and associated wetlands. High resolution, subfossil cladoceran assemblages retrieved from Kings Billabong and Zhangdu Lake of the Murray and Yangtze Rivers, have strongly responded to human disturbances in the past. Ratios of littoral to planktonic (L:P) assemblages of subfossil cladocerans and the number of ephippial remains in Kings Billabong indicated the shift in hydrology and ecology of Kings Billabong, and ecological stress as a result of changes in naturally occurring dry-wet cycles following river regulation (1927 AD). Similarly, the subfossil cladoceran assemblages and their ephippia in Zhangdu Lake also reflected the impacts of the construction of the Three Gorges Dam (1954) in the Yangtze River on hydrology and ecology of the wetland.

Keywords
Subfossil cladocerans, River Murray, Yangtze River, Kings Billabong, Zhangdu Lake, alternative stable states in ecosystem, ecological resilience

1. Introduction

The Murray and Yangtze Rivers of Australia and China are two of the world's large river systems, which make significant contributions to global ecosystem services including the provision of food and water resources to the people living in these river basins (Palmer et al. 2008). Over the past century, the water from the Murray and Yangtze Rivers have been heavily abstracted for hydroelectricity, agriculture and industrial developments causing significant disruptions in frequency, duration and amount of flows in the river and the river channels, connecting to many adjacent floodplain wetlands (Walker et al., 1995; Sun et al. 2012). The alterations of the natural flow regimes of the Murray and Yangtze Rivers have disrupted natural flood pulses and substantially modified hydraulic residence time, water level, nutrient fluxes and the species diversity, food webs, and the biological community structure and dynamics with the associated downstream floodplain wetlands (Kingsford, 2000; Chen et al., 2011). In the 1950s–1970s, extensive lakes and floodplains across the Yangtze River Basin were reclaimed as polders for agriculture and rural development. Consequently, the flood retention capacity was decreased, many lakes were disconnected from the main channel of the Yangtze by embankments and sluice gates, and eutrophication was rampant (Yu et al., 2011).

Disruption of the natural variability in connectivity and hydrological regime by excessive water abstraction or river-flow regulation, has threatened the ecological integrity including the biodiversity of the large river systems (Sheldon et al., 2002, Yang et al., 2006). Majority of floodplain wetlands of the River Murray and Yangtze over the past century have undergone a significant eutrophication as a result of the loss of submerged, littoral macrophytes (Reid et al. 2007, Yang et al., 2008). Multi-proxy responses, including the response of diatoms and physical and chemical records to ecosystem of Taibai Lake, show that the lake experienced initially mixed turbid and clear water state, with algal-macrophyte dominated regime following the 1950s alterations in the hydrological regime, land reclamation and building of reservoirs in China. However, after the 1990s the lake shifted to hyper-eutrophic due to dominant algae biomass and the loss of submerged macrophytes density (Liu et al. 2012). The wetlands of low water levels with dominant littoral vegetation in the Yangtze have experienced prevalent eutrophication following a large scale hydrological alteration during the 1950s (e.g. Qin et al., 2009).

Australia and China are facing increasing pressures in water and food securities, as well as rapid industrial growth in both Murray and Yangtze River basins. The use of palaeolimnological techniques can reveal long term changes in diversity and assemblages of biota in the past, as a result of disturbance in the river systems. Based on subfossil cladocerans, this paper will assess the responses of two floodplain wetlands, Kings Billabong (Australia), and Zhangdu Lake (China) of the River Murray and Yangtze River to human disturbances on the river systems during the 20th century.

2. Study areas

2.1 Kings Billabong (River Murray system)

Kings Billabong (34° 14' S & 142° 13' E) is located along the River Murray near Mildura (North West Victoria), Australia (Figure 1). Historically, Kings Billabong is a significant site of the Nyeri Nyeri aboriginal community, providing a rich source of goods and services. In its natural condition, the billabong was intermittently filled at times of high flows, and was exploited by the aboriginal community. However, since the early migrant settlement was intensified in the Victorian Mallee region, from 1891 to till 1923 (www.murrayriver.com.au), the landscape of the reserve changed substantially (MCMA, 2006). The natural flow of the River Murray has been significantly modified by the construction and operation of a series of locks, weirs and upstream storages (Gippel and Blackham, 2002). These changes have affected the hydrology, and, in particular, the variability, duration and frequency of flows in the River (Gippel and Blackham, 2002). Regulation was imposed on the River Murray in 1927 and 1937, when Locks 11 and 15 were built at Mildura and Euston, both in close proximity to Kings Billabong, disrupting the natural hydrology of Kings Billabong (Gippel and Blackham, 2002).

2.2 Zhangdu Lake (Yangtze River system)

Zhangdu Lake (30° 39' N & 114° 42' E), an associated floodplain wetland of the Yangtze River system, is located in Hubei Province, Central China (Figure 2). Historically, Zhangdu Lake would receive flood pulses from the Yangtze River during the high flows. However, as a result of the construction of dams and reclamations, Zhangdu Lake disconnected from the

Yangtze River in the 1950s. Reclamation and water conservancy construction in the 1970s further led to a change in its shape in the 1980s. Following the reclamation of 50 square meters, in 2005, Zhangdu Lake was seasonally reconnected with Yangtze River for restoration by World Wildlife Fund. Zhangdu Lake is located within Xinzhou District, Wuhan City, Hubei Province of Central China about 1km from the Yangtze River (Figure 2).

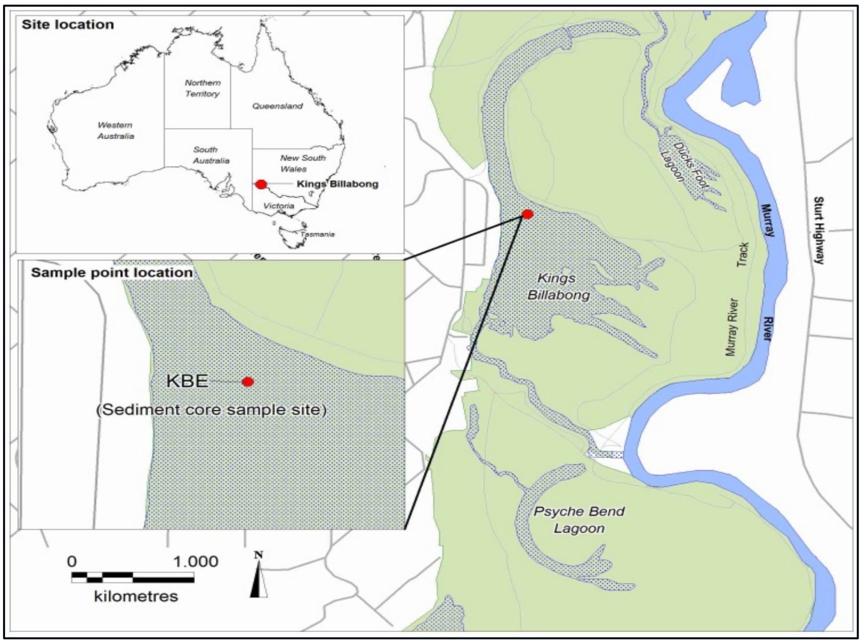

Figure 1: Kings Billabong of the River Murray system in Southeast Australia. KBE was the deepest point of the lake, where a sediment core for this study was taken.

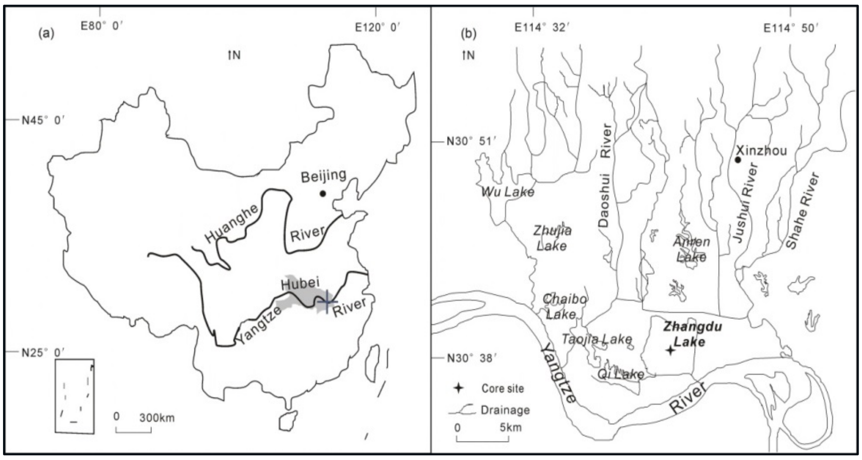

Figure 2: Zhangdu Lake of the Yangtze River in Hubei Province of China. Sampling points (+) shown is the deepest points of the lakes were where sediment cores were taken for this study.

3. Methods

Subfossil cladoceran remains retrieved from the lake sediment of Kings Billabong and Zhangdu Lake over the past century were analysed. For Kings Billabong, a high resolution subsampling was carried out in every 1cm interval of the 94cm long core. For Zhangdu Lake, the subsampling of 45cm long core was carried out in every 1cm interval for up to 27cm, and in every 2cm intervals for up to 45cm. Approximately 3-4g of wet sediment from each subsample from all lakes was treated with 100 ml of 10% KOH solution, and heated at 60°C on a hotplate for at least 45 minutes. The sub-sample mixture was then sieved through a 38 µm mesh. More than 200 identifiable cladoceran remains were counted at 400 x magnification. The dry weight percentage of each sediment sample was measured to calculate the counted portion of remains present per gram of dry sediment (Kattel et al., 2008). Cladoceran taxa were identified following (Frey, 1986; Shiel & Dickson, 1995; Szeroczyńska and Sarmaja-Korjonen, 2007). The 210Pb dating was carried out for all sites, where the radionuclide activity was detected at 51cm for Kings Billabong and 45cm for Zhangdu Lake. Constrained incremental sums of squares (CONISS) analysis were performed in TILIA. The chord-distance dissimilarity coefficients of samples were used for stratigraphically-constrained quantitative zonations of the samples (Grimm, 1987). Indirect ordination techniques, such as Detrended Correspondence Analysis (DCA) were also performed for assessing nature of species alignment in the ordination (Ter Braak, 1995).

4. Results

4.1 Kings Billabong

The subfossil assemblage of cladocerans in Kings Billabong showed four distinct changes in ecosystem. Until the 1890s (Zone I) littoral cladoceran, Dunhevedia crassa, Alona guttata, Chydorus sphaericus and Graptoleberis testudinaria dominated. During this period the abundance of planktonic, Bosmina meridionalis was low (Figure 3). In the 1890s-1950s (Zone II), total littoral cladocerans began to decline, but small littoral species such Alona guttata became abundant. The abundance of total planktonic cladocerans was influenced by higher abundance of planktonic B. meridionalis and some Daphnia records during the 1950s-1970s, coinciding with the timing of the 1956 flood in the River Murray (Zone III) (Figure 3). Although total littoral cladocerans declined, 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 sphaerics dominated, while the littoral D. crassa declined significantly. Mean time a high density of resting eggs was also recorded in the sediment (Figure 3).

4.2 Zhangdu Lake

The subfossil assemblage of cladocerans in Zhangdu Lake showed three distinct changes in ecosystem. During the period c. 1880s-1960s (Zone I), the total planktonic abundance was recorded high, which was mainly dominated by planktonic *Bosmina* sp. The density of littoral cladocerans was very low except a few, characteristics of both littoral and planktonic habitats dwelling species, *Chydorus sphaericus* (Figure 4). However, in the c. 1960s-1980s (Zone II), following the construction of the Three Gorges Dam in the Yangtze (c. 1954), the abundance of the total littoral cladoceran began to increase with presence of some of the common species of cladocerans such as *Acroperus harpae*, *Alona guttata*, *Alona rectangula*, *Chydorus sphaericus*, *Graptoleberis testudinaria* and *Sida crystallina*. During the c. 1990s-2000s (Zone III), the trend of the abundance of littoral cladoceran species was increasingly positive. The dominant species during this period was *Acroperus harpae*, *Alona intermedia*, *Alona affinis*, *Kurzia lattissima*, *Leydigia leydigi*, *Alona guttata*, *Camptocercus rectirostris* and *Disparalona rostrata*. The concentration of the cladoceran resting eggs was also recorded high during this time (Figure 5).

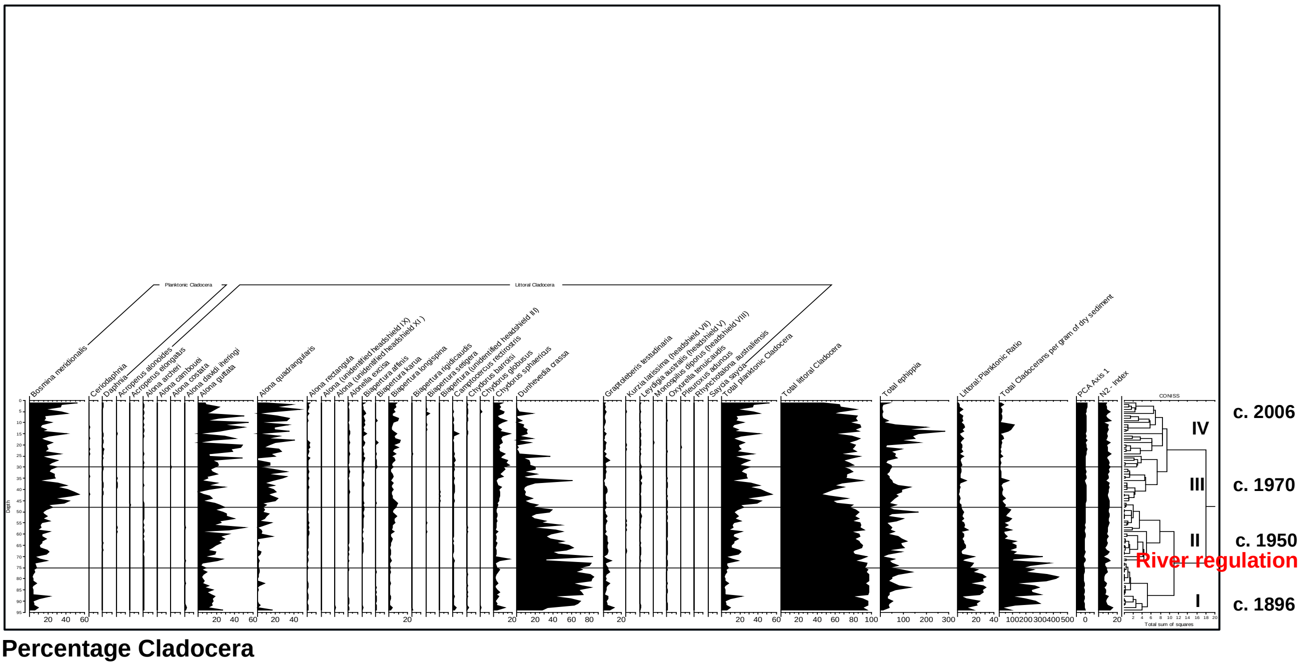

Figure 4: Percentage composition and N2 diversity index of subfossil cladocerans, their response to past hydrological change in the River Murray system.

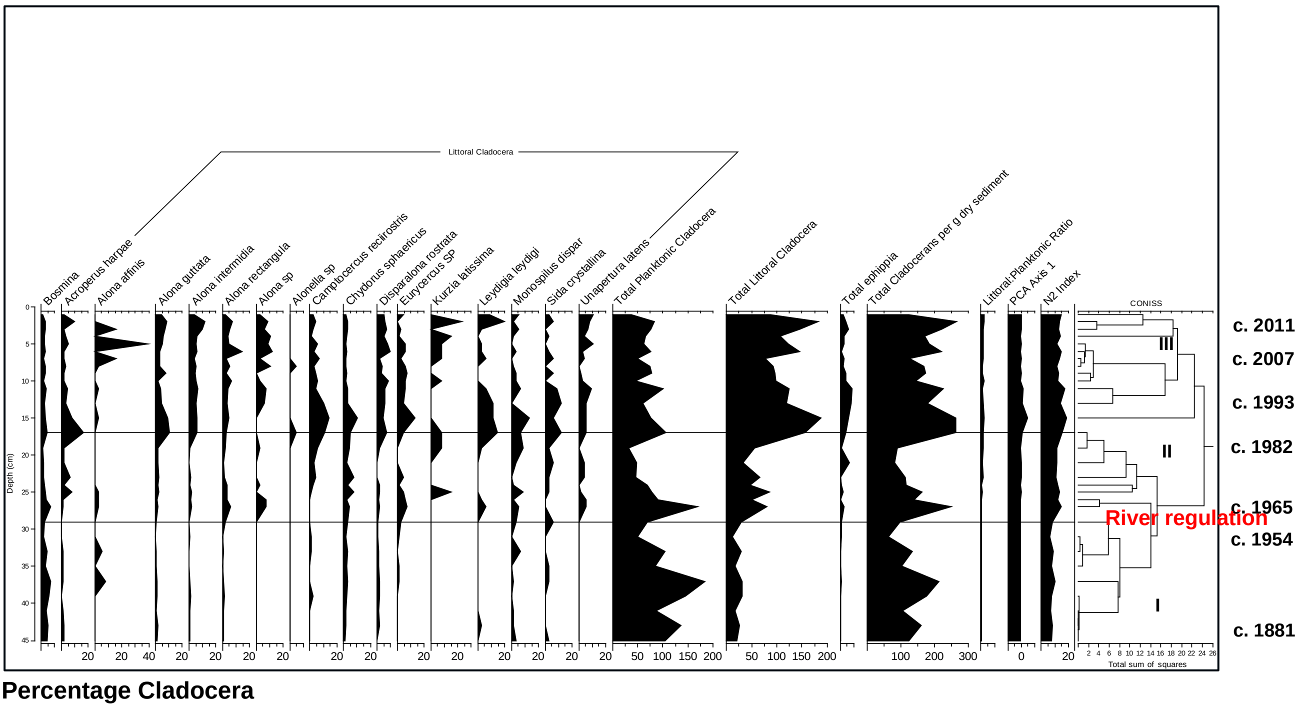

Figure 5: Composition (%) and N2 index of subfossil cladocerans in Zhangdu Lake, and their response to past hydrological change in the Yangtze.

5. Discussion

Cladoceran assemblages of Kings Billabong and Zhangdu Lake both have strongly responded to human-induced hydrological alterations in the Murray and Yangtze Rivers over the past century. Long standing turbid water in Kings Billabong may have triggered the condition limited growth of submerged vegetation, as it used to be occurring in shallower environment with dry-wet cycles prior to river regulation. By the early 2000s, planktonic *B. meridionalis*, and littoral *A. guttata* and *Biapertura longispina* dominated the wetland complex. The conditions during the post-river regulation period reflects 'stress' among the cladoceran community as revealed by a high density of resting eggs recovered from the sediment (e.g. Nevalainen et al. 2011). A rapid rate of decline of submerged littoral preferring species *D. crassa* from Kings Billabong following river regulation reflects that the magnitude of the impact of river regulation was high on this species. Unlike Kings Billabong, Zhangdu Lake responded to downstream water shortages in the river canal connecting to the lake, following the construction of the Three Gorges Dam in 1954. The condition of decreasing water volume led to a significant loss of lentic habitat. The decline in lake depths and the loss of lentic habitats led to extreme ecological stress among the cladoceran community in c. 1990s-2000s as revealed by a high number of resting eggs recorded in the sediment. Sarmaja-Korjonen (2003) reported that wetlands experiencing a transition of climates (e.g. Pleistocene – early Holocene), strong predator-prey interaction (e.g. fish) and increased land use change in the catchment (e.g. release of chemicals) had high number of cladoceran ephippia in sediment.

Kings Billabong and Zhangdu Lake have shown a tendency of regime shifts in ecosystems. In Kings Billabong, presence of aquatic vegetation complexity prior to 1900 AD had increased species richness and water quality, while the open water habitat characteristic to longer flood duration in recent decade has led to increased turbidity and richness of filter-feeding zooplankton taxa (e.g. *Bosmina*). Van den Brink et al. (1994) reported that characteristic floodplain wetlands with short annual flood duration were dominated by well-developed aquatic vegetation, and organisms associated to these macrophytes such as Bacillariophyceae and scraping zooplankton taxa associated with aquatic macrophytes. In Zhangdu Lake, irrespective to increased littoral substrata due to low water level, following the construction of Three Gorges Dam, the lake experienced a sustained eutrophication, and was also indicated by presence of small mud dwelling cladoceran species such as *Alona* and *Leydigia leydigi* indicative of polluted condition (Hofmann 1996).

6. Conclusions

Evidence of subfossil assemblage of cladocerans from Kings Billabong and Zhangdu Lake over the past century suggests that the human-induced river regulation in Murray River (Australia) and Yangtze River (China) have altered hydrology and ecology of these large wetlands. The hydrological and ecological processes may be further exacerbated by other driving forces, including agricultural and industrial developments and rapid climate warming over the past 30 years. The deteriorating conditions of wetlands following the large scale impacts on the river systems have shown a tendency of regime shift in ecosystems. Strong restoration measures are needed to improve the ecosystems of these wetlands through management of water both quantity and quality, in Australia and China's river basins.

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Section 3: Resilience and regime shifts

Understanding the impacts of multiple stressors and associated regime shifts in shallow wetlands

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Abstract

Understanding the dynamics and drivers of regime change is essential for effective wetland management. Much evidence suggests that nutrient-enriched, shallow, permanent lakes and wetlands typically exist in either two alternative stable states or regimes: a clear-water state dominated by macroscopic plants or a turbid-water state dominated by microscopic phytoplankton. In European lakes, where phosphorus is often limiting, macroscopic plants typically dominate when total phosphorus (TP) is less than 50 µg L^{-1} and phytoplankton dominate when total phosphorus exceeds 150 µg L^{-1}. Predicting which state will dominate between these two thresholds is more difficult because feedback mechanisms hinder macroscopic plants invading a phytoplankton-dominated system and vice versa. Hysteresis occurs because there is not a simple linear relationship between nutrient concentration and the abundance of phytoplankton or macroscopic plants. Non-linear dynamics prevail and regime change can only occur when nutrient thresholds and associated feedback mechanisms are overcome. Although nutrient-driven state changes are well documented, other state changes can be driven by water regime, salinity and organic matter loadings. Research on wetlands in south-western Australia indicated that a multi-state model was applicable to perennial salinised wetlands where salinity, rather than nutrient concentration, was the main water quality driver. The finding that a dual state model did not apply to Western Australian wetlands with a seasonal water regime indicated that water regime is also influential. Developing conceptual models of regime change provides a powerful tool for integrating data on physical, chemical and biological features of standing waters into concepts that can generate testable predictions and guide restoration activities.

Keywords
Resilience, multiple stressors, regime shifts, shallow wetlands

1. Introduction

The world's wetlands are threatened by multiple stressors, including land use change, water extraction, infrastructure development, pollution and invasive species (MEA, 2005). They are also affected by two major indirect drivers: global population growth and increasing economic development. The expansion and intensification of agriculture to feed an ever-increasing world population has changed the quantity and quality of water sustaining aquatic ecosystems. Unfortunately, the trade-offs between increased agricultural production (and human well-being) and the decline in aquatic ecosystems are complex and often lack simple or easy solutions (Gordon et al., 2008). Historically, changes in land use and hydrology associated with the production of food, fibres and fuels, have resulted in multiple 'bottom up' impacts on aquatic ecosystems. However, the 'top down' impacts of global climate change are likely to be the dominant direct drivers of biodiversity loss and ecosystem change by the end of this century (MEA, 2005).

The development and testing of conceptual ecological models that examine the impact of multiple (top-down and bottom-up) stressors on aquatic ecosystems, and recognise that responses may be nonlinear, is an essential activity for identifying critical processes and predicting changes, particularly the possibility of ‘ecological surprises’ or ‘catastrophic regime shifts’ (Scheffer et al., 2001; Folke et al., 2004; Mayer & Rietkerk, 2004; Gordon et al., 2008).

This paper describes the conceptual models of ecological change and regime shifts associated with altered hydrological processes and eutrophication, salinisation and acidification that have occurred in shallow aquatic ecosystems (wetlands, shallow lakes and river pools) in urban and agricultural landscapes in south-western Australia. Extensive land clearing for agriculture, and, to a lesser extent, urban development, combined with a poor understanding of local and regional environmental processes, has resulted in broad scale and major impacts on terrestrial and aquatic ecosystems within the region. This has provided the opportunity to examine the ecological responses of shallow aquatic ecosystems to multiple stressors and to develop insights that might be useful for wetland management and restoration where these impacts occur in other regions of the world.

2. Conceptual Models of Ecological Change

Models describing gradual ecological change and threshold-initiated regime shifts, based on precipitation–vegetation interactions in agricultural zones, were developed by Gordon et al. (2008). In these models, agricultural modification of water flows resulted in ecological regime shifts that operated across a range of spatial and temporal scales. Ecological dynamics were defined both by internal dynamics (such as vegetation growth) and external forces such as precipitation and drought. Regime shifts occurred when external forces or gradual internal changes altered a system from one set of mutually reinforcing processes to another. General versions of these models are presented in Figure 1. The simplest model of ecological change conforms to a linear relationship between an ecosystem state (or regime) and an environmental (external) stressor. This model describes gradual change and implies that the transition from a ‘desirable’ or ‘undisturbed’ state A, to the degraded state B, can be prevented by reducing the external driver. Similarly, restoration of the system from state B back to state A can be achieved by reversing the external driver.

The threshold model describes a non-linear response to a change in the external driver. An abrupt change, or regime shift, occurs from A to B over a relatively small change in the external driver. This model implies that reducing the external stressor to below the threshold value will restore the system from B back to A. In the hysteresis model the threshold of change differs between the pathway of ‘collapse’ from A to B and the pathway of ‘recovery’ from B to A. To return to A, the external driver must be reduced to a threshold below that of the initial threshold causing the collapse from A to B. These pathways provide a visual description of the alternative stable states model of Scheffer (1989, 1998) and Moss (1990) for shallow European lakes under-going eutrophication.

Ecosystems are maintained within a regime through internal dynamics (feedback mechanisms). External disturbances or changes in internal relationships, or both, can trigger a regime shift. Non-linear behaviour does not always indicate the presence of alternative dynamic regimes but forms part of a set of conditions which include the presence of positive internal feedback mechanisms (Scheffer & Carpenter 2003). Ecosystem collapse occurs when an irreversible threshold is passed (Figure 1).

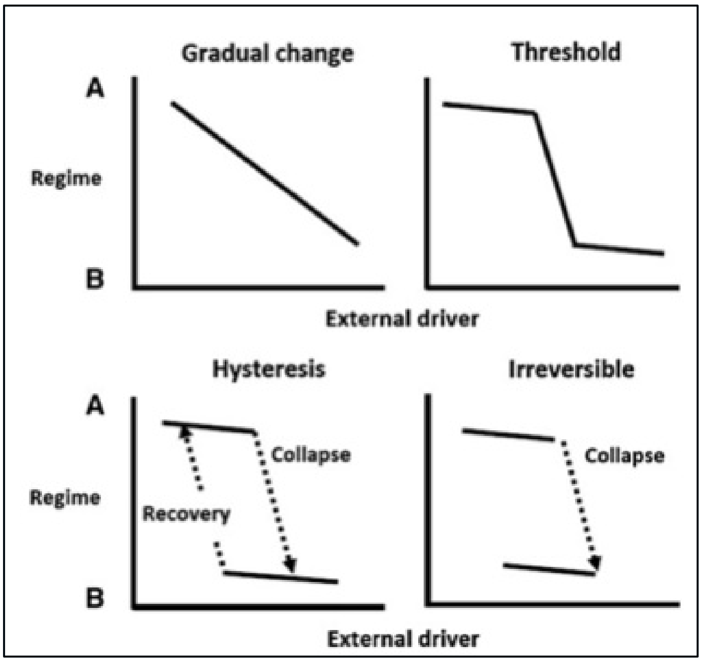

Figure 1: Conceptual models of ecological change. The simplest model (upper left) is of gradual or linear change, which is often reversible. The threshold model (upper right) describes more abrupt change at a defined value of the environmental factor of concern. The hysteresis model (lower left) also includes thresholds but the value of the threshold differs between the processes of degradation and restoration. The final model (lower right) includes a threshold that once passed results in ecological collapse and irreversible change. These models are based on Gordon et al. 2008 and Finlayson et al. (2013).

3. Eutrophication and alternative states

The best known model of ecosystem dynamics for shallow lakes and wetlands describes the existence of two 'alternative stable states driven by eutrophication (Moss 1990; Scheffer 1990). The state variables are the abundance of submerged macrophytes and phytoplankton, and the controlling variable (driver) is nutrient loading (facilitated by changes in the light regime). This model was developed in Europe by researchers studying permanent, eutrophic shallow ecosystems (Moss 1990; Scheffer 1990) and the success of Northern Hemisphere restoration projects based on this theory (e.g. Coops and Doef 1996; Moss et al. 1996) has initiated great interest from restoration ecologists and natural resource managers elsewhere in the world. The challenge for researchers examining ecosystem dynamics globally has been to apply existing models to different types of wetlands and to determine whether the controlling variables and responses to perturbation of these systems are the same.

The concept of two contrasting ecological regimes occurring in shallow lakes affected by eutrophication was extended by Scheffer and van Nes (2007). They found that nutrient-enriched shallow lakes can be dominated by free-floating plants, submerged charophytes, submerged angiosperms, green algae, or filamentous cyanobacteria at different points along a gradient of eutrophication. Gradual species replacements may occur across levels of a controlling variable that are interrupted at critical points by more dramatic shifts between ecological regimes, as exemplified by the clear water/turbid water contrast. In accordance with Scheffer and van Nes (2007), there is evidence from Australian studies that within these five regimes there may be different dominant taxonomic groups, such as charophytes or angiosperms in submerged macrophyte-dominated systems and cyanobacteria or chlorophytes in phytoplankton-dominated systems. The implications of this for restoration are that different levels or thresholds of a controlling variable may be necessary to cause a shift in dominance, depending on the characteristics of the dominant taxa.

4. Multiple stressors and regime shifts

Australian wetlands are often degraded by large, landscape-scale processes (e.g. land clearing that have resulted in hydrological changes and associated impacts, such as anthropogenic salinization, acidification, and sedimentation) often in addition to nutrient enrichment. Controlling variables identified in Australian wetlands include an increase in nutrients, an increase in salinity, and hydrological changes (lower water levels). Transitions to new ecological regimes include not only a turbid, phytoplankton-dominated ecological regime but also a turbid, sediment-dominated regime and a regime dominated by clear water overlying a benthic microbial mat.

Changes in hydrological processes have been identified as having a major influence on the development and expression of eutrophication, salinisation and acidification in urban and agricultural landscapes (Davis et al., 2010). The change from a seasonal to permanent hydrological regime or vice versa, appears to be very important in determining the type of ecological regime shift that might occur.

Many systems experiencing hydrological change and anthropogenic salinisation in the Western Australian wheatbelt were found to move between a saline, macrophyte-dominated regime and a hypersaline, benthic microbial mat-dominated regime (Davis et al., 2010). Experimental investigation of germination responses in glasshouse trials undertaken by Sim et al. (2006a) revealed a negative linear response of the dominant macrophyte, *Ruppia* sp., to increasing salinity, with a threshold of 45 g L^{-1}, beyond which no germination occurred. However the impact of salinisation was confounded by the change from a seasonal to permanent hydrological regime. Seasonally, drying systems appeared to conform to a threshold model with the presence of a macrophyte-dominated regime controlled by salinity alone. Benthic microbial communities dominated under hypersaline conditions and macrophytes seldom occurred where salinities remained above 45 g L^{-1}. In contrast, perennial systems followed a hysteresis model where either regime could potentially be present at relatively low salinities. Experimental trials by Sim et al. (2006b) revealed that the feedback mechanisms maintaining the submerged macrophyte regime appeared to be stronger and more resilient than those maintaining the benthic microbial community-dominated regime within the intermediate salinity range considered. Submerged macrophytes also appeared to be stronger competitors showing almost complete dominance in all treatments <45 g L^{-1}, even if already dominated by benthic microbial communities or (algal or bacterial) plankton blooms.

5. Implications for wetland restoration

Models of conceptual change and alternative stable states theory can inform the development of thresholds to guide management and restoration of wetlands. Although many factors can influence aquatic plant growth, in the absence of other limitations, the availability of nutrients is often the prime driver for changes in plant dominance. Where nitrogen is not limiting, a submerged macrophyte-dominated regime is unlikely to occur at phosphorus concentrations greater than 0.05 mgL^{-1} TP. In contrast, phytoplankton dominance always occurs above 0.15 mgL^{-1} TP (Jeppesen et al. 1990), suggesting that these concentrations are the benchmarks for hysteresis between submerged macrophyte and phytoplankton dominance.

Although non-linear models of ecosystem dynamics (including threshold dynamics, alternative stable states, and slow/fast cycles) may provide a more valuable tool for the management and restoration of Australian wetlands than gradual continuous or stochastic models, it must also be noted that the former models have not been strongly validated because the complexity of study ecosystems makes it difficult to undertake replicated studies. Although these models have great value for management, the replication and time required to gather conclusive evidence may be prohibitive. Describing ecosystem dynamics in temporary wetlands is further complicated by the need to incorporate seasonal changes (related to both hydrology and annual plant life cycles) and other aspects of climatic variability.

The main value of conceptual ecological models for restoration and management is that they provide a framework explaining how wetland ecosystems function in relation to a number of controlling variables. Real progress has been made in the identification of both nutrient and salinity thresholds between ecological regimes for Australian wetlands. These thresholds can guide targets both for management and to inform the potential success of restoration efforts under the current nutrient or salinity regime.

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Resilience of floodplain vegetation to hydrological and climate change

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Abstract

Variable and unpredictable hydrological disturbances are a prominent feature of river-floodplain ecosystems, especially in drylands. Vegetation in these habitats typically exhibits a high degree of resilience to both floods and droughts, conferred by a range of traits at individual plant, population and community levels. Changes to hydrological disturbance regimes, resulting from anthropogenic activities and climate change have the potential to exceed the limits of such resilience mechanisms and thereby transform the composition, function and identity of these systems. Knowledge of vegetation resilience and its limits in floodplain ecosystems is therefore critical for effective decision making regarding their conservation and natural resources management. Here, I synthesise recent research concerning the resilience of vegetation in floodplains to flooding and drought. Key mechanisms of resilience operating at the level of individual plants, populations and communities are identified. Potential limits to resilience mechanisms, as well as the factors influencing these, are also explored including the capacity of mechanisms of resilience to hydrological disturbances to confer resilience to other disturbances (e.g. warming). Finally, the adaptability and transformability of floodplain vegetation are discussed, especially with respect to autonomous and planned adaptation to climate change.

Keywords
Resilience, floodplain vegetation, hydrology, climate change

1. Introduction

The definitive feature of floodplain ecosystems is variability in the presence of surface water as a result of overbank flooding. Although floodplain hydrological regimes vary widely between regions, according to climate, and locality, in relation to topography, all floodplain ecosystems, by definition, are subject to periods of inundation and drying. Organisms that inhabit floodplains therefore persist, in one way or another, despite relatively high levels of environmental variability that are driven largely by hydrology. Floodplain plants are often perceived as being particularly resilient to hydrological disturbances and exhibit a diverse array of traits that enable their persistence in the face of hydrological variability and other common disturbances (e.g. fire, grazing etc.; Brock et al. 2006).

Land use change, water resources development and climate change have significantly modified hydrological regimes of floodplains around the world and floodplain ecosystems have been degraded as a result (Tockner and Stanford 2002). In many cases, the limits of resilience and adaptive capacity of floodplain plant species and vegetation are likely to have been exceeded. With further dramatic changes to floodplains and their hydrologic regimes projected over the coming century, understanding the resilience and adaptive capacity of floodplain plants and vegetation is critical for managing the conservation of biodiversity as well as the protection of important ecosystem goods and services (Capon et al. 2013). An awareness of the potential limits of vegetation resilience is likely to be particularly useful for guiding management efforts.

2. Resilience traits of floodplain plants and vegetation

Floodplain plants display a wide range of traits that facilitate resilience to hydrological disturbances across different hierarchical levels of biological organisation (Table 1). Amongst individual plants, resilience traits typically provide either resistance to disturbance or the ability to recover following a disturbance. Traits that facilitate flood or drought tolerance can be considered as resistance traits to these specific disturbances while plastic traits, such as heterophylly or the ability to elongate shoots, roots or petioles in response to inundation, can be perceived as mechanisms of resistance to fluctuating water levels. Recovery to hydrological disturbances of individual floodplain plants is facilitated by traits such as the capacity to resprout or rapid growth responses to favourable conditions.

Floodplain plants that do not resist or recover from hydrological disturbances as individuals typically have life history traits that enable persistence through hydrologic changes at a population level (Table 1). For the most part, these traits involve a diversity of responses to environmental conditions within particular life history stages of the population (e.g. only a certain portion of soil seed banks germinating in response to single germination events (Brock et al. 2006)).

At the level of plant communities, resilient vegetation is often associated with high levels of diversity in terms of both species responses to environmental conditions as well as functional redundancy (Table 1). Dynamic vegetation communities that are able to respond quickly to hydrological changes through shifts in composition are also often considered to be resilient, e.g. desert floodplains (Brock et al. 2006). Facilitative relationships amongst plant species may further contribute to increased resilience of plant communities to some disturbances (e.g. nurse plants can enable understorey herbs to survive for longer during periods of drought (Flores & Jurado 2003)). At a landscape scale, patch mosaics comprising different vegetation community types, as well as different hydroseral stages of particular vegetation types, may serve to increase resilience to disturbances of vegscapes (Table 1) since such patterning reduces the risk that all plant communities will be similarly affected. The presence of refuge habitats within the landscape can similarly enhance long-term vegetation resilience (James et al. 2013). Connectivity is a somewhat debatable issue since greater connectivity may promote the spread of invasive species. For the most part, however, connectivity amongst communities at a landscape scale is perceived as promoting resilience since species are able to disperse and persist in favourable patches, including refuges, while disturbances occur in other patches.

Table 1: Examples of resilience traits exhibited by floodplain plants at different levels of organisation

Individual plants	***Plant populations***	***Vegetation communities***	***Vegscapes***
Resistance traits:	Dormant propagule banks	Response diversity	Heterogeneity, e.g. refuges, patch mosaics
Flood tolerance	Variable germination (bet hedging)	Redundancy	Connectivity

Drought tolerance	Iteroparity / polycarpy	Dynamic	
Tolerance of fluctuating water levels, e.g. heterophylly, shoot/root/petiole extension	Multiple reproductive strategies	Facilitation	
Recovery traits:	Multiple dispersal vectors		
Rapid response (e.g. growth)	Genetic variation (or constancy)		
Resprouting			

3. Limits of resilience

Resilience mechanisms amongst floodplain plants and plant communities vary regionally according to the type of hydrologic (and other) disturbance regimes that occur. In general, a shift is apparent in floodplain vegetation, with increasing hydrologic variability and unpredictability from communities comprised mainly of species that are persistent and tolerant at an individual level, to dynamic assemblages comprising a greater proportion of species relying on reliant on life history strategies. In the very unpredictable floodplains of inland Australia, for example, few persistent strategic species (i.e. trees) are present. Instead, floodplain vegetation comprises a high proportion of seed bank ephemerals and highly dynamic and heterogeneous communities and vegscapes (Brock et al. 2005). In contrast, predictable, regular floodplain wetlands are often dominated by aquatic and semi-aquatic species that reproduce predominantly via asexual mechanisms (Cronk and Fennessy 2001). Such patterns indicate limits to particular mechanisms of resilience between and within floodplain systems in relation to the specific disturbance regimes faced.

Many of the traits mentioned above with respect to hydrological disturbance can also facilitate resilience to other disturbance (e.g. fire) and are therefore likely to contribute to vegetation resilience in the face of climate change (Table 2; Capon et al. 2013). The rapid pace, broad extent and non-discriminatory character of climate change, however, may exceed the limits of these resilience mechanisms to cope with disturbances (Visser 2008). The interactive effects of climate change and other stressors (e.g. river regulation) are likely to further hamper the capacity of plants and plant communities to resist and recover from disturbances. Weak points in mechanisms of resilience can be detected at each level of biological organisation and these are likely to be the elements most susceptible to climate change impacts (Table 2).

Table 2: Examples of weak points associated with key resilience traits across different levels of biological organisation. Potential contribution of resilience to general resilience is also indicated.

Resilience trait	***Contributes to general resilience?***	***Weak points***
Individual plants:		
Stress tolerance	no	changes to disturbance regimes

		competition from exploiters
Plasticity	possibly	changes to triggers response times
Plant populations:		
Propagule banks	yes	reservoirs
Spatial diversity	possibly	refuges, connectivity
Plant communities and vegscapes:		
Facilitation	possibly	keystone / nurse species
Patchiness	possibly	loss of asynchrony

4. Implications for management

Water resources development and land use change around the world have significantly altered hydrologic disturbance regimes under which floodplain plants have evolved and vegetation communities developed. Typically, these changes have culminated in reduced flow variability, often leading to shifts in vegetation composition including encroachment by invasive perennial species with high levels of individual resilience. In contrast, climate change is projected to result in increased hydrological variability and unpredictability through the intensification of hydrological cycles (e.g. more rainfall in high intensity events). Perennial, stress-tolerant species adapted to specific disturbance regimes may therefore by most vulnerable to climate change while species with resilient life history strategies may be favoured.

Given differences in the types and degree of resilience across hierarchical levels, it is crucial that planning, management and monitoring of floodplain vegetation consider multiple scales. Management efforts may be guided by an understanding of weak points of resilience mechanisms at each scale (Table 2). Interventions that protect, restore or enhance weak points relative to management goals could be prioritised. For instance, at population and species levels, high value stress tolerant plants (e.g. dominant, long-lived trees) might be key targets for management efforts while keystone species or nurse plants might be targeted at a community level. Protection, restoration or enhancement of refuges and reservoirs, as well as their connectivity, will be particularly important at community and landscape levels. Resilient vegscapes may be further promoted through management interventions that maintain heterogeneity and asynchrony of the disturbance-drive impacts.

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Resilience in aquatic ecosystems: developing predictive models to explain the effects of anthropogenic stressors on Murray-Darling Basin billabongs

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Abstract

Freshwater ecosystems are among the most threatened in the world (MEA 2005) and have been identified as one of the ten Australian ecosystems most vulnerable to tipping points. The floodplain lakes and wetlands (billabongs) of the Murray-Darling Basin (MDB) are hotspots of productivity and diversity and provide important breeding, feeding and refuge habitat for a range of floodplain river biota, as well as important ecosystem goods and services by way of flood mitigation, nutrient cycling and sediment trapping. Nonetheless, MDB billabongs are threatened by water resource and agricultural development and climate change. In recognition of these threats, water dependent ecosystems of the MDB are currently subject to expensive and controversial management measures involving water buy backs estimated to cost up to $30 billion and the subsequent delivery of environmental water. The need to understand the critical drivers of change and the internal system interactions that underlie ecosystem responses in floodplain river ecosystems has never been greater. This project will develop ecosystem response models that will not only identify the critical threatening drivers, but also provide the guidance necessary to rehabilitate these important ecosystems.

Keywords

Resilience, aquatic ecosystem modelling southeast Australia, River Murray, billabongs, Murray Darling Basin

1. Introduction

Critical to our capacity to understand billabong responses to water resource and agricultural developments, is the paucity of long term records of ecosystem condition which would provide important benchmarking information as well as an understanding of the natural variability of the systems. The long-term ecological histories that can be obtained via palaeoecology have clear potential to aid understanding and future management of billabongs. Palaeoecological records from MDB billabongs, largely focused on records of change in diatom and cladoceran communities (Ogden 2000, Tibby et al. 2003, Gell et al. 2005a, Reid et al. 2007, et al. 2013), suggest dramatic ecological degradation over the last century in response to land use and hydrological changes following European settlement after around 1850. Arguably the most ecologically significant basin-wide change in the MDB has been the loss of aquatic macrophytes and subsequent increase in phytoplankton

(Ogden 2000, Gell et al. 2005b, Reid et al. 2007, Reid 2008). This change appears to have occurred rapidly in response to reduced photic depth likely caused by system-wide increases in sediment flux associated with catchment soil erosion following initial landscape disturbance (Ogden 2000, Prosser et al. 2001, Reid et al. 2007). The shift from macrophyte to phytoplankton has the potential to affect biodiversity (Scheffer 2004, Declerck et al. 2005, Davidson et al. 2013) and trophic structure, not only of billabongs themselves, but also of the riverine landscape more broadly as billabongs are centres of productivity that provide important feeding and breeding habitat for riverine biota and are centres of productivity, that affect the entire floodplain river system (Hillman 1986, Bunn and Boon 1993, Shiel et al. 1998).

Importantly, not all MDB billabong records show the same clear pattern of macrophyte loss. For example, macrophyte loss is not evident in the Ovens (Ogden 2000) or Goulburn Rivers (Reid 2002). These patterns do not correlate with patterns of land use intensity, so it has been suggested that macrophyte loss depends on the underlying resilience of billabongs to state switches rather than variation in the intensity of stressors (Ogden 2000). In particular, Ogden (2000) argued that the larger, deeper billabongs of the Murray River are more susceptible to macrophyte loss because they are closer to a threshold in photic depth (ie photic depth = mean depth) than the smaller, shallower Ovens billabongs (where photic depth > mean depth). Ogden's (2000) hypothesis has important implications for both our understanding of how these important freshwater ecosystems function, but also how degraded and/or threatened billabongs may be most efficiently restored and managed. However, the suite of studies cited above are focused largely on one or two sites (Gell et al. 2005b, Reid et al. 2007, Fluin et al. 2010); therefore, while they provide insight on the extent of macrophyte loss, the studies do not collectively represent a systematic, robust study design capable of fully testing this critical hypothesis. Billions of dollars are being invested in water buybacks in the MDB; the benefits of this investment are likely to be limited if water clarity-trophic state relationships are not addressed and more fully understood.

2. Methods

A synthesis of diatom sediment records from 17 billabongs on the Murray, Murrumbidgee and Goulburn Rivers (Figure 1) was carried out to explore patterns in the response of billabong ecosystems to anthropogenic stressors over the past two centuries. All but one of these records (Sinclair's Flat) extend well beyond the last two centuries, with several extending more than a thousand years. Most records, therefore, cover the critical pre- to post-European settlement boundary and also provide information on the natural variability of systems. Because the principal interest of this research is on changes to the trophic structure of billabongs in relation to pelagic vs littoral productivity, the synthesis compares the relative abundances of preserved diatoms grouped according to habitat preference. The major habitat groups in billabongs are planktonic, epipelic (living on mud surfaces) and epiphytic (living attached to plants or other hard surfaces). A further group consists of a suite of genera with poorly defined habitat preferences that are typically associated with variable or frequently disturbed environments (collectively referred to hereafter as 'small *Fragilaria*') (Sayer 2001, Reid and Ogden 2009)

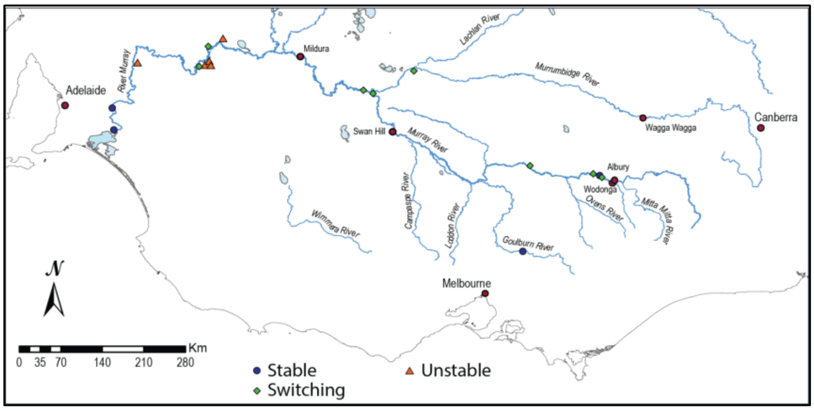

Figure 1: Locations of billabongs included in the preliminary synthesis. Symbols indicate the response type inferred from diatom assemblages

3. Results

An ordination of the diatom assemblages of all samples (sediment depths) and all sites is shown in Figure 2. The position of samples in ordination space reflects the relative abundances of the major habitat groups in each sample, as illustrated by the bubble plots of the full 17 billabong data set (Figure 2). These show samples plotting to the lower left typically have abundant planktonic diatoms (Figure 2a), those to the lower right typically have abundant epiphytic diatoms (Figure 2c), while those plotting to the top have abundant small *Fragilaria* (Figure 2d).

The ordinations also reveal patterns in the diatom assemblages of billabongs and how these assemblages vary over time. Importantly, the patterns of temporal variation in the billabong diatom records are similar within classes, giving rise to the following nomenclature and characterisation:

- Stable – records are dominated by epiphytic diatoms and vary little over time (e.g. Callemondah 1 in Figure 4).
- Switching – records are dominated by epiphytic or planktonic diatoms. While some switching billabongs appear to have undergone periods of planktonic dominance prior to European settlement, the planktonic state predominates in the post-European phase and the most recent samples always contain >55% planktonic diatoms (e.g. Hogans 1 in Figure 4).
- Unstable – assemblages vary greatly through time and contain relatively high abundances of 'small *Fragilaria'* (e.g. Tanyaka in Figure 4).

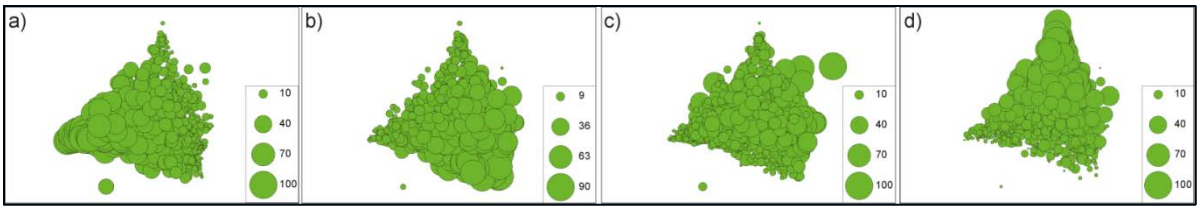

Figure 2: MDS bubble plots of the full 17-billabong data set based on a resemblance matrix of Bray-Curtis similarity measures using relative abundances of diatoms grouped according to preferred habitat (planktonic, epiphytic, epipelic, small *Fragilaria*). Bubble size indicates the relative abundance of planktonic (a), epipelic (b), epiphytic (c) and small Fragilaria (d) diatoms.

Explanatory analyses of the data using both multivariate regression trees and random forests confirm that the simple typology explains more of the variation in the data separation based on pre- and post- European settlement (era). Regression tree analysis showed that the typology explained 14% of the variation in the species level diatom data whereas era explained 8 %. A further classification of era and type combined explained 25% if the variance. Random forest analysis showed that the error in classification was lower for typology than for era, confirming typology as a better descriptor of diatom assemblage than era.

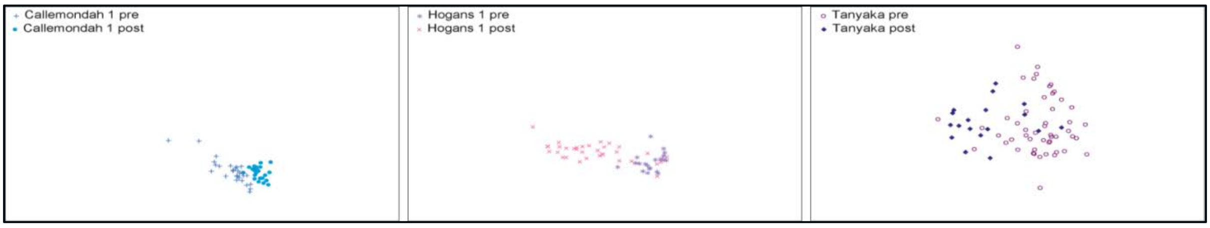

Figure 4: MDS ordination plots of diatom assemblages from three billabong sediment records. The three records are examples of 'stable' (Callemondah 1), 'switching' (Hogans 1) and 'unstable' (Tanyaka) billabong response types.

There is also a spatial pattern in the distribution of the aforementioned response types (Figure 1):

Stable billabongs are found in the upper middle Murray, the Goulburn and in the lower Murray below the Murray gorge. As noted above, Ogden (2000) also found that Ovens River billabongs are relatively stable.

Switching billabongs are found in the mid-Murray (Hume Dam to Darling confluence) and lower Murrumbidgee. Within this group, phases of pre-European plankton dominance are apparent further downstream (from around the Murrumbidgee confluence), but not upstream.

Unstable billabongs are confined to the Murray from the Darling confluence to the Murray Gorge.

4. Discussion

The observed response types, and the spatial patterns in those types, support Ogden's (2000) hypothesis that billabong size and depth are important factors controlling the likelihood of plant loss. Thus, the smaller channels of the Murray tributaries create smaller and shallower billabongs that have a propensity to be stable because reductions in photic depth do not result in substantial portions of the bed being removed from the photic zone (Figure 5a). In contrast, the larger, deeper billabongs of the mid-Murray are more susceptible to state changes because similar photic depth reductions result in substantial portions of the bed being removed from the photic zone (Figure 5b). The responses observed, however, do suggest a need to modify this hypothesis. Most notably, the unstable response type suggests that not all billabongs naturally establish metastable states of macrophyte or phytoplankton dominance. Similarly, the evidence that some 'switching' billabongs apparently underwent periods of phytoplankton dominance prior to European settlement highlights the potential that mechanisms other than anthropogenic sediment pulses may trigger macrophyte loss. A range of drivers have been suggested as causes of

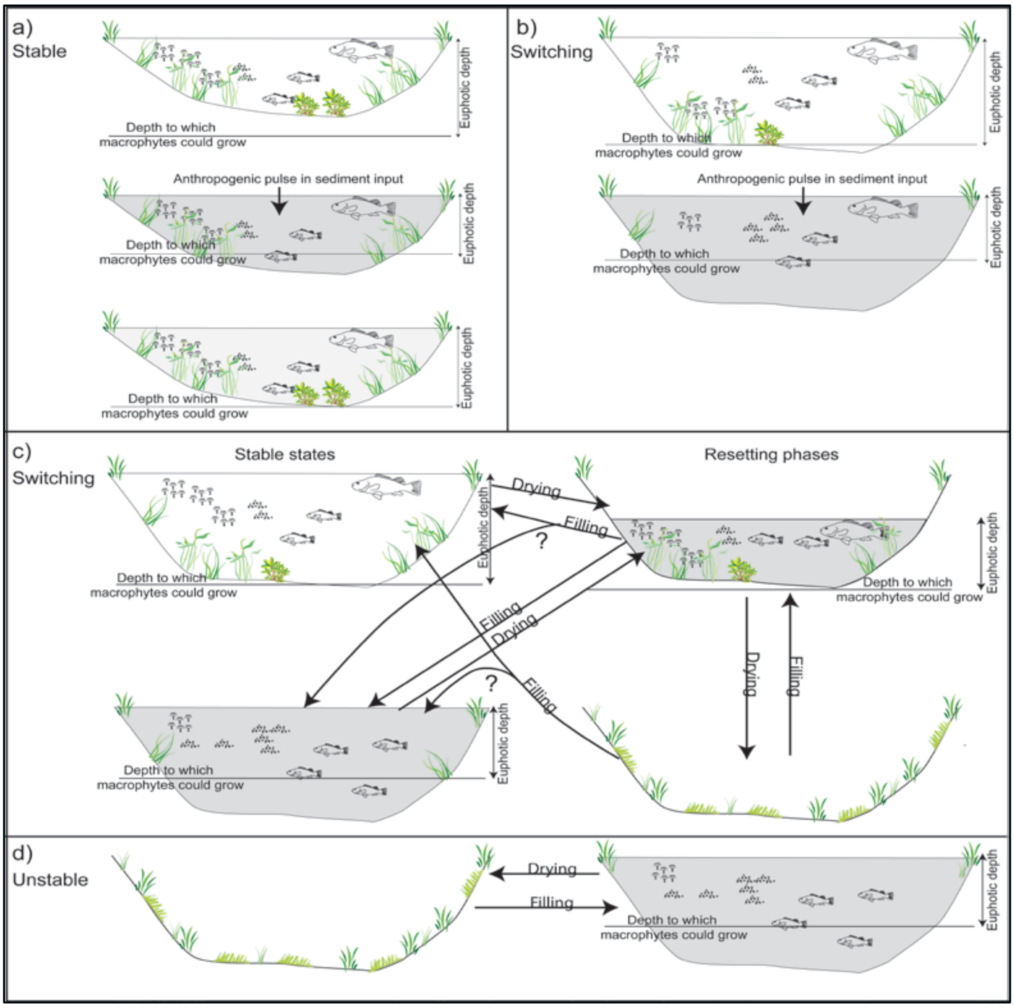

Figure 5: Conceptual models explaining the relationships between billabong geomorphology, hydrology and the proposed response types. Stable billabongs are resilient to reduced photic depth because pulses in anthropogenic sediment input do not reduce photic depth enough to remove the majority of the bed from the photic zone (a). Switching billabongs are less resilient to reduced photic depth because pulses in sediment input result in the removal of the majority of the bed from the photic zone (b). In both cases, feedback processes may act to strengthen the original or new state once sediment influx is reduced. Switching billabongs may also be reset to either state by drying events (c). Unstable billabongs dry frequently and so fail to develop stable states (d).

macrophyte loss in shallow lakes across the world, including eutrophication, storms, and top-down controls of zooplankton grazers and water fowl feeding activities (Mitchell 1989, Moss & Leah 1982, Scheffer et al. 1993, Jones & Sayer 2003). While such mechanisms may play a role in billabongs, the clustering of billabongs exhibiting periods of pre-European phytoplankton dominance in the Murray downstream of the Murrumbidgee confluence, and the clustering of unstable billabongs further downstream, suggests hydrological variability may be a factor. These reaches are subject to more variable hydrology and higher rates of evaporation, both of which are likely to result in more variable billabong water levels. Accordingly, we propose that hydrological variability may serve to modify the dichotomy of response types that emerge from the Ogden (2000) hypothesis and give rise to the response types illustrated in Figures 5c and 5d. Downstream of the Murrumbidgee, the drier climate and more variable hydrology introduces the potential for more frequent and extreme drying events (Figure 5c). These drying events may act to return substantial portions of the bed to the photic zone, allowing recolonisation by macrophytes. Alternatively, complete drying may also precipitate loss of macrophytes, which may or may not return following refilling. Whether macrophytes become established may depend on a suite of factors, such as the timing of filling, the duration of the dry phase, or stochastic factors such as colonisation by fish. Finally, for unstable billabongs, drying phases may be too frequent for strictly aquatic communities (planktonic or benthic) to establish, so these systems are dominated by opportunist taxa such as amphibious or water tolerant terrestrial plants and algae adapted to benthic and pelagic habitats (Figure 5d).

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Palaeolimnological evidence for resilience lose under nature perturbations in a lake ecosystem

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Abstract

Critical transitions in natural system could produce surprising changes, and sometimes results in unacceptable outcomes. Recently, the studies on critical transitions have become interests among scientists around the globe. However, until recently, most of these studies are mostly based on laboratory, or manipulated by field experiments. Although the related theories of critical transitions are improving, in reality, the verifications of such transitions through tests in natural systems are rare. One of the main reasons is due to a lack of long term historical data available for this study. Palaeolimnological approach can produce long term data, however some biases, such as sediment compaction and taphonomy, can still prevent a robust conclusion. In this study, we have chosen a mountain lake, Lugu, in Yunnan (China). Lugu Lake represents the historical records of diatom-inferred environmental change over the past 30 ka. The subfossil sedimentary diatom assemblage in this lake shows two significant tipping points at around 15 ka and 1 ka BP respectively. We have then aggregated the diatoms community data in order to get an even-time sequence dataset to calculate early warning signals (EWS) in the vicinity of these tipping points. We found that the recovery rates of diatoms before these tipping points were significantly declined indicating a slowing down of the ecosystem of the Lugu Lake. Our results suggest that the long term high resolution palaeolimnological data can reduce the biases in EWS reconstructions, and may show a critical slowing down phenomenon in ecosystem before being collapsed.

Keywords

Critical transition, palaeolimnology, resilience, lake ecosystems, Lugu Lake, China

1. Introduction

Critical transitions in natural system could produce surprising changes, and sometimes result in unacceptable outcomes (Scheffer et al., 2001). Recently the studies on critical transitions have become interests among scientists around the globe (Barnosky et al., 2012; Carpenter et al., 2009; Nicholson et al., 2009). However, until recently, most of these studies are focused mostly on laboratory based, or manipulated by field experiments (Carpenter et al., 2011; van Nes & Scheffer, 2005). Although the related theories of critical transitions are improving, in reality, the verifications of such transitions through tests in natural systems are rare (Wang et al., 2012). One of the main reasons is due to a lack of long term historical data available for this study. Palaeolimnological approach can produce long term data, however some biases, such as sediment compaction (Carstensen et al., 2013), can still prevent a robust conclusion. In this study, we have chosen one of the mountain lakes in Yunnan, China, Lugu Lake. In order to get an even temporal sequence, we have aggregated the samples, and the aggregated data was then used to calculate the generic early warning signals before tipping points.

2. Study area

Lugu Lake is located in the edge of Qianghai-Xizang plateau (south-west China), and lies in the boundary of Sichuan and Yunnan provinces (Figure 1). It is located in sub-tropical area, with altitude of 2685 m a.s.l. The lake is fresh water with the maximum depth of 94 m, and mean depth ~40 m. The surface area is 50.5 km2 with a catchment area of 171.2 km2. The lake is oligo-mesotrophic (SD: 15 m-11 m, TP=28 ug L-1) due to seldom human activities in the catchment. A sediment core with 9.7m long was retrieved from the deepest part of the lake in 2011 using a Kullenberg Uwitech Coring Platform System. Radiocarbon ages of the core were determined by AMS at the Poznan Radiocarbon Laboratory of Poland. The core represents the historical records over past 30 ka in Lugu Lake.

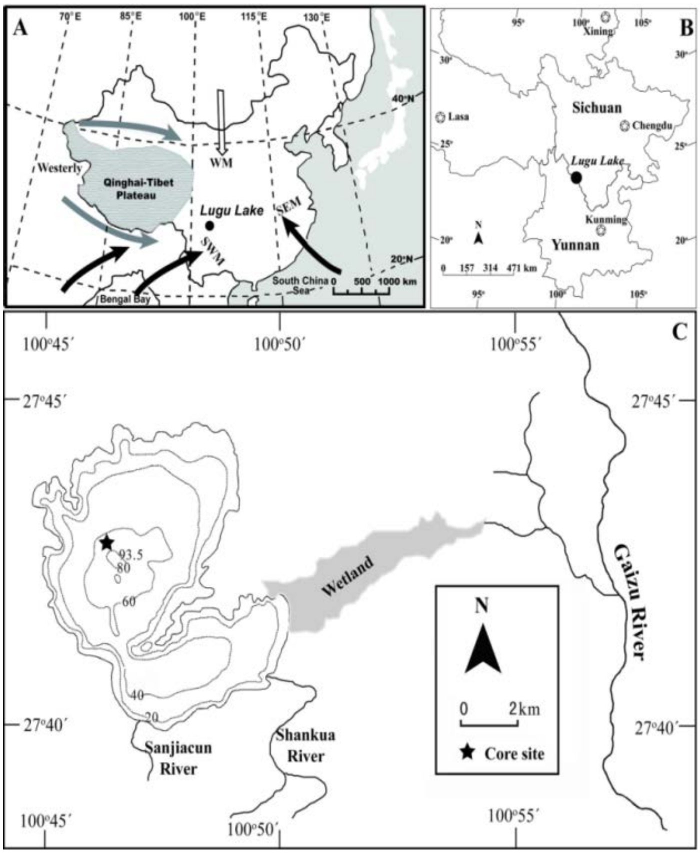

Figure 1: The location of Lugu Lake

3. Methods

Fossil diatoms slides were prepared according to the standard procedures. Two proxies, i.e. diatom abundance and DCA axis 1 scores were employed in this study for the purpose of tipping point detections and early warning signals calculation. Tipping points were examined with an excel add in (http://www.beringclimate.noaa.gov/regimes/), and the generic EWS were measured following the method by Dakos et al. (2012) and calculated in R with 'early-warnings' package (Dakos et al., 2012). The samples were aggregated to get a temporal even sequence to reduce the biases of lake sediment compaction before EWS calculation.

4. Results

The diatoms were mainly dominated by planktonic species such as Cyclotella rhomboideo-elliptica during the period 3ka- 2.5ka. Then, the diatoms communities were dominated by non-planktonic species such as Staurosira sp. and Achnanthes sp. until 1.5 ka. After 1.5 ka, Cyclostephanos dubius and Cyclotella sp. were bloomed and became the dominated species, with small Fragilaira sp. decreased constantly until recently. The statistical analysis revealed that the subfossil diatom assemblage in this lake showed two significant tipping points at around 15 ka and 1 ka BP respectively (Figure 2). Abrupt change could be found in both diatom abundance data and DCA data at the two times. Therefore, the generic EWS were discussed for the two states in Figure 2.

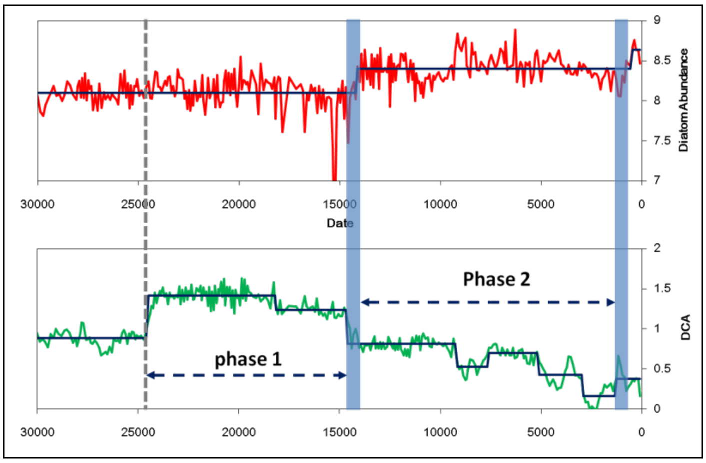

Figure 2: Abrupt changes in the history of Lugu Lake presented by both diatom abundance (above panel) and communities (below panel)

The results of EWS calculations are presented in Figure 3. In phase 1, the autocorrelation of diatom communities was increasing in the vicinity of tipping point, and recovery rate of ecosystem after disturbance began slowing down. The standard deviation was increased at the beginning but declined afterward. In phase 2, the autocorrelation was increased with a Kendal tau value of 0.794, and the recovery rate was deceasing with a Kendal tau value of -0.794. The standard deviation was also rising at phase 2, and the Kendall tau was 0.743 which indicated a significant rising trend in the variance of diatom communities. In summary, in both phases, the recovery rates declined while the ecosystem was approaching to alternative states. Sensitivity tests to Gaussian Kernal smoothing bandwidth and the rolling windows size were then done to identify the effects of different parameters on the results. The tests suggested that the above results were not affected by a wide range of bandwidths and windows sizes. The autocorrelation and recovery rate showed same patterns under different parameters which indicated that the results were robust. The same method was then used in diatom concentration data, and generic EWS were calculated. The results did not show any patterns, and both autocorrelation and variance seemed to change randomly.

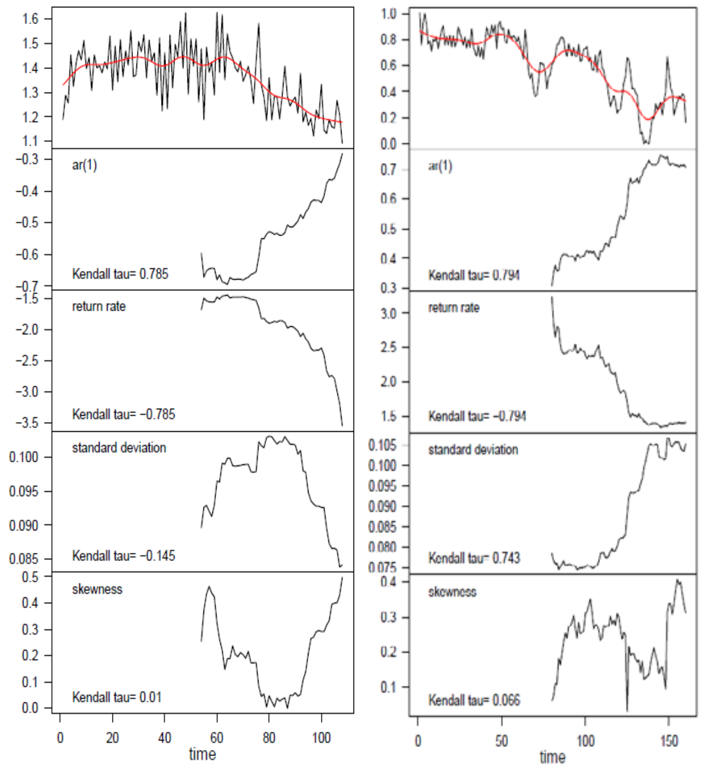

Figure 3: Generic early warning signals in Lugu Lake. The left panel represents EWS in phase 1 and the right panel represents EWS in phase 2.

5. Discussion and conclusions

It is obvious that the abrupt change and its related research became one of the hot topics. Scientists are worrying that global ecosystems may be exposed to catastrophic change in the near future. There is no clear mechanism to understand what is behind such a catastrophic shift. The ideas of finding generic early warning signals are thought to useful. Scientists have made a lot of achievements in understanding the mechanisms of change. However, this has not yet been proven to be robust. It is reasonable that we should rigorously probe the secrets of abrupt change, with right methods, using proper proxies. Previous models suggested that the palaeo-data was defective due to sediment compaction (Carstensen et al., 2013). In this study we aggregated the sediment data to avoid the effect of compaction, and we found that aggregated palaeolimnological proxies could be used for abrupt change studies. We suggest palaeolimnologists should pay more attention to the sediment compaction. The aggregated proxies would then be a good choice to make the results robust.

We found that the recovery rates of diatom communities before both tipping points were significantly declined, indicating the slowing down of the ecosystem recovery of Lugu Lake. The phenomenon of critical slowing down in phase 2 was obvious, as both autocorrelation and variance were rising. In phase 1, rising autocorrelation and decline of the recovery rate indicated slowing down of the ecosystem, but variance did not show any critical slowing down property. However, the autocorrelation in phase 1 was negative, and variance was rising first, and then declined afterwards. Therefore, the fluctuations of diatoms communities during this period may have been highly sensitive to external drivers, and higher resolution data was needed to get matched patterns in autocorrelation and variance. However, the recovery rates in both phases declined, suggesting that the critical slowing down can be used to guide early warning signals detection in Lugu Lake, which has little human disturbance in the study period.

We also calculated generic early warning signals with diatom abundance data in Lugu Lake, but the results did not show any pattern of early warning signals. The autocorrelation and variance were both randomly fluctuating. It suggests that critical transition is the structure change of the system which could be caused by feedbacks (Scheffer et al. 2001). Therefore, a proxy which can represent the target's structure, rather than volume or quantity, should be excavated for EWS detection. In this study, the DCA score is a proxy which can reflect the diatom community change, and diatom abundance just represents the concentration of diatoms in the ecosystem. It is not surprising that the latter cannot show any EWS. This study also suggests that not all proxies from an ecosystem can be applicable for EWS detection.

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Section 4: Knowledge transfer and use of management approaches

The Practice of the Danube River Basin Management in Europe and knowledge transfer to a management of the Huai River in China

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Abstract

To prevent water pollution and improve the living environment along the Huai River, plenty of measures have been taken by the government, however, conspicuous effect hasn't been reached. So it is necessary to summarize the work that has been done in the river basin abroad, where basin water ecosystem and water quality have improved greatly. In this article, the Danube River in Europe is chosen and its basin management practices are summarised into 7 aspects: legal framework, management agency, cooperation mechanism, management planning, monitoring networks, wetland conservation, and public participation. Through the above analysis, suggestions to Huai River are given. It is suggested that the river basin management departments of government should publish a more comprehensive policy about the water pollution control, and improve public participation mechanism.

Keywords
Danube, Huai River, river basin management

1. Introduction

Huai River is located in the east of China. About 200 million people live in the watershed of Huai River, so the water quality of Huai River is important to people living in the catchment. In 1995, a serious water pollution incident happened and attracted great attention from the government. This incident further led to the water quality protection and pollution prevention program in the Huai River. Over the past twenty years, the water quality of the Huai River is still not good enough and there is a need to learn from the lessons of work done to prevent and control water pollution of the Danube River, whose water ecosystem ranged from poor in the late 20th century to relatively balanced quality nowadays.

2. Study area

The Danube River is located in the centre area of Europe, and flows through 14 countries, which are Germany, Austria, Czech Republic, Slovakia, Hungary, Slovenia, Bosnia and Herzegovina, Serbia, Montenegro, Romania, Bulgaria, Rep. of Moldova, and Ukraine. It is 2857km long and drains an area 801,463 km^2. The population is about 80.5 million (Popovici, 2011).

3. Pollution prevention in the Danube

The Danube River has its official river basin management agency, which is ICPDR, short for International Commission for the Protection of the Danube River. ICPR is founded in 1994 whose members include all countries of the Danube River with catchment is greater than 2,000km^2.

4. Results

The major experience of the work in the Danube is as follows.

4. 1 Legal framework

The foundation of International Commission for the Protection of the Danube River (ICPDR) is based on the sign of the convention of protection and sustainable use in Danube River, which means that ICPDR was supposed by international law initially. The members of ICPDR signed Danube River Protection Convention on 29 June 1994 in Sofia-Bulgaria, which is a legal basis for the cooperation in the field of environment and water. The DRPC is a legal framework for the cooperation to assure the protection of water and ecological resources and their sustainable use in the Danube River Basin.

A powerful Management Agency：ICPDR (Figure 1).

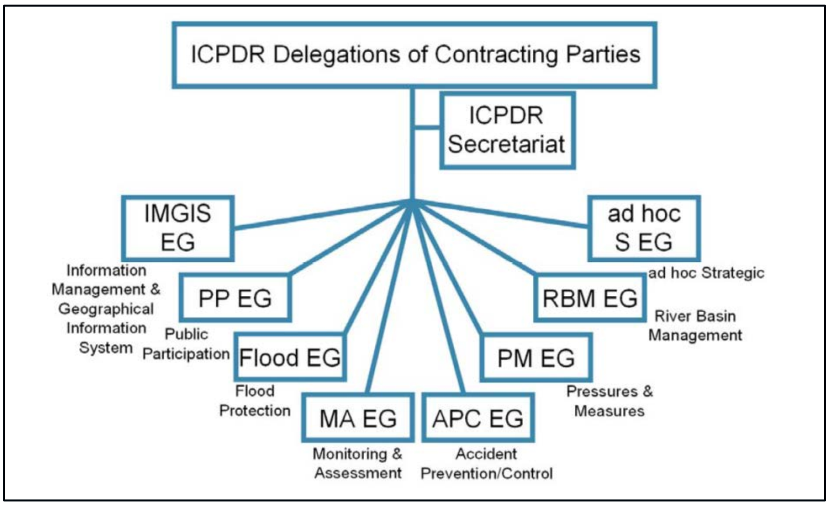

Figure 1: The organization structuring of ICPDR (ICPDR, 2011)

ICPDR is established based on the DRPC. The main functions of ICPDR are: this should have the mandate to ensure conservation, improvement and rational use of surface waters and ground water; to reduce inputs of nutrients and hazardous substances; control floods and ice hazards; reduce pollution loads to the Black Sea. Since 2000 the ICPDR is the coordinating body for implementing the EU Water Framework Directive in DRB. The Convention established the International Commission for the Protection of the Danube River which acts as a forum for cooperation and dialogue on water related issues and environmental issues dealing with water.

4.2 Equity and effectively cooperation mechanism in Multilevel

ICPDR divided the whole basin into three different coordination levels: basin-wide level, the national level or internationally coordinated sub-basin level, the subunit level. ICPDR mainly deal with the issues on river basin level, and also help deal with bilateral or multilateral cooperation problem (ICPDR, 2009a).

4.3 Effective River Basin management planning

The Danube River Basin Management Plan (DRBMP) is a detailed plan of all countries of the Danube River which specifies the status of water in the Danube River and the actions needed to achieve “good status” as a basis for sustainable development. It was finalised in

December 2009. This plan includes a joint programme of measures and evaluation on measures implementation, and also reflects the status of water of the DRB waters and significant water management issues. It draws many conclusions on investment and funding, which contributes significantly to the river basin management (ICPDR, 2009a).

4.4 Setting up of monitoring networks and early warning signals

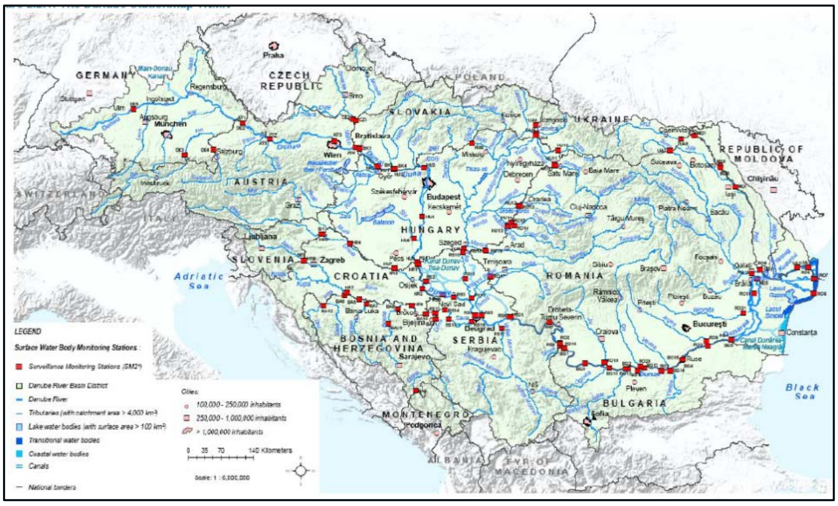

Figure 2: The distribution of TNMN monitoring station in Danube[4]

In the Danube River, TransNational Monitoring Network (TNMN) began in 1985. This was formally put into use in 1996 (Figure 2). This was established to know more about the changes of the water quality and mainly consisted of 11 control sections. In 2007, TNMN was corrected to monitor the water quality real-timely, monitor the water quality of surface water long-termly and provide the change of water quality of groundwater when transnational contamination accident occurs if necessary.

TNMN in the Danube River mainly monitors contents including organic pollution (organic carbon from point and non-point source), nutrient pollution (nitrogen and phosphorus from point and non-point source), hazardous substance pollution (point and non-point source), the effect of alternating fluvial facies on water quality and water quality of groundwater (ICDPR, 2009b; Joint Action Program, 2007).

In April 1997, ICPDR set up the warning system and put it into use. The warning system is the Danube River Accident Emergency Warning System (DAEWS). The DAEWS provides immediate information on sudden changes in water characteristics, like accidental river pollution incidents having transboundary effects, to assist the responsible authorities and water users in the downstream countries to make preventive measures in time (Hartong et al.,1997).

4.5 Retain very broad wetland at the mouth

In early 1980s, the biosphere reserve is set up to protect the ecosystem of the Danube River Delta. Within the biosphere reserve, the reclamation and potentially damaging activities are banned by law. The existing nature reserve is considered to be undisturbed zones which are totally protected.

What's more, as part of the Danube Delta Strategy and Management Plan, a priority action program is being prepared. The plan will be a "statutory plan" legally binding for all national agencies, answerable to the Ministry of Environment (UNEP, 2012).

4.6 Effectively public participate mechanism

In the Danube River, the plans developed by both ICPDR and the government are disclosed to the public to consult for suggestions. In addition, organisations such as relevant international organisations and non-governmental organisations participate in the management of the river basin of Danube in the position of observers (Shelton et al., 1994; ICDPR, 2009). For instance, these organisations provide relevant information and propagandise the Danube Day, which takes place in June 29th every year.

4.7 The contrast of the Danube and Huai River

The Danube River Basin is about three times larger than the Huai River Basin. It means that the impact of human activities, including industry, agriculture and municipal activities, on the water resources and environment of Huai basin is much higher then it of the Danube River. Table 1 below shows the contrast of two river basins.

Table 1: The comparison of the natural, social and economic condition between the Huai River, the Rhine and the Danube.

The River	*length/km*	*drainage area / km2*	*mean annual runoff /(m3.s-1)*	*mean basin precipitation /mm*	*Per Capita GDP / $*	*population density /(per.km-2)*	*Urbanization Rate /%*
Huai River	1000	270000	1969	888	1400	740	<40
The Danube	2850	821000	6430	500-1000	7200	106	58

4.8 Recommendations for the Huai River Basin Management

Needs powerful administrative machinery: Publish a law to give legal support to the commission; and make the mechanism functions cover all aspects of river basin management instead of inclining to the water conservancy.

Needs effective public participation: Sharing of environmental monitoring data in multilevel organisations; improve the citizen's environmental protection consciousness by the government; and perfect the laws and regulations about environmental NGO and encourage the development of environmental NGOs.

Needs monitoring and warning system: Add physical and chemical properties of groundwater project in the monitoring items, which is good for both the research on movement and transformation of pollutants in groundwater and supervision the behaviour of illegal enterprise to pollute the groundwater directly; and construct the warning system independent from the monitoring to minimise the loss of property and the influence on people's life and health when sudden pollution accidents occur.

Need to keep basic reserved areas for regulating: Set up natural reserve in the area of wetland at the mouth to maintain water ecosystem balance; and make special plans and laws to protect the wetland in the river basin efficiently.

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Successful restoration of a tropical shallow eutrophic lake: strong bottom-up but weak top-down effects recorded

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Abstract

Fish manipulation has been used to restore lakes in the temperate zone. Often strong short-term cascading effects have been obtained, but the long term-perspectives are less clear. Fish manipulation methods are far less advanced for warm lakes, and it is debatable whether it is, in fact, possible to create a trophic cascade in warm lakes due to the dominance and high densities of fast-reproducing omnivorous fish. However, other important aims of fish manipulation, for instance, the removal of benthic feeding fish, are to reduce disturbance of the sediment, which not only affects the nutrient level but also the concentration of suspended organic and inorganic matter with reduced clarity as a result, and hampers growth of submerged macrophytes. We conducted a biomanipulation experiment in two basins of Chinese Huizhou West Lake that has remained highly turbid after extensive nutrient loading reduction. A third basin was used as control (control-treatment pairing design). Removal of a substantial amount of plankti-benthivorous fish was followed by planting of submerged macrophytes and stocking of piscivorous fish. We found strong and relatively long-lasting effects of the restoration initiative in the form of substantial improvements in water clarity and major reductions in nutrient concentrations, particularly total phosphorus, phytoplankton and turbidity, while only minor effects were detected for crustacean zooplankton grazers occurring in low densities before, as well as after the restoration. Our results add importantly to the existing knowledge of restoration of warm lakes and are strongly relevant, not least in Asia where natural lakes frequently are used extensively for fish production, often involving massive stocking of benthivorous fish. With a growing economy and development of more efficient fish production systems, the interest in restoring lakes is increasing world-wide. We found convincing evidence that fish removal and piscivores stocking combined with transplantation of submerged macrophytes may have a substantial role in conservation and management of warm water lakes.

Keywords

Shallow lakes, restoration, bottom-up and top-down effects, tropical china

1. Introduction

For 50-100 years, eutrophication has posed a serious threat to lakes worldwide. High nutrient loading has resulted in turbid water, excessive blooms of nuisance algae, dominance of coarse fish and loss of biodiversity. While countries in the developing world face an alarming increase in eutrophication of lakes as a result of the rapid economic development, large efforts have been devoted to combat eutrophication by reducing the external loading of nutrients, especially phosphorus (P), not least in Western Europe and

North America (Sas, 1989, Jeppesen et al., 2005a, Schindler, 2009). In consequence, the P loading from sewage and industrial sources has declined significantly since the 1970s (overviews in Sas, 1989, Jeppesen et al., 2005b). This has led to reduced phytoplankton abundance, lower cyanobacteria dominance and often a shift in fish community structure towards lower abundance and proportions of coarse fish with a resultant reduction in the fish predation on zooplankton. However, many lakes have responded slowly to nutrient loading reduction (Sas, 1989, Marsden, 1989, Jeppesen et al., 2005a, b), partly due to internal P loading (release from the stored P pool in the sediment) (Søndergaard et al., 2013) and also due to biological resistance (Jeppesen and Sammalkorpi, 2002). The latter reflects that the fish after an external nutrient loading, the reduction may exert a continuous predation pressure on large-bodied grazers (e.g. *Daphnia*) and thereby maintain the grazing pressure on phytoplankton at a low level, also diminishing the number of benthic animals that stabilise and oxidize the sediment. Furthermore, persistence of benthivores (e.g. carp, *Cyprinus carpio*, bream and *Abramis brama*) that stir up the sediment while feeding and translocate nutrients from the sediment to the water (Meijer et al., 1994, Breukelaar et al., 1994) contributes to maintain high internal P loading and high inorganic turbidity. Moreover, grazing by herbivorous waterfowl such as coot (*Fulica atra*) and mute swan (*Cygnus olor*) may hamper the recovery by delaying the recolonisation of submerged plants (Perrow et al. 1997, Mitchell and Perrow, 1998).

To re-enforce recovery, several physicochemical and biological methods have been used (for an overview see Cooke et al., 2005). The typical measure applied to overcome biological resistance is removal of plankti-benthivorous fish. This method has been extensively used in north temperate lakes, particularly in Europe. Removal of approx. 75% of the planktivorous and benthivorous fish stock over a 1-2 year period has been recommended to avoid regrowth and to stimulate the growth of potentially piscivorous fish (Hansson et al., 1998, Jeppesen and Sammalkorpi, 2002, Mehner et al., 2002). An alternative or supplementary method to fish removal is stocking of piscivores (for a review see Drenner and Hambright, 1999). To reinforce recovery of submerged macrophytes active planting (entire plants, turions, plant fragments or seeds) has been used (Jeppesen et al., 2012), in some cases also by constructing plant exclosures to protect the macrophytes against grazing by waterfowl and fish (Søndergaard et al., 1996). In the exclosures the macrophytes can grow in a grazing-free environment from where they may spread seeds, turions or plant fragments to the entire lake, thereby augmenting the chances of successful colonisation (Søndergaard et al., 1996, Mitchell and Perrow 1998, Lauridsen et al., 2003).

In north temperate lakes, efficient fish reduction in eutrophic lakes has generally led to dramatic cascading effects in the form of reduced phytoplankton biomass, dominance by large-sized zooplankton and improved transparency (Carpenter and Kitchell, 1993, Hansson et al., 1998, Søndergaard et al., 2008). The effects of fish manipulation may cascade to the nutrient level as well. A reduction ranging from 30 to 50% in lake concentrations of total phos- phorous (TP) has been recorded in the relatively successful fish manipulation experiments conducted in shallow and stratified eutrophic lakes (Søndergaard et al. 2008). A significant contributory factor is increased growth of micro-benthic algae owing to improved light conditions at the sediment surface (Hansson, 1990, Genkai-Kato et al., 2012). More benthic algae and less sedimentation of phytoplankton due to intensified grazing, and more benthic animals due to reduced fish predation, may all result in a higher redox potential in the surface sediment, potentially reducing the P release (Søndergaard et al., 2005, Zhang et al. 2013). So far, the long-term perspectives of fish removal are less promising. A gradual return to the turbid state and higher abundance of zooplanktivorous fish after 5-10 years has been reported in many case studies (Søndergaard et al., 2008). Moreover, stocking of piscivorous fish has often been less successful than fish removal (Drenner and Hambright, 1999).

Experience with lake recovery is far less advanced for warmer lakes than for temperate lakes (Jeppesen et al., 2005b, 2012). Studies conducted in (sub) tropical and Mediterranean lakes have shown that nutrient loading reduction may improve the ecological state via a declining algal biomass and increased water transparency (Jeppesen et al., 2005a, Coveney et al., 2005, Romo et al., 2005, Beklioğlu and Tan, 2008). However, it is debatable whether the fish manipulation approach used in cold temperate lakes to re-enforce recovery can be used with success in warmer lakes as well (Jeppesen et al., 2005b, 2012). The high species richness and high densities of plankti-benthivorous fish, with dominance of omnivores, a few efficient predators and several cohorts lead to higher predation on zooplankton in warmer lakes than in temperate lakes (Lazzaro, 1997, Meerhoff et al., 2003, 2007, Teixeira-de Mello et al., 2009). It is therefore likely that a removal-induced reduction of the biomass of planktivorous fish will be compensated by fast adjustment of the remaining population, and the impact will consequently be of short duration (Jeppesen et al., 2012). Hence, it may be more difficult to provoke and maintain a pelagic trophic cascade effect in subtropical and tropical lakes than in temperate lakes. However, an important effect of removal of benthic feeding fish, such as carp and bream, is reduced stirring of the sediment, which not only affects lake nutrient levels but also the concentration of suspended organic and inorganic matter (Breukelaar et al. 1994). Not least in systems with high dominance of carp, either naturally or stocked, as is the case for many Chinese lakes (Jia et al., 2013), carp removal may potentially lead to substantial improvement of lake water quality due to reduced disturbance.

2. Method

In the present study we conducted a biomanipulation experiment in two parts of Chinese Huizhou West Lake that has remained highly turbid after an extensive nutrient loading reduction (Li et al., 2009). Removal of plankti-benthivorous fish was followed by planting of submerged macrophytes and stocking of piscivorous fish. Our working hypothesis was that albeit a major trophic cascade mediated by enhanced zooplankton grazing (due to the high density of small fish in warm freshwaters) could not be expected, fish removal would lead to clear-water conditions and low nutrient levels due to reduced disturbance of the sediment, and water clarity would be maintained via introduction of submerged macrophytes.

3. Results and discussion

We found strong and relatively long-lasting effects of the biomanipulation initiative encompassing removal of plankti-benthivorous fish, macrophyte transplantation and stocking of piscivorous fish. This included substantial improvements in water clarity and major reductions in the concentrations of nutrients, particularly TP, phytoplankton (Chl_a) and turbidity (TSS), while only minor changes were detected for the crustacean zooplankton grazers occurring in low densities before, as well as after biomanipulation. Thus, we found evidence for strong bottom-up effects, while top-town effects apparently played a minor role in the improvement of the ecological status of the lake.

We observed a particularly strong decline in TSS and ISS, which may reflect both a reduction in fish disturbance of the sediment due to the removal of benthivorous fish and a reduced risk of resuspension following the development of submerged macrophytes. The dominant benthivorous fish were common carp and crucian carp that feed mainly on benthos (Richardson et al., 1995, García-Berthou, 2001) by sucking up sediments from where they pick up edible items and release the sediment to the water (Scott and Crossman, 1973). Such feeding behaviour can cause high sediment resuspension. Both common carp (Breukelaar et al., 1994, Zambrano et al., 2001, Wahl et al., 2011) and crucian carp (Richardson et al., 1995) have been shown to cause high sediment resuspension in temperate lakes, and resuspension may be even higher in warm lakes where the fish forage more actively due to the higher temperatures (Lankford and Targett, 1994). Tilapia can also suspend sediment and increase turbidity in shallow systems (Jiménez-Montealegre et al. 2002). Furthermore, fish biomass values were high (nearly 2500 kg ha^{-1}) compared to most temperate lakes (Jeppesen and Sammalkorpi, 2002). Re-establishment of submerged macrophytes may further stabilise the sediment and it reduces sediment resuspension (Hamilton and Mitchell, 1996). Several biomanipulation studies conducted world-wide have observed effects on TSS after removal of benthivorous fish (Meijer, 1994, 1999, Jeppesen et al., 2007, Ibelings et al., 2008).

Fish removal in temperate lakes has also resulted in major reductions in total nitrogen (TN) and TP (Jeppesen et al., 2007, Ibelings et al., 2007, Søndergaard et al., 2008). In our study, the most clear response emerged for TP, which, on average, declined by >70% compared to both the level before restoration and the level in the reference lake. Fish feed on benthos and excrete nutrients in the water column and thus fish removal may reduce the translocation of nutrients from benthic habitats to water column (Glaholt and Vanni, 2005, Vanni et al, 2013). Submerged macrophytes oxidize the sediment and increase its capacity of binding inorganic P, thereby reducing P release to the water column (Carpenter and Lodge, 1986). In our study, the dominant species after plant introduction (*Vallisnaria*) has relatively well developed root systems (Xie et al., 2005) and thus a high potential of oxidizing the sediment, thereby reducing the P release. Moreover, high water clarity and improved light conditions may allow development of benthic algae and consequently cause a reduction in the sediment P release (Hansson, 1990, Genkai-Kato et al., 2012, Zhang et al., 2013) in shallow lakes, such as Huizhou West Lake.

In north temperate lakes appearance of large-sized zooplankton has been shown to be of key importance for enhancing water clarity after biomanipulation. The large-sized zooplankton increase the grazing pressure on phytoplankton, as has been evidenced in several studies by a major increase in the zooplankton: phytoplankton biomass ratio and a larger body size of cladocerans (Hansson et al., 1998, Jeppesen et al., 2004, Søndergaard et al., 2008, Jeppesen et al., 2012). Higher water clarity, in turn, promotes growth of submerged macrophytes, further stabilising the lake ecosystem through a number of positive feedback mechanisms (Moss, 1990, Scheffer et al., 1993, Jeppesen et al., 1998). In our study the phytoplankton biomass (Chl_a) also decreased substantially after biomanipulation. However, top-down effects appeared to have been playing the minor role as we found no clear change in the zooplankton community – the abundance of cladocerans remained low and small species such as *Chydorus* sp., *Alona* spp. and *Moina micrura* continued to dominate. This lack of response by the zooplankton may be attributed to continuously high fish predation even after fish removal. Many of the fish species present in the study lakes spawn several times per year, including crucian carp and tilapia (Pan et al., 1991), and young-of-the-year fish are thus abundant all year around to prey on the large-bodied zooplankton, as seen in other warm lakes (Havens and Beaver, 2011, Meerhoff et al., 2007, 2012). It has been suggested that dominance of small-bodied zooplankton may be attributed to the higher temperatures which may render small-sized forms superior competitors for physiological reasons (Moore et al., 1996), but a recent study clearly indicates that large-bodied zooplankton may become dominant even in warm lakes if fish predation is absent (Iglesias et al., 2011). Consequently, we attribute the low abundance and dominance of small-sized zooplankton to continuously high fish predation after biomanipulation.

Submerged macrophytes can reduce phytoplankton through bottom-up effects and consequently promote clear water due to both direct and indirect suppression (Jeppesen et al., 1998). Macrophytes and associated periphyton may compete with phytoplankton for nutrients (Sand-Jensen and Borum, 1991), reduce nitrogen and phosphorus availability for phytoplankton through denitrification in macrophyte beds, diminish nutrient release from the sediment or produce allelopathic substances against phytoplankton (Gross et al., 2007). The unchanged chlorophyll (Chl_a) to TP ratio indicates, however, that allelopathic effects did not play an important role in controlling phytoplankton in our restored basins. Following the restoration the TN:TP ratio rose in both basins relative to the unrestored site (significantly in only one of the basins), which might indicate less limitation of phytoplankton by nitrogen. Nitrate nitrogen was also lower, not least in the winter season. However, as no change occurred in Chl_a:TP with decreasing Chl_a:TN and as the TN:TP ratio increased after restoration, we suggest that bottom-up effects through reduced phosphorus availability are the likely main mechanisms behind the phytoplankton reduction after restoration in Huizhou West Lake.

Due to the lack of efficient grazers, improved light conditions mediated by a reduction in the concentrations of suspended solids may potentially favour phytoplankton growth if natural colonisation of macrophytes is delayed. Speeding up the re-establishment of submerged macrophytes via transplantation may therefore be a useful tool in restoring and maintaining the clear state in warm lakes. The high longevity of the restored clear water state in Huizhou West Lake (so far more than 5-8 years) may in part be due to the high coverage and dominance of *Vallisneria*. *Vallisneria* is well-rooted and is a meadow-forming species with a high capacity of reducing sediment resuspension from fish. *Tilapia* and common carp graze submerged macrophytes to a certain degree (Petr, 2000, Miller et al., 2007), but *Vallisneria* is less vulnerable to grazing than *Hydrilla* (Van et al. 1998). Another factor may be the maintenance of fishing, mainly tilapia and crucian carp (ca. 450 kg ha^{-1} $year^{-1}$) in the restored site following the first manipulation; however, the maintenance catches are similar to the commercial catches in the unrestored site of mainly silver carp (ca. 500 kg ha^{-1} $year^{-1}$) (Gao et al., 2014).

4. Conclusion

Our results have important implications for lake restoration, not least in Asia where natural lakes so far have been used extensively for fish production with often massive stocking of benthivorous fish. With a growing economy and the development of more efficient fish production systems, the interest in restoring lakes is increasing. Scientists, not least in China, have argued for massive stocking of silver carp and bighead carp to control phytoplankton growth and not least nuisance cyanobacteria. This suggestion is supported by analyses using minimal models with all their limitations (Attayde et al. 2010), whereas the results of field experiments are ambiguous and mostly negative (Wang et al. 2008). Our results provide strong evidence that fish removal combined with transplantation of submerged macrophytes may have strong and relatively long-lasting effects on the water clarity in warm shallow lakes despite a low zooplankton grazing potential, which is probably due to high fish predation on the zooplankton.

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Optimisation of a protocol for the quantification of black carbon in peat soils

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Abstract

Black carbon (BC), resulted primarily from the combustion of fossil fuels and biomass, is a key component of PM2.5 in aerosols and could cause global warming. More than seven methods have been developed for quantifying BC by using a range of materials. However, no single method is regarded suitable for all materials being used. Unlike the materials such as sediments or loess, have suitably fitted methods for measuring the BC, there was no appropriate method being used for soils with high organic materials (e.g. peat soils). Among all methods, dichromate oxidation and chemothermal oxidation (CTO375) could quantitatively estimate the BC and stable carbon isotopes of the BC simultaneously. These methods are least expensive, and could be carried out in most laboratories. Here, we have compared the two quantitative approaches, modification and optimisation methods for BC extraction from peat soils. The results show that the CTO375 method may destroy a foremost component of BC in peat soils which is produced by wildfire (lower than 800°C), and not suitable for measuring the BC in peat soils. Through the test of BC reference materials (wood char), increasing the number of 0.1mol/L NaOH for 12h to twice could remove humic acid in peat soils completely and would not cause the content of BC lower than from the method being used originally (new method: 48.7-50.2%, n=3; original method: 48.4-55.8%, n=4). In all, dichromate oxidation method, a step wide removal of humic acid is better to measure the content of BC in peat soils.

Keywords
Peat soils, black carbon, dichromate oxidation, CTO375

1. Introduction

Black carbonis (BC) produced by incomplete combustion of fossil fuels and vegetation, like soot and char (Schmidt and Noack, 2000). The BC is largely resistant to oxidation and biological decomposition, therefore, contributes to a stable carbon reservoir in soils and sediments (Preston and Schmidt, 2006). In addition, the global emissions of BC could be as high as 7500 Gt yr-1, and BC might be the second most important human emission next to the carbon dioxide in terms of its climate forcing (Bond et al. 2013). Quantifying the BC reservoir in deposition archives offers clues in where the missing BC resides and historical trends of the atmospheric BC (Cong et al., 2013). There is about 1600 Gt carbon stored in the soils carbon pool, of which 28% were stored in wetlands. Peatlands, highly organic wetlands, could store 0.07 Gt carbon every year (Shrestha et al., 2010). Black carbon, as an important type of organic carbon in peatlands, is affected by historical BC deposition fluxes and influence total carbon accumulation rates through altering the microbial activity

(Lehmann and Joseph, 2012). Therefore, accurately quantifying the BC contents in peat soils is critical not only for studying palaeoecological evidences by documenting it in peat profiles, also for investigating the BC stocks and stable carbon pools in peatlands.

At present, a number of methods have been used to quantify the BC in the soils and sediments, such as quantifying the charcoal or BC through the microscopic count (Morales-Molino et al, 2013), the acid dichromate oxidation method (Wang et al, 2013), the chemothermal oxidation method (CTO375) (Paroissien et al, 2012) and the thermal/optical carbon analyser method (Han et al, 2012). Hammes et al. (2007) used seven methods to quantify the BC in twelve materials from soil, water, sediment and the atmosphere except peat soils. Their results showed that the BC contents measured by different methods were differed significantly; no single method was found suitable for all materials. Peat soils are unique and have many characteristics such as high organic matter content, vegetation litter and pollen that differ from other materials. These characteristics could have influenced the results of BC quantification. Hence developing a suitable method to accurately quantify the BC in peat soils is necessary.

Black carbon in soils and sediments is defined as a carbonaceous substance of pyrogenic origin, which is resistant to thermal or chemical degradation (Hammes et al, 2007). Both CTO375 and acid dichromate oxidation method could quantitatively estimate contents and stable carbon isotopes of BC simultaneously. These methods are least expensive, and could be carried out in most laboratories for determining the BC in soils and sediments, but scarcely in peat soils. Our current study was to compare the two above-mentioned methods, modified and optimised for a suitable method to quantify the BC in peat soils.

2. Materials and methods

The types of materials were collected from three different depths in a peat profile (Shenjiadian peatlands (N46°34.864′, E130°39.873′). Sanjiang Plain, Northeast China). Detail information of Shenjiadian peatlands were shown in Gao et al (2014). Loss-on-ignition (LOI) (Lamarre et al, 2012) and humification degree (Wang et al, 2010) were measured to compare the organic matter contents in different materials. The characteristics of materials are presented in Table 1. The high LOI value means high organic matter content in the materials; high humification degree indicates high humic acid in the materials. The analysis results showed that the study materials have typical characteristics of peat soils, therefore we used these samples in the study to compare the different methods of measuring BC in peat soils.

In this study, we used the dichromate oxidation method developed by Song et al. (2002), and the CTO375 method developed by Gustafsson et al. (1997, 2001). Both methods removed inorganic carbonates by HCl, and quantified the residual carbon as BC by using a continuous-flow isotope ratio mass spectrometer (CF-IRMS) finally. The difference between them was how to remove NPOC (non-pyrogenic organic carbon) in samples. The dichromate oxidation method used 0.1 mol/L of NaOH solution to remove humic acid, and removed kerogen by K2Cr2O7 and H2SO4 mixed solution, but in the CTO375 method, all NPOC were removed by oxidized at 375°C for 24 h in the presence of excess oxygen.

In order to optimise for a suitable method, we added one step further (removing humic acid by 0.1 mol/L NaOH solution twice in the pretreatments) in the pretreatment of dichromate oxidation method. Black carbon reference material (wood charred, produced in Department of Geography, University of Zurich, (Hammes et al., 2006)) was used to evaluate whether the added step could cause the BC loss.

3. Results and discussion

3.1 Comparison of two methods to measure black carbon in peat soils

The contents of BC in three types of materials measured by dichromate oxidation and CTO375 method were shown in Figure 1a. The results measured by dichromate oxidation method were higher than those that obtained from CTO375 method, and more clearly in materials with high organic matter contents (A and B). Humification degree in material A is lower than that in B. High organic matter contents and low humification degree caused the results measured by dichromate oxidation are six times higher than that obtained by CTO375 method. The original dichromate oxidation method could not remove humic acid by 0.1 $molL^{-1}$ NaOH solution completely in peat soils, and residual humid acid caused the oxidation time decreased in the next step, some of kerogen and plant litter may residue and lead the results higher than the real BC contents. In all, the original dichromate oxidation method could not determine the BC in peat soils which contain amount of organic matter and plant residues.

In all three types of materials, the results of BC in type C with the low LOI were similar between different methods. Type C were located at the bottom of peat and top of silt, where the plant litter had been decomposed. In this kind of materials, the NPOC could be removed through pretreatment in dichromate oxidation method, and the results measured by this method could be regarded as the real BC contents. The contents of BC in type C materials obtained by CTO375 method were lower than that obtained by dichromate oxidation method. Similar with other deposition archives, the sources of BC in peat soils are natural sources (e.g. wildfire) and anthropogenic sources (Rius et al., 2009; Cong et al., 2013). The BC, produced by incomplete burning, was major sources of the BC in peat soils and mainly produced under low temperature (800°C) (Matt-Davies and Belcher, 2013). Hammes et al (2007) had measured the BC contents in grass charred and wood charred, and the results were much lower than those obtained by other methods. The CTO375 method might have destroyed the BC which formed under low temperature (e.g. 1000°C) (Hammes et al., 2007). This indicates that CTO375 method might not be suitable for measuring the BC formed by incomplete combustion. Wildfire is an important source of BC in peat soils and could not be ignored. Therefore, the CTO375 method was not appropriate to measure the BC in peat soils.

3.2. Optimisation of dichromate oxidation method to measure black carbon

Low temperature of BC sources and high content of organic matter in peat soils, may affect the results of BC contents obtained by these traditional methods, and lead they could not determine the BC accurately. It is necessary to develop a new method to remove the interference of organic matter and determine the BC contents in peat soils. Unlike CTO375 which may destroy an important type of BC in peat soils, dichromate oxidation method cause the BC no loss and could be optimised for determining BC in peat soils. Organic matter and plant litter might remain after pretreatment and the results in higher than the real BC contents in peat soils by traditional dichromate oxidation method. Optimisation protocol to remove the organic matter and plant litter completely in the pretreatment could lead to a new dichromate oxidation method more accurately. The way to remove the plant litter in the pretreatment of traditional dichromate oxidation method is the final step (dichromate oxidation). The roots and rhizosphere soil get from wetland plants communities (*Equisetum fluviatile*) were used to test whether the dichromate solution could remove plant litter. The result measured by dichromate oxidation method in this material is 1.7%, which is far lower than the carbon contents in this material. The result showing after 60 hour use of dichromate oxidation, this has removed the plant residue completely.

The most effective method for removing the humic acid is 0.1 mol/L NaOH solution, and the 12 hours of the NaOH solution in traditional method could not remove the humic acid in peat soils completely. Therefore, increase the times of 0.1mol/L NaOH solution to remove the humid acid is necessary. Through pre-experiment, the colour of the NaOH solution was colourless after added in the third times which may indicate that the humic acid had been removed under the twice of NaOH solution processes. However, the extended the NaOH process may destroy the BC in peat soils. We used the BC reference material wood charred (Hammes et al., 2006, 2007) to determine whether the added steps could cause the BC loss. The content of the BC in wood charred measures by new dichromate oxidation method and traditional method (Hammes et al., 2007) have been shown in Figure 1b. The result measured by new protocol is similar to that of measured by traditional dichromate oxidation method in four other laboratories (Hammes et al., 2007). The result in Figure 1b shows that the steps of added NaOH solution twice did not cause the BC loss. Therefore, the new protocol could remove the humic acid in peat soils completely and determine the BC contents in peat soils more accurately.

Table 1: Materials Characters of different position in peat profile; values are means with standard errors (n=3).

Materials type	***LOI (%)***	***Humification degree***	***Colour***	***Classification***
A	69.42±1.71	34.53±0.03	yellowish-brown	peat soil
B	65.12±4.98	40.87±0.87	brown	peat soil
C	16.68±4.39	22.80±1.76	black	sediment

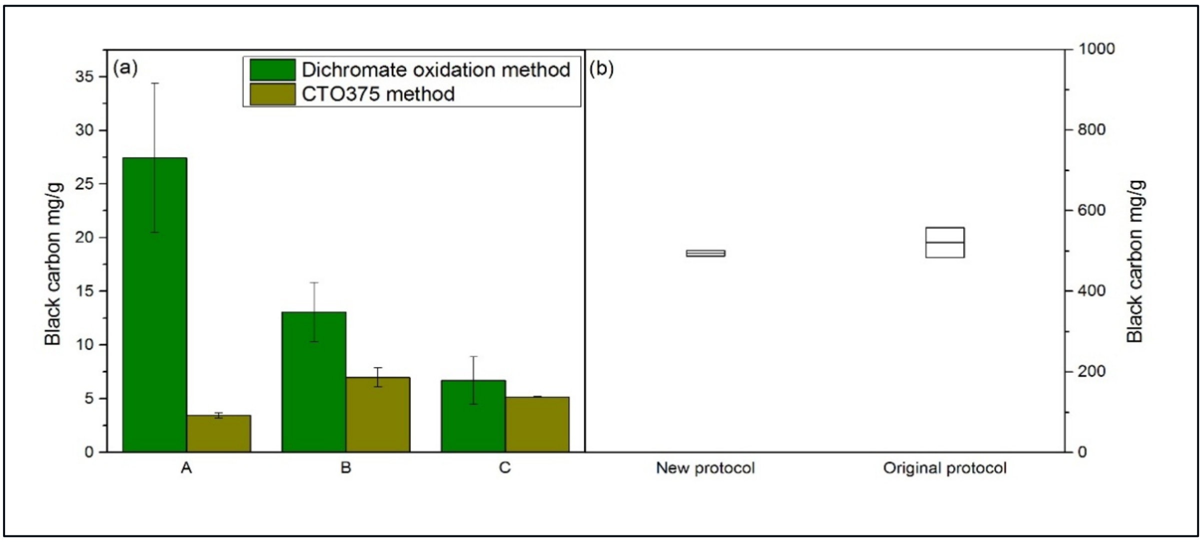

Figure 1: (a) Effect of different methods on black carbon concentrations measured in three types of materials. The values are means with standard errors (n = 3). (b) Effect of new and original protocol (Hammes et al., 2007) on black carbon concentrations in black carbon reference materials (wood charred (Hammes et al., 2006)).

4. Conclusions

Because the CTO375 method may destroy the BC produced by low-temperature (e.g. wildfire) and traditional dichromate oxidation method could not remove the organic matter and plant litter completely, these traditional methods are not suitable for determining the contents of the BC in peat soils. Through optimising the dichromate oxidation method, increase the steps of NaOH solution could remove the humic acid in peat soils completely and nearly no BC loss in this process was noticed. Humic acid in peat soils could be removed completely through the NaOH solution and lead to the plant litter to be oxidized by dichromate totally in the next step. In all, the new protocol of dichromate oxidation method is suitable for determining the BC in peat soils.

Acknowledgements

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Diatom-based ecological classification of shallow lakes in the Badain Jaran Desert (Inner Mongolia, China)

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Abstract

In recent years in northern China, widespread drought-induced lake shrinkage and desiccation have been observed. In that context, it is crucial to study these threatened shallow lake ecosystems. This paper focuses on the diversity and composition of diatom assemblages in surface-sediment samples and assesses the potential for a diatom-based classification of shallow lakes in the Badain Jaran Desert, Inner Mongolia. Surface-sediment samples and associated limnological data were collected from 42 sites. The first axis of a principal component analysis on the environmental variables is highly correlated with the salinity gradient and related factors such electrical conductivity (EC) and the concentrations of major ions. Diatoms were absent from the 16 most saline lakes in the dataset, i.e. with values for salinity > 80 g L^{-1}. The 26 remaining lakes, in which diatoms were found, were classified using hierarchical cluster analysis and similarities between samples were mapped using non-metric multidimensional scaling (NMDS). Three lake groups were defined that closely correlate with lake water EC. The type-specific diatom taxa for the three lake types were determined by using indicator species analysis (IndVal). The diatom flora of subsaline lakes is remarkable as it is a mixture of taxa often reported in alkaline springs such as *Encyonopsis descripta var. asymmetrica*, *Encyonopsis krammeri*, *Achnanthidium caledonicum* and *Achnanthidium thermale* and taxa more often associated with brackish conditions. A rapid loss of diversity as salinity increases is observed, in agreement with previous studies. The taxa type-specific in mesosaline lakes are cosmopolitan, halophilic and epipelic species, such as *Brachysira aponina*, *Nitzschia prolongata*, *Proschkinia complanata* and *Seminavis pusilla*. Overall, the diatom communities in lakes of the Badain Jaran Desert exhibit a distribution pattern that closely corresponds with that observed in saline lakes elsewhere.

Keywords

Diatoms, ecological classification, shallow lakes, Inner Mangolia, Badain Jaran Desert

1. Introduction

Shallow lakes are particularly vulnerable to the impacts of human activities and climate change and as such are among the most threatened ecosystems in the world. In recent years drought-induced lake shrinkage and desiccation have been observed in some parts of northern China. In general, lake waters have become salinised, and freshwater marsh has been replaced by salty marsh, threatening the populations of endangered waterfowl species as well as the aquatic ecosystem (Liu et al., 2013). In that context of rapidly changing conditions, it is important to gather data on the limnological variables and the distributions of aquatic organisms of shallow lakes from the arid regions of China.

The Badain Jaran Desert is a remote sandy desert in arid north-western China. This desert is remarkable due to its unique landscape in which megadunes coexist with >100 permanent shallow, spring-fed, seepage lakes (Hofmann, 1996; Yang, 2000; Yang and Williams, 2003). Most of the desert is uninhabited and therefore relatively unaffected by human activities. It is therefore an attractive field area to carry out a biological assessment of its natural shallow lakes.

Among the biological organisms that can be found in lakes from the world arid regions, diatoms have been widely used as indicators of lake water conditions, especially salinity, in a variety of modern and paleo-ecological studies (Fritz et al., 1993; Gasse et al., 1995; Wilson et al., 1996; Gell, 1997; Reed, 1998; Davies et al., 2002; Ryves et al., 2002; Yang et al., 2003; Shinneman et al., 2009; Reed et al., 2012).

In a previous paper (Rioual et al., 2013) presented a diatom-based transfer function for electrical conductivity and its application to short sedimentary diatom records. The aims of the current paper are (i) to further explore the physico-chemical characteristics of a set of 42 lakes in the Badain Jaran Desert, (ii) to classify these lakes according to their diatom assemblages and (iii) to identify diatom species with high specificity and fidelity to certain type of lakes along the salinity gradient.

2. Material and methods

2.1 Diatom samples collection and analysis

In Rioual et al. (2013) described the geography and meteorology of the study area and of the lakes in detail and collection of diatom data were given Rioual et al. (2013) also corrected identification of diatom taxa including *Encyonopsis eifelana* Krammer*, Encyonopsis subminuta* Krammer & Reichardt and *Encyonopsis thumensis* Krammer and *Encyonopsis krammeri* Reichardt. Also the counts of *Nitzschia angustiforaminata* Lange-Bertalot and *Nitzschia perminuta* (Grunow) M. Peragallo were merged and assigned to *Nitzschia cf frustulum* (Kützing) Grunow. Other taxa *Cymbella laevis* var. lata Krammer, *Navicula veneta* Kützing and *Sellaphora blackfordensis* Mann & Droop were *Cymbella subhelvetica* Krammer, *Navicula libonensis* Schoeman and *Sellaphora parapula* Lange-Bertalot, respectively. *Navicymbula pusilla* var. pusilla (Grunow) Krammer and *Navicymbula pusilla* var. lata Krammer are *Seminavis pusilla* (Grunow) Cox & Reid and *Seminavis lata* (Krammer) were updated based on Rioual (Rioual et al., 2014)

3. Numerical methods

3.1 Analysis on environmental variables only, 42-lake dataset

A Piper diagram and a Gibbs' plot were used to compare hydrochemistry of lakes and define water types (Hassan et al., 2012). The Piper diagram was generated using the US geological Survey program GW_Chart (water.usgs.gov/nrp/gwsoftware/GW_Chart/GW_Chart.html).

Relationships among the environmental variables were explored using principal component analysis (PCA), first on the full dataset (42 lakes), then on the subset of lakes with preserved diatoms (26 lakes). Prior to the analyses, environmental variables with skewed distributions were transformed to reduce the influence of extreme values. Three types of transformation, square-root, log10, log10 (x + 1) were tested for their ability to normalise the measured variables. The CANOCO version 4.5 program (ter Braak and Šmilauer, 2002) was used for this analysis and generating the ordination diagram.

3.2 Analyses on the diatom dataset, 26-lake dataset

To determine taxonomical similarities between the surface-sediment samples and to promote site typology, the samples were classified via a hierarchical cluster analysis (HCA), using Ward's linkage method with Bray-Curtis (also called Sorensøn) distance measure. Bray-Curtis similarities were measured on square-root transformed diatom relative percentages data. Only taxa with relative abundances ≥ 2% were included in the cluster analysis. The number of groups (k) was determined using a simple rule of thumb that sets the number to $k \approx \sqrt{n/2}$, with n as the number of objects (= samples) (Mardia et al., 1979).

The statistical significance of between-group differences was tested using Multi Response Permutation Procedure (MRPP). This analysis calculates an A-statistic, which is a descriptor of within-group homogeneity. The value of A ranges from -1 to +1. If the A-statistic is closed to +1, the clusters are completely different; if the A-statistic approaches 0, the heterogeneity between groups equals what would be expected by chance; if the A-statistic approaches -1 the clusters are homogeneous (Rimet et al., 2011). Bray-Curtis similarities measured on square-root transformed diatom relative percentages data were also used as the distance measured in MRPP.

Similarities were then mapped by non-metric multidimensional scaling (NMDS). NMDS is an ordination method that reduces the complexity of community data into fewer dimensions. The method is iterative and repeats the ordination calculations at random starting points until a reliable solution was found. Species are plotted at the centre of their distribution across samples (Stanish et al., 2011). A numerical measure of the closeness between similarities in the lower dimensional and the original spaced is called stress. The stress has a value between 0 and 1, with 0 indicating perfect fit and 1 indicating worst possible fit (Hassan et al. 2012). In addition, environmental variables were fitted as vector and surface on the NMDS plots to help with the interpretation. Fitted surface of environmental variables are done using generalised additive models (Oksanen, 2013). Only taxa with relative abundances ≥ 2% were included in the NMDS.

Besides classifying the lake types, it is also interesting to detect and describe the value of different species as indicators of type-specific environmental conditions. To that end, the indicator species analysis (IndVal), a method proposed by Dufrêne & Legendre (1997), was used to identify indicator species characterising groups of samples. The method produces indicator values for each species in each group expressed as the product of the specificity (relative frequency in groups) and fidelity (relative average abundance in groups). The statistical significance of the species indicator values is evaluated using a randomisation procedure. The indicator value is at its maximum when all individuals of a species are found in a single group of sites (high specificity) and when the species occurs in all sites of that group (high fidelity) (Dufrêne & Legendre, 1997).

Species diversity was assessed using rarefaction analysis, that permits an estimation of species richness for samples of different sizes when scaled down to a common size by considering the relative frequencies of individuals (Birks and Line, 1992). The diversity measure derived from rarefaction is called the expected richness, E(Sn).

HCA, MRPP, NMDS and IndVal analyses were performed using R statistical software (R Core Team Development, 2012) and in particular the packages "vegan" (Oksanen et al., 2013) and "labdsv" (Roberts, 2013).

4. Results

4.1 Environmental data and water chemistry

In the Piper diagram (Figure 1A), most lakes are located on the right hand-side of the main plot (diamond). This indicates that the large majority of the lakes are chloride-dominated with either sulphates or carbonates as sub-dominant anions. Only 5 lakes are carbonate-dominated. All 42 lakes have sodium as the dominant cation. According to the Gibb's plot (Figure 1B), that shows the variation of the weight ratio Na / (Na + Ca) as a function of salinity, evaporation is the mechanism that controls water chemistry in all sites investigated.

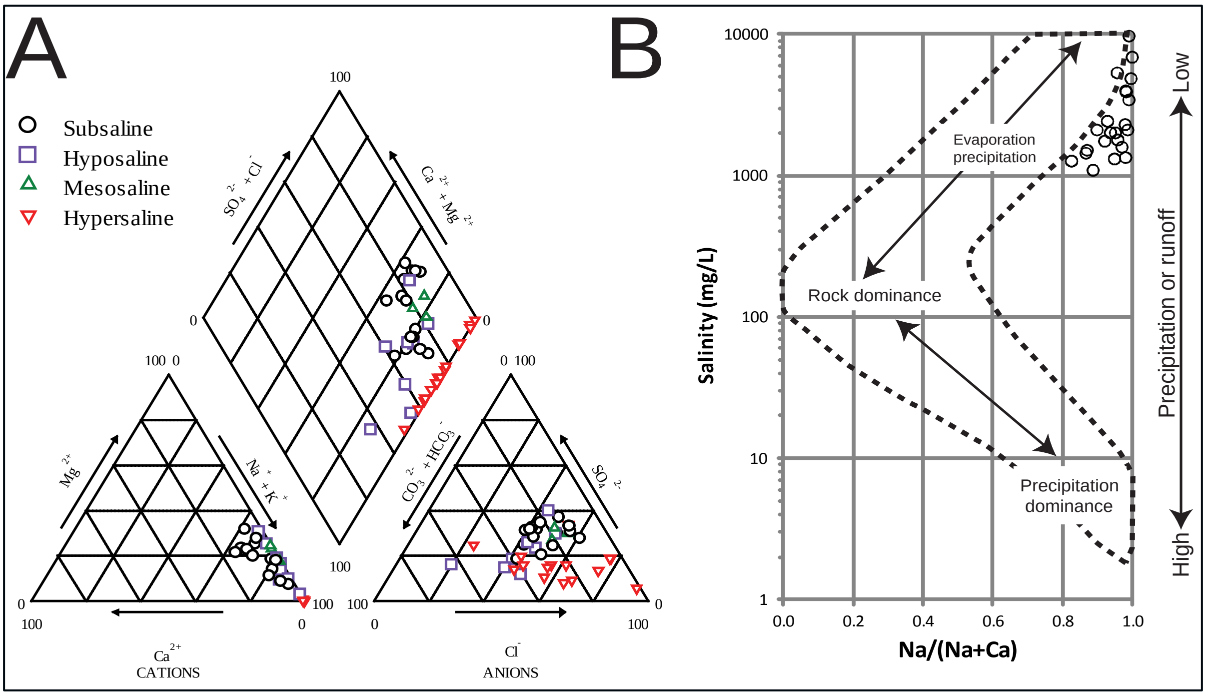

Figure 1: Chemistry of water samples from the 42 lakes of the Badain Jaran dataset: (A) Piper diagram and (B) Gibb's plot in which the dashed line represents the boomerang-shape envelope described by Gibbs (1970) for surface waters from various regions of the world.

Diatoms were only present in 26 of the 42 lakes sampled. The 16 lakes with surface sediment devoid of preserved diatoms were the most saline and alkaline in the training set with values for EC > 50,000 µS cm^{-1}, salinity >80 g L-1 and pH >10. The 16 lakes were only considered in the analyses of the environmental data (exploratory PCA, Piper diagram), but not included in further analyses. The first two axes of the PCA on the 42 lakes explained 49.6 and 14.1% of the total variance, respectively (Figure 2A). The first axis is highly correlated with salinity, EC and all major ionic concentrations, and separates the hypersaline lakes (>50 g L^{-1}) from the less saline lakes. For the PCA on 26 lakes with diatoms, with the hypersaline excluded (Figure 2B), the first two axes explain 38.9 and 24.6% of the total variance, respectively. As in the previous PCA, axis 1 also reflects the major gradient of salinity and separated the mesosaline and hyposaline lakes from the subsaline lakes. None of the 26 lakes was identified as a clear outlier.

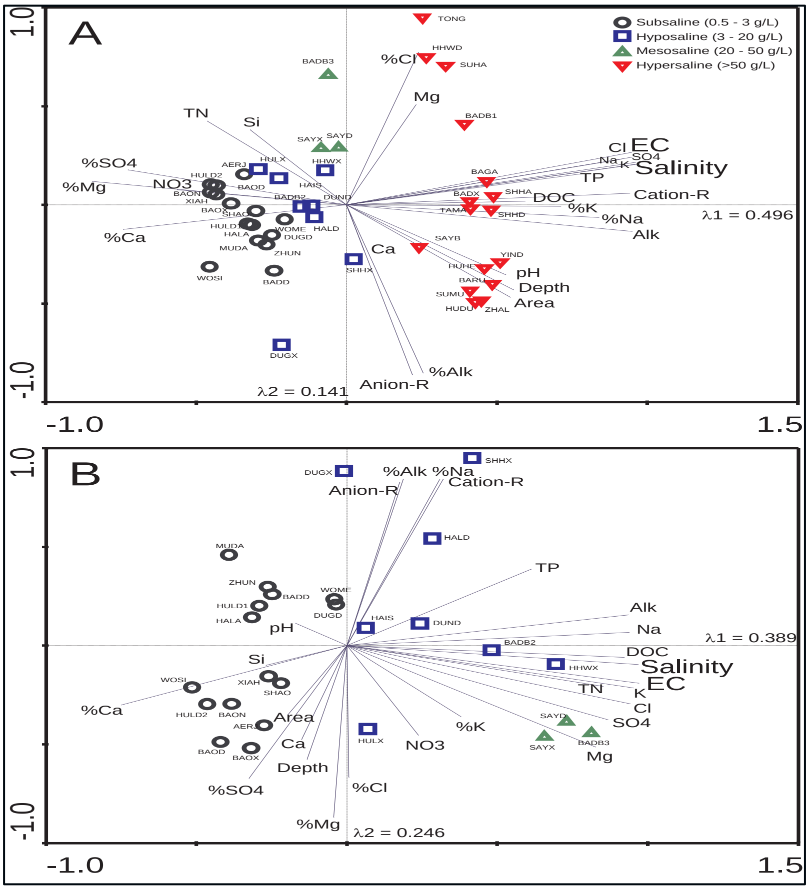

Figure 2: Principal component analysis ordination biplot on axes 1 and 2 of A) the 42-lake dataset and B) the 26-lake dataset (with diatoms present in surface sediment). Lakes are marked with open symbols and environmental variables with arrows. The salinity ranges follow the classification of Hammer et al. (1983).

4.2 Diatom assemblages

After transformation of the diatom counts to relative percentages, taxa that could not be resolved to the lowest taxonomic designation possible (species or variety) were excluded from further analysis. This left a total of 118 species identified in the 26 surface-sediment samples analysed.

Hierarchical cluster analysis of the diatom assemblages is shown in the form of a dendrogram (Figure 3). Using the rule of thumb given earlier ($k = \sqrt{26/2} = 3.6$), 3 or 4 clusters should be determined from the diatom dataset. As box plots (Figure 4) and NMDS (Figure 5) clearly reveal 3 clusters, this number of cluster was also retained for HCA. This segregation is mainly defined by the EC gradient as shown by box plots (Figure 4). On the right side of the HCA dendrogram, cluster 1 includes 15 subsaline lakes. On the left-hand side, cluster 2 regroups the five most saline lakes in the dataset, ranging from hyposaline to mesosaline. In between, cluster 3 includes six hyposaline lakes of intermediate salinity. The subsaline lakes (cluster 1) also tend to be less eutrophic than the hypo- and mesosalines lakes included in clusters 2 and 3. There is no significance difference in water depth between the clusters as all lakes investigated are shallow (Figure 4). The validity of these clusters is confirmed by the MRPP test, which shows that the between-group differences are significant (A-statistic = 0.244, p-value <0.001%). The NMDS axis 1 represents the EC gradient as shown by the fitted surface and vector (Figure 5).

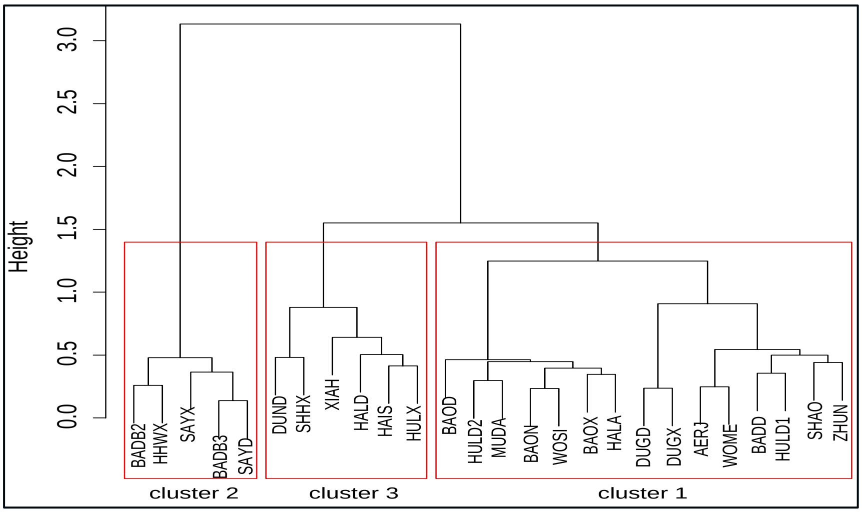

Figure 3: Dendrogram resulting from the hierarchical cluster analysis (Ward's linkage; Bray-Curtis distance) of diatom assemblages from the 26 surface-sediment samples with preserved diatoms. using measure.

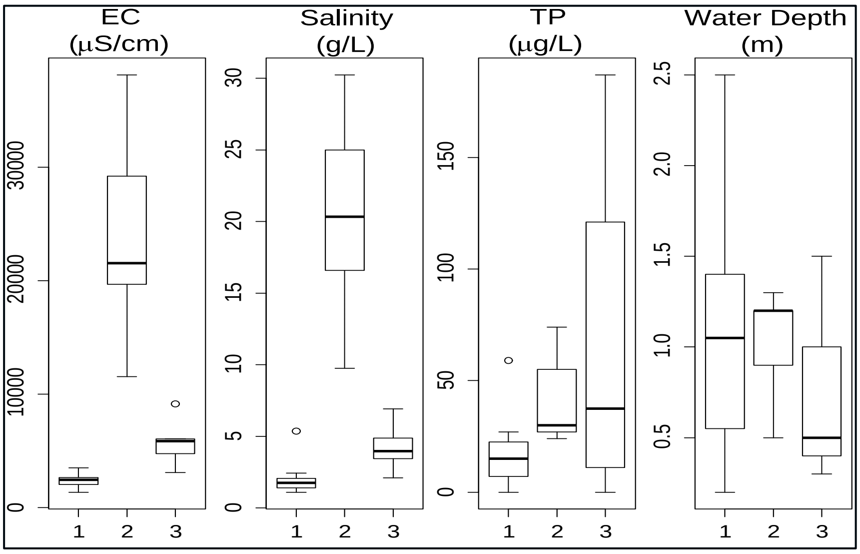

Figure 4: Box plots showing the median values and quantile distributions of selected environmental variables in lakes of the three groups identified by hierarchical cluster analysis. The 25-75 percent quantiles (excluding outliers) are drawn using a box. The median is shown with a horizontal line inside the box. The whiskers represent the upper and lower "inner fence", i.e. are drawn from the edge of the box up to the largest/lowest data point less than 1.5 times the box height. Outliers, i.e. values outside the inner fences, are shown as circles if they lie further from the edge of the box than 3 times the box height.

Species indicator values analysis reveals that 21 species have a significant score (IndVals > 50%, P<0.5) and are good indicator species of each cluster of lakes defined by HCA and NMDS. The main representative taxa for the subsaline lakes in cluster 1 are *Encyonopsis descripta* var. *asymmetrica* Krammer, *Nitzschia denticula* Grunow, E. krammeri, and Achnanthidium caledonicum (Lange-Bertalot) Lange-Bertalot. For the mesosaline lakes in cluster 2 the main representative species are *Seminavis pusilla*, *Nitzschia prolongata* Hustedt, *Brachysira aponina* Kützing and *Proschkinia complanata* (Grunow) D.G. Mann. In cluster 3, the best indicator taxa are *Halamphora subcapitata* (Kisselew) Levkov, *Entomoneis paludosa* var. *subsalina* Cleve, *Navicula cincta* (Ehrenberg) Ralfs and *Nitzschia elegantula* Grunow.

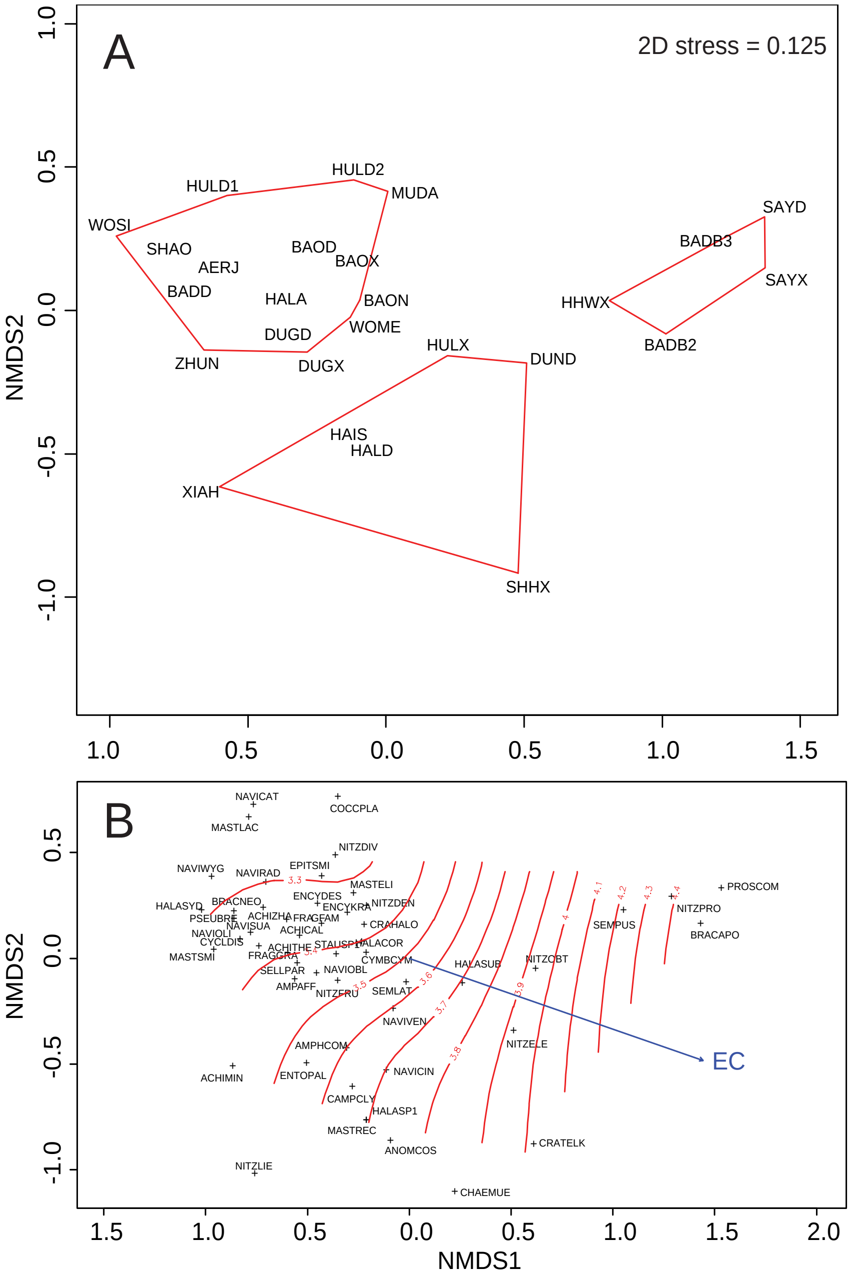

Figure 5: Non-metric multidimensional scaling (NMDS) ordination plot of Badain Jaran surface-sediment samples according to their diatom assemblage composition showing plot of the sites (A),

and plot of the main diatom species with fitted vector and surfaces for EC, the main environmental variable (B). The distance matrix was calculated based on the Bray-Curtis algorithm.

The relationship between the EC gradient and diatom species richness, as expressed by the values of expected richness E(Sn) that were calculated by rarefaction analysis, is displayed in Figure 6.

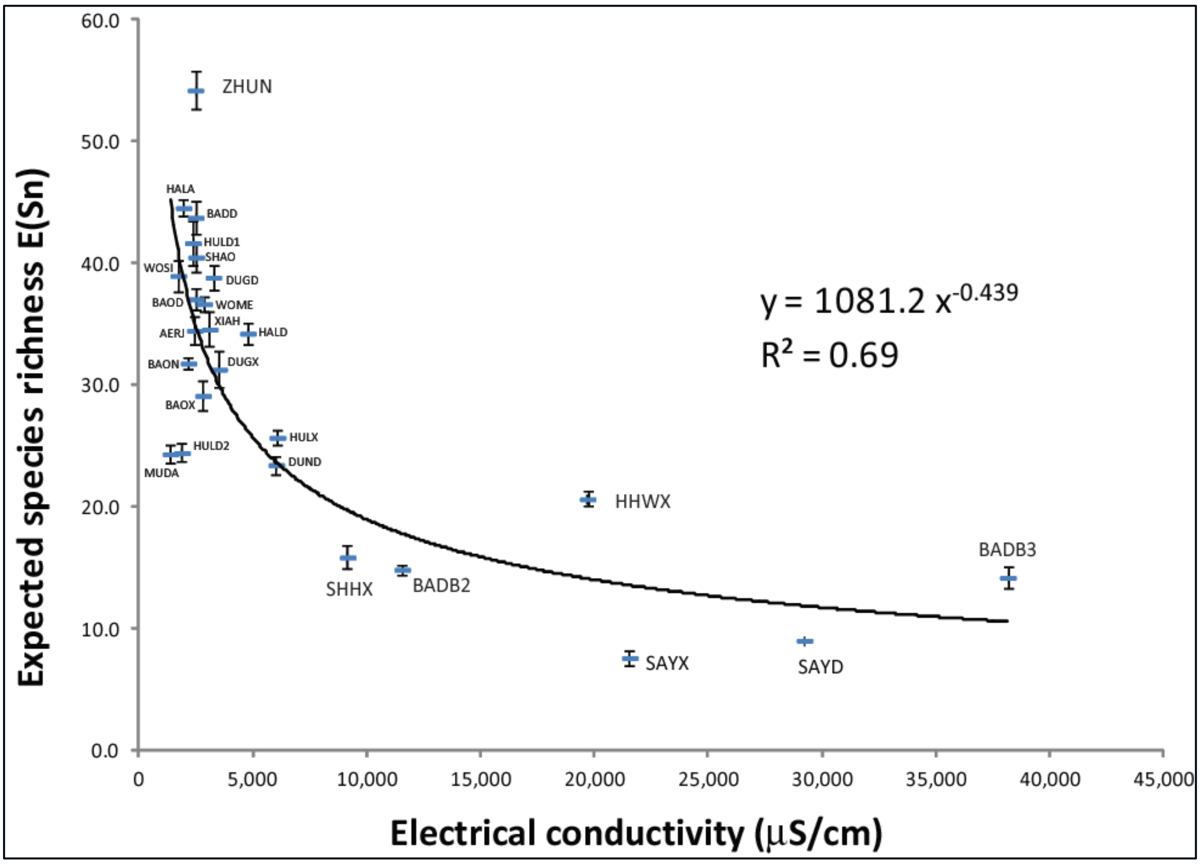

Figure 6: Regression between Electrical conductivity and expected species richness, E(Sn), in surface sediment samples from the Badain Jaran Desert. The values of E(Sn) and their standard error were calculated by rarefaction analysis. The lowest diatom count was n =494 as the site HAIS with low diatom count was excluded from the analysis.

4.3 Discussion and conclusion

The ionic composition compositions of the lakes investigated were very similar, dominated by chloride and controlled by evaporation. PCA showed that the main environmental gradient was EC, i.e. the overall ion concentration. This homogeneity in water chemistry may explain the overriding influence of EC on the distribution of diatoms in the lakes of the Badain Jaran Desert. A similar situation was observed in Western Victoria, Australia (Blinn 1995) but it contrasts with other studies on saline lakes, for which brine composition was found to be important (e.g. Gasse et al., 1995, Wilson et al., 1994).

Several of the best indicator taxa for the subsaline lakes (cluster 1) have been found associated with alkaline springs. The type populations of *E. descripta* var. *asymmetrica* and *E. krammeri* were described from alkaline springs in Germany (Krammer 1997; Reichardt 1997). *E. Krammeri* is also common in karstic springs on Majorca Island (Delgado et al. 2013) and in carbonate and tufa springs of the Italian Alps (Cantonati et al., 2012). A caledonicum is an attached form reported as common in springs and seepage areas (Wojtal et al., 2011). *Achnanthidium thermale* Rabenhorst is often associated with alkaline springs (Hofmann et al., 2011). *Nitzschia denticula* (synonym of *Denticula kuetzingii* Grunow) is reported as an epiphytic or epipelic form in various biotopes including springs (Gasse, 1986) and was also found in carbonate and tufa springs of the Italian Alps (Cantonati et al. 2012). Besides these spring-associated species, the other type-specific diatoms for cluster 1 (Table 1) illustrate the high diversity of habitats available in subsaline lakes, as they include epiphytic (*Cymbella cymbiformis* var. *nonpunctata sensu* Cumming et al. 1995), tychoplanktonic (*Staurosira* sp), planktonic (*Cyclotella distinguenda* Hustedt) and several epipelic species (*Brachysira neoexilis* Lange-Bertalot, *Mastogloia elliptica* (*C. agardh*) Cleve, *Navicula oligotraphenta* Lange-Bertalot & Hofmann, Nitzschia cf frustulum and *Sellaphora blackfordensis* Mann & Droop). The diversity of microhabitats is particularly enhanced when a large area of the lake is colonised by macrophytes such as *Potamogeton crispus* L., *Stuckenia pectinatus* (L.) Börner and *Chara* sp. and with the development of reed beds (*Phragmites*) by the shore. Diatom assemblages therefore include heliophilic species (such as *N. denticula*, *Brachysira* and *Achnanthidium*) and shade-tolerant (= sciaphilic) species such as *Encyonopsis* and *Cymbella* (Stevenson et al., 1991).

Interestingly, an assemblage with similar species composition was reported from a shallow, Chara-dominated and periodically drying lake from Western Anatolia (turkey) by Barinova et al. (2014).

In the hyposaline lakes of cluster 3 the type-specific indicator taxa are *E. paludosa* var. *subsalina*, *H. subcapitata*, *N. cincta* and *N. elegantula. E. paludosa* var. *subsalina* is an epipelic, brackish water species. *Halamphora subcapitata* is widely distributed in saline lakes and mineral springs (Levkov, 2009). In British Columbia, Canada, its distribution was similar to the one observed in the Badain Jaran desert lakes as it also ranged from subsaline to hyposaline lakes (Cumming et al., 1995). *N. elegantula* is an epiphytic and epipelic, cosmopolitan species inhabiting marine coasts and inland waters with high electrolyte content (Lange & Tiffany, 2002; Snoeijs & Potapova, 1995, Witkowski et al., 2000). It is a species characteristic of chloride dominated waters in Africa (Gasse et al. 1995) and Spain (Reed, 1998). *Navicula cincta* is a cosmopolitan species, common in electrolyte-rich fresh- to brackish water and classified as halophilic, alkaliphilic and epipelic (Antoniades et al., 2008; Gasse 1986; Witkowski et al., 2000). By contrast with our study, *N. cincta* was reported as characteristic of hypersaline lakes in Saskatchewan, Canada (Hammer et al., 1983) and in Western Victoria, Australia (Blinn, 1995). This illustrates the fact that the apparent optima and tolerance ranges of a species in a particular dataset is influenced by the range and spread of values in that dataset. It is possible that the range of *N. cincta* was underestimated in our dataset in which subsaline lakes dominate. On the whole, the indicator species for cluster 3 have lower specificity than those in the other two clusters (Table 1). This indicates that this “intermediate” group is less well-defined than the other two.

The type-specific taxa of mesosaline lakes (cluster 2) are *B. aponina*, *N. prolongata*, *P. complanata* and *S. pusilla*. *B. aponina* is a cosmopolitan, epipelic, halophil species (Snoeijs, 1993, Wolfe and Kling, 2001). *N. prolongata* is a marine to brackish-water species, probably cosmopolitan (Witkowski et al. 2000). *P. complanata*, is marine, probably cosmopolitan (Witkowski et al. 2000). *S. pusilla* is a brackish epipelic species (Snoeijs, 1993). The diatom diversity is much lower in these lakes. This trend probably reflects not only the increasing salinity but also the lower diversity of habitats available for diatoms to develop. In particular the macrophyte beds found in the subsaline and hyposaline lakes (clusters 1 and 3) are far less developed in the mesosaline lakes (cluster 2) leaving a much more homogenous environment dominated by epipelic forms.

In general, the specificity of indicator diatom species for each group of lakes was high showing that it is possible to identify and rely on these species as indicators of each type of lakes. However, only three species reached specificities of 1, i.e. 100%, namely *E. descripta* var. *asymmetrica* and *Mastogloia elliptica* (*C. agardh*) Cleve in cluster 1 and *P. complanata* in cluster 2 (Table 1). This suggests that even indicator species that show a strong preference for one type of lake can be found in the other types of lake and are therefore tolerant of a fairly wide range of salinity (EC) conditions. This confirms the findings of Potapova (2011) who demonstrated that there are no discontinuities in diatoms species turnover along the salinity gradient.

In our dataset diatom diversity decreases as salinity increase (Figure 6). A similar relationship was observed for diatoms in saline lakes from Western Victoria, Australia (Blinn, 1995) and western North America (Blinn, 1993), and fits with the general relationship between salinity and species richness observed for all biota of salt lakes described by Hammer (1986) in which the number of species is greatest in subsaline waters, decreases dramatically at low salinities and then continues to decrease gradually with increasing salinity. From these results, it may be deduced that if the subsaline lakes were to become more saline because of climate change or human activities (e.g. water extraction) a significant loss of diversity will occur.

Recent climate warming is clearly recorded for this region (Piao et al., 2010). Despite this on-going trend of rising temperature, no large change in diatom-inferred EC and lake area were inferred from palaeolimnological and remote sensing data over the past few decades (Rioual et al., 2013). The reason for this stability is that local atmospheric precipitation makes only a small contribution to recharging the groundwater system of the Badain Jaran Desert (Ma et al. 2014) and that the main recharge source of lake water is groundwater (Wu et al., 2014). This would suggest that these lakes are resilient to short-term natural fluctuations in climate. However, as show for another arid area of northern China (Li et al., 2014), these groundwater-dominated systems are very sensitive to any drawdown of groundwater caused by human activities. The protection of these unique lakes therefore depends on a sustainable economic development of the region that does not cause excessive extraction of groundwater for domestic water use and irrigation.

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Environmental control on testate amoebae communities and local transfer functions in a Sphagnum-peatland in China

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Abstract

Testate amoebae are a diverse and abundant group of soil protozoa that constitute a large proportion of microbial biomass in many ecosystems, and probably play important roles in ecosystem functioning. These microorganisms have attracted the interest of palaeoecologists because the preserved shells of testate amoebae and the known hydrological preferences of many taxa allow the reconstruction of past hydrological change. In ombrotrophic peatlands surface wetness reflects hydroclimate, so testate amoebae play an increasingly important role in the reconstruction of Holocene climate change. Previous studies, however, have been geographically restricted, mostly to North America and Europe. We studied the ecology of testate amoebae in peatlands from central China in relation to hydrology, pH and metal concentrations. We found that the testate amoeba community structure was correlated with depth to water table (DWT), and that the hydrological preferences of species generally matched those of previous studies. We developed a weighted average DWT transfer function that allows the prediction of water table depth with a cross-validated mean error of less than 5 cm. Our results demonstrate the potential for testate amoebae to be used for palaeohydrological reconstruction in China. Such studies could contribute to our understanding of the Holocene climatic changes in China, particularly regarding past Asian monsoon activity.

Keywords

Testate amoebae, Ecology, Hydrology, Transfer function, Peatland, Middle Yangtze Reach, China

1. Introduction

As one of the most important ecosystems, peatlands play key roles in biodiversity conservation, carbon cycling, climate modifying and maintenance of water resources. Hydrological condition is one of the important factors influencing the health and development of peatland. The Dajiuhu peatlands located near the middle reach of the Yangtze River, are the largest peatlands in central China, which were rapidly expanded from the late Pleistocene period to the early Holocene when the monsoon driven hydrological changes were occurring since the last deglaciation (Zhao et al., 2007, Zhu et al., 2009, Xie et al., 2013). Abundance of a range of hydro-climatic proxies have been used for reconstructing the history of the peatlands in the region. However, researches on some micro-fossils like testate amoebae are still relatively scarce.

Testate amoebae are a group of Protozon which are highly abundant in the surface of Sphagnum-dominated peatlands. Each testate amoeba has an organic shell in which the single cell is living. As there are distinct morphological characters, short life cycles and sensitive to environmental gradients, they have been widely used as an excellent proxy for ecological and hydro-climatic changes.

This paper aims to investigate the testate amoebae diversity and distributions in the Dajiuhu peatlands, and infer their relationships to environmental variables. A local testate amoebae-based hydrological transfer function was developed for the future palaeohydrological reconstructions.

2. Methods and materials

The Dajiuhu peatlands (E109°56′-110°11′, N 31°24′-31°33′) was located at Shennongjia Mountains in central China. Where the region was also thought to be one of the most important refugias for biodiversity (Zhang et al. 2007), a total of 48 points were selected to reflect the full characteristics of the habitat types in the Dajiuhu peatland in early August 2012. A hole was drilled at each point for testing the depth water table (DTW). In order to obtain a robust value of water table, we tested the DTW after 3 days in order to a fully equilibration of the water level in the hole. Other environmental variables like pH, conductivity and dissolved oxygen (DO) were also tested in the field.

The top 0-5 cm Surface Sphagnum moss was collected in plastic bags for testate amoebae analysis. In the laboratory, Sphagnum samples were boiled for 5 minutes, and then sieved at 250 and 15 μm with the intermediate material retained. In this study, some fine organic materials were removed using the 15 μm sieve which could underestimate the abundance of smaller taxon (Payne and Mitchell, 2009). However, the 250 and 15 μm sieve can result in most testate amoebae population characters and community structure.

The material retained in the 15 μm sieve was centrifuged at 3000 r/min for five minutes. The residues were stained with Safranine for microscopic analysis. We identified and counted at least 150 individual testate amoeba shells for each sample (Payne and Mitchell, 2009) based on taxonomy given by Charman et al. (2000), Penard (1902) and Meisterfeld (2002).

A total of 33 samples were finally selected as due to low abundance of testate amoebae shells for appropriate numerical analysis.

Testate amoebae percentage data were square-root transformed. In order to evaluate the relations between testate amoebae community and environmental variables, we calculated the Shannon–Wiener diversity index (SDI) as follows:

$$S.I. = -\sum_{i=1}^{S}\left(\frac{X_i}{N_i}\right)\times \ln\left(\frac{X_i}{N_i}\right) \quad (1)$$

Where X_i is the abundance of each taxon in a sample, N_i is the total abundance of the sample, and S is equal to the species richness of the sample (Shannon and Weaver, 1949). The SDI values usually fall between 1.5 and 3.5 (Margalef, 1972).

The Canonical Correspondence Analysis (CCA) ordination was applied to investigate the environmental controls on testate amoeba communities using CANOCO version 4.53 (Lepš and Šmilauer, 2003).

As the ordination results showing DWT has a strong controlling on testate amoebae community, we therefore developed an amoebae-based hydrological transfer function using simple weighted-averaging (WA) technique, WA Tol_Inv, WA Tol_Cla, and weighted-average partial least squares (WA-PLS) approaches (ter Braak and Looman, 1986; ter Braak and Barendregt, 1986, (ML, Birks 1995). The testate amoeba transfer function performance has conventionally been determined using leave one out (jack-knifing) and boot-strapping. The transfer function analyses were carried out in C2 package (Juggins 2010).

3. Results and discussion

3.1 Testate amoebae diversity and biogeography

There was a total of 33 testate amoebae taxon and types in this study, including a new species *Nebela jiuhuensis*. Four taxa (*Nebela barbata*, *N. jiuhuensis*, *N. collaris* and *Pontigulasia compressa*) were removed before the statistical analysis due to their low abundance in the sample (Figure 1). The most abundant taxa is *Assulina muscorum* which appeared in most samples. Most testate amoebae taxa in this study are cosmopolitan or ubiquitous distribution. However, the taxa *Archerella* (*Amphitrema*) *flavum* which are very abundant in boreal peatlands is missed in this study. Another taxa *Hyalosphenia papilio*, a major testate amoeba taxon of *Sphagnum* peatlands, only reported with low abundance in the present site but was not found this time. Some taxa like *Argynnia dentistoma* and *A. caudate* were very limited distribution in China and they have been thought to be belonging to genus *Nebela* in the Chinese testate amoebae lists until recently (Qin et al., 2011). Other tata *Nebela barbata* was reported in freshwater lakes in China (Shen, 1983) also appeared in this study.

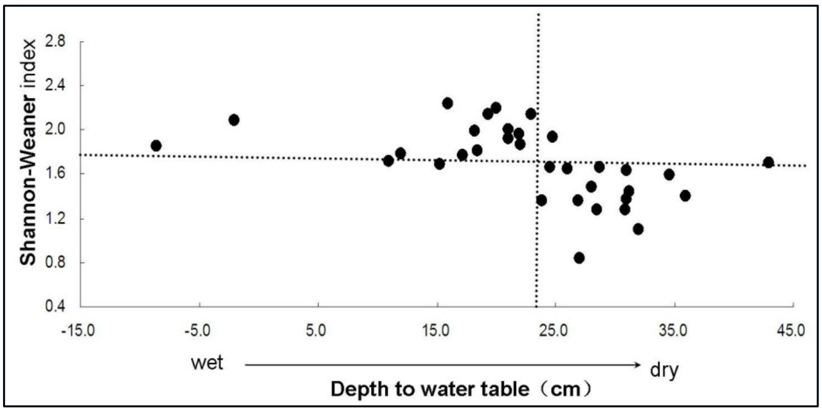


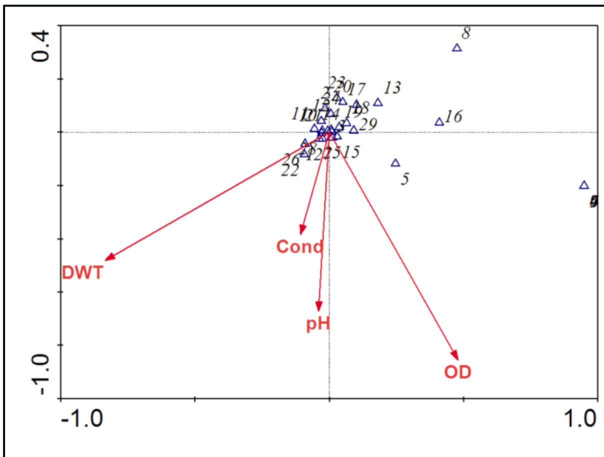
Figure 1: Shannon-Weaner index of testate amoebae along the water depth gradient of the peatland.

The reasons for testate amoebae geographical distribution patterns could be complicated (Bobrov et al., 2014). The wetlands type, temperature gradient, environmental conditions, and geographic barriers could be the reasons for the missing taxon. However, the relatively limited investigations on peatlands testate amoebae in China makes a small TA data set, so it is too early to draw the conclusion if these taxa are endemics or cosmopolitan (Qin et al., 2011, 2013)

3.2 Testate amoebae community and environmental relations

Shannon Diversity Index (SDI) showed that diversity of testate amoebae community changing along the water depth. The SDI value is decreasing with the wet to dry habitats gradient, especially became relative to low after the DTW deeper that 25-30cm. This result indicated that the SDI value can be a potential indicator for peatland hydrology.

The CCA results showed that DWT, OD, conductivity and pH have influenced the testate amoebae community compositions, in which DTW is the controlling factor (p=0.002) (Table 1, Figure 2). The first two axes explained 84.6% environmental information, with axis 1 explaining 61.2% (Table 2). This suggests that the measured environmental variables include the main hydrochemical factors to which TA respond. In the CCA ordination diagram (Figure 2) species distributions are greatly different along the hydrological gradient. Taxon like *Assulina muscorum*, *Assulina muscorum*, *Corythion dublium* and *Trigonopyxis arcula* are positively correlated with DWT while *Aygynnia caudada*, *A. dentistoma*, *Arcella catinus* type, *Centropyxis cassis* type, *C. aculeata* type, *Difflugia acuminata*, *D. lanceolata*, *D. oblonga* type, *D. pristis* type, *Euglypha rotunda* type, *E. strigosa* type, *E. tuberculata* type, *Quadruella symmetrica* are negatively correlated with DWT. The general distribution of testate amoebae along the DWT gradient reported here is consistent with previous studies in highlighting the role played by peatlands hydrology or surface wetness in the ecological studies of testate amoebae.



Variable	F
DWT	4.5**
DO	3.22*
Cond	0.75
pH	0.68

Figure 2: CCA analysis shows that depth to water table (DWT) is the controlling factor to testate amoebae communities (1. *Arcella catinus*, 2. *Assulina muscorum*, 3. *Centropyxis cassis*, 4. *C. aculeate*, 5. *Cyclopyxis arcelloides-type*, 6. *Difflugia acuminate*, 7. *D. lanceolata*. 8. *D. oblonga*, 9. *D. pristis* type, 10. *Euglypha rotunda* type, 11. *E. strigosa* type, 12. *E. tuberculata* type, 13. *Habrotrocha angusticollis*, 14. *Heleopera sphagni*, 15. *Hyalosphenia subflava*,16. *Lesquereusia modesta*, 17. *Argynnia dentistoma*, 18. *Nebela lageniformis*, 19. *N. militaris*, 20. *N. penardiana*, 21. *N. tincta*, 22. *Plagiopyxis callida*, 23. *Quadruella symmetrica*, 24. *Sphenoderia lenta*, 25. *Trigonopyxis arcula*, 26. *Trinema linare*, 27. *Picea stomata*, 28. *Argynnia caudata*, 29. *Corythion dublium*).

Table 1: CCA result shows the relations of testate amobae and environmental gradient

Axis	***1***	***2***	***3***	***4***
Eigenvalue	0.144	0.055	0.026	0.011
Species-environment relations	0.765	0.686	0.679	0.662
Cumulative percentage variance of species data (%)	16.4	22.6	25.6	26.8
Cumulative percentage variance of species-environment relation (%)	61.2	84.6	95.5	100.0
Sum of all eigenvalues	0.879			

Table 2: Hydrological transfer functions development of testate amoebae by using different, weighted averaging methods

Models	***RJack2***	***RMSEPJack***	***Rboot2***	***RMSEPboot***
WA_Inv	0.54	7.45	0.46	8.89
WA_Cla	0.56	6.78	0.5	9.27
'WA Tol_Inv	0.39	8.94	0.34	9.32
WA Tol_Cla	0.45	8.22	0.38	9.61
WA_PLS (1)	0.34	8.6	0.28	9.35
WA_PLS (2)	0.3	8.56	0.19	9.57
WA_PLS (3)	0.22	8.99	0.16	10.28
WA_PLS (4)	0.19	9.2	0.15	11.44
WA_PLS (5)	0.18	9.46	0.15	13.46

3.3 Testate amoebae-based hydrological transfer function development

The community composition patterns indicated that testate amoebae are sensitive to the surface moisture of the peatland. Therefore, we further developed a species hydrological transfer functions (Figure 3). The models were compared by the regression R^2 value between observed and model estimated values and RMSEP. The WA Cla model provided advantage over other models, as its performance for DWT was relatively good with an RMSEPJack of 6.7 cm (Figure 3). The previous studies suggested that the WAPLS model was the best model as it could support the best performance (Booth, 2007). However, the WA transfer function developed in this study would be used for the local hydrological reconstructions during the Holocene. The leave one out (jack-knifing) and boot-strapping are two ways to determine the different models. It is therefore suggested that jack-knifing has a better performance.

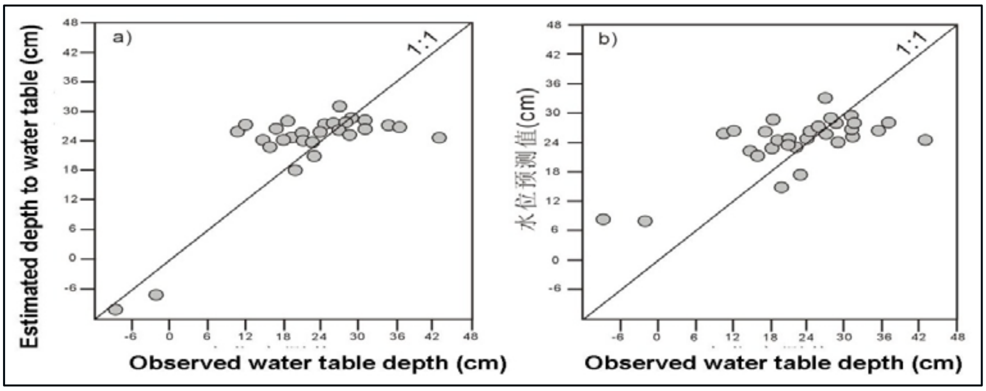


Figure 3: Hydrological transfer functions of testate amoebae and depth to water table, using Jack-knifing (a) and bootstrapping (b) cross-validations

3.4 Other influences on testate amoebae community

In addition to hydrology, the ordination also showed possible effects of other environmental variables on testate amoebae community composition, Such as pH, conductivity and dissolved oxygen. This was reflected also in previous studies on the fine-scale patterns of testate amoeba communities in relation to micro-topography in Sphagnum peatlands (Mitchell et al., 2000, Qin et al., 2013).

In peatlands ecological studies of testate amoebae and other micro-organisms (such as diatom) show that depth to water table is generally the most important environmental variable, while pH is often the secondary significant variable. At the present site, testate amoeba community composition show some relationships with pH, conductivity and dissolved oxygen. The close relationship between testate amoebae and other environmental variables suggest a complex response of testate amoebae in a finer scale assessment (Lamentowicz et al., 2010, Qin et al., 2013).

4. Conclusions

Testate amoebae are a group of freshwater protozons, which are sensitive to environmental change and therefore have been used widely in paleoenvironmental reconstruction. This study investigated the response of testate amoebae diversity and community composition to environmental variable in Dajiuhu peatlands, as it was necessary to to understand better the patterns of testate amoebae ecology in peatlands across Central China. A local hydrological transfer function was developed, and would be used for local paleoenvironmental reconstruction during the Holocene.

Acknowledgement

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Developing and applying two methods that use subfossil Australian chironomid (non-biting midge) as proxies for past climate and environmental change

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Abstract

Methods that will use the fossilised remains of non-biting midge larvae (chironomids) preserved in lake sediments to reconstruct past changes in the Australian climate and freshwater lake system are under development. The first method will create a model (transfer-function) to reconstruct past summer temperatures and lake trophic conditions based on the temperature and nutrient level tolerance of Australian chironomid species living in south Australian lakes today. The second method will be based on the stable oxygen and deuterium isotope composition ($\delta^{18}O$ and δD) of the chironomid head capsules (HCs) from South Australia.

Previous studies have shown that the fossilised HCs of non-biting midge larvae act as a 'time capsule' that preserves the stable isotope of the lake water in which they live. A temperature effect will be one of the most important controls on lake water $\delta^{18}O$ in southern Australia. Therefore I will be able to use $\delta^{18}O$ from fossilised chironomid HCs as another method for reconstructing past changes in temperature. Deuterium (δD) from chironomid HCs can possibly be used for nutrient relationship inference however, this has not been explored. Both of these methods will be applied to chironomid remains extracted from lake sediment deposits in southern Australia. This project will be the first to develop a chironomid stable isotope method for reconstructing past conditions in the Southern Hemisphere, and the first worldwide representation on the use of both chironomid transfer function stable isotope methods from the same sites.

The development and application of these new proxies will help us to understand the impact by humans and climate change on lake and wetland system and further, to develop strategies for management and restoration of freshwater bodies for promoting ecological resilience.

Keywords
Subfossil chironomids, Australia, temperature transfer functions, climate change

1. Introduction

Application of proxies that have the potential to provide long-term and reliable palaeoclimate and environment records is a key to better reconstructing and understanding the past environmental change. Ideally these proxies should be widely distributed and commonly available in the environment. Chironomids (non-biting midges) occur in virtually all permanent and semi-permanent terrestrial water bodies. There are four larval stages of chironomids (from 1st instar to 4th instar), at the end of each instar, a chitinous subfossil head capsule is shed and preserved in the lake sediments. The development time is influenced by the combination of temperature and food availability. The pupal stage usually lasts 3-4 days and the timing of emergence is dependent on water temperature and light intensity. Due to this short life cycle, chironomids can respond fast to any changes and provide a snapshot of the local environmental condition. In this study, we aim to explore the potential applications of the subfossils of chironomid HCs in past environmental change reconstructions of wetlands across the southeast Australia.

2. Methods

We sampled 45 natural and artificial freshwater bodies across the east and south east Australia (Figure 1) for this study. The transect covers a distance of 3,700 km along the east coast of Australia from Crater Lakes National Park, Queensland to Mount Field National Park, Tasmania (17.24˚S to 42.67˚S, 140.18˚E to 153.26˚E) (Figure 1). The climate ranges from monsoonal tropical in the north, to cool temperate in the south and hence there are large temperature and precipitation gradients in the data sets. Elevation of the sites ranges from sea level to 2,048 m above the mean sea level (a.s.l). The data set comprised 29 natural lakes and 16 artificial water bodies across the eastern and south eastern coasts of Australia.

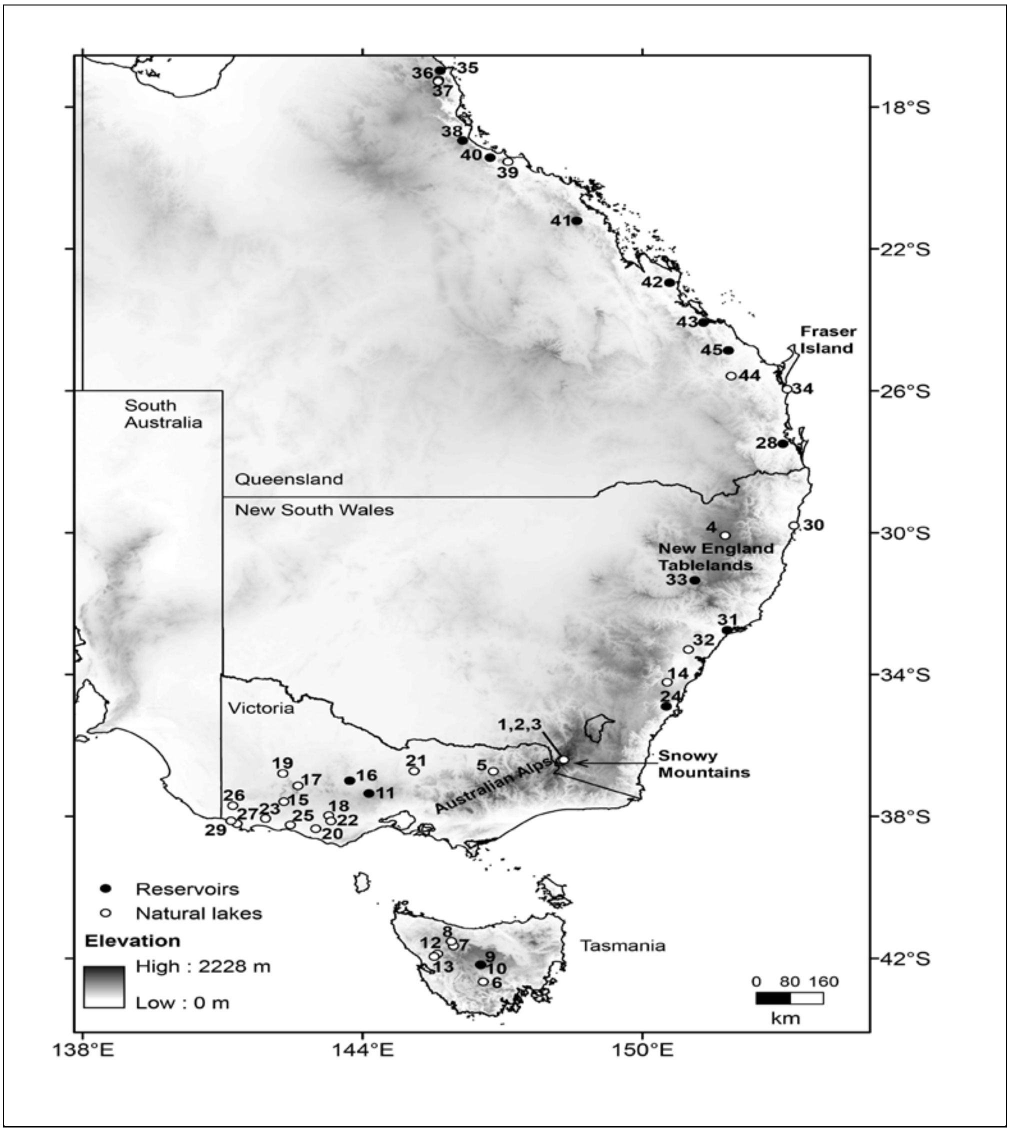

Figure 1: Map of eastern Australia with the 45 study sites identified

3. A limnology survey of freshwater system of eastern Australia

Prior to analyse the chironomid species, a limnology survey of 45 natural and artificial water bodies extending across the whole of eastern Australia from the tropics to Tasmania was conducted to examine the land-use, climate, limnology nexus (Chang et al., In press). A broad variety of physio-chemical, land-use and climatic parameters were measured. Reservoirs and other artificial water bodies respond to stressors in their catchments in a similar fashion to natural lakes but tend to be less nutrient rich possibly due to shorter residence times and active management. pH and salinity are strongly correlated in the dataset. Bedrock has a strong influence on pH in freshwater lakes but all highly saline lakes are alkaline, irrespective of bedrock. High concentrations of anions in saline lakes preclude the existence of acid conditions by binding available hydrogen ions. Almost all lakes fall on salinity axes that indicate marine origin for their salts. An assessment of the TN:TP molar ratios from the lakes in this dataset indicates that productivity in Australian lakes could be limited by both nitrogen and phosphorus. Future research using macro-nutrient enrichment experiments should be pursued to confirm this preliminary observation. There is a strong positive correlation between regional aridity and lake eutrophication. This is typical of semi-arid and seasonally arid environments and reflects the concentration of nutrients due to evaporative flux in shallow basins with high residence times.

4. Chironomid transfer functions to reconstruct palaeoclimate and environment

A transfer function based on the modern distribution of chironomids (Diptera: Chironomidae) species to model mean February temperature in southeast Australia was developed (Chang et al., under review). The chironomid HCs were used and the transfer function is based on chironomid species assemblages along with 18 environmental and climatic variables sampled from 33 natural and artificial lakes range from subtropical Queensland to cold temperate Tasmania (northern Queensland lakes are excluded). Multivariate statistical analyses (CCA, pCCA) were used to study the distribution of chironomids in relation to the environmental and climatic variables. Eight variables were found significant ($P<0.05$) for explaining variance, these are precipitation (12%), mean February temperature (9.5%), pH (9.5%), specific conductance (8.2%), total phosphorous (8%), potential evapotranspiration (8%), chlorophyll-a (6.9%) and depth (6.2%). Further pCCA analyses shown mean February temperature (MFT) is the least entangled and most robust variable explained by chironomid species distribution. MFT was modelled in C2 software, using partial least squares (PLS) and generated a transfer function with coefficient of determination, $r2$ jackknifed= 0.69, root mean squared error of prediction, RMSEP = 2.3˚C and maximum bias of 2.15°C.

The first chironomid-based reconstructions of mean February temperature applying this transfer function for Blue Lake, Mount Kosciuszko National Park using data published in Schakau (1993) is also conducted. The purpose of this application is mainly for model validation. The reconstruction shows an overall pattern with high correlation to the Milankovitch driven summer insolation curve and the chironomid based summer temperature reconstruction from Eagle Tarn and Platypus Tarn, Tasmania (Rees and Cwynar 2010) and the vegetation history reconstructed based on pollen from the Kosciuszko area.

There is a great potential to develop a chironomid based nutrient transfer function as total phosphorous appeared as the second significant variable that controlling the species distribution. Re-integration of northern Queensland lakes will make the model more robust and applicable to a broad variety of environment. This work will be carried out shortly. This will provide a new method for lake trophic status reconstruction in additional to diatoms (Tibby 2004) and cladoceran (Kattel and Augustinus 2010) from Australasia.

5. Chironomid stable isotopes as an innovative approach in palaeoenvironmental studies

Growth of chironomids is strongly controlled by water temperature. Chironomids have a chitinous head capsule that we can measure stable isotopes on them. Wooller et al (2004) demonstrate a positive correlation between the mean annual air temperature above a lake and $\delta^{18}O$ of the chironomids in the lake. Therefore, chironomids may be used as a proxy for modern and paleo-water temperature. The modern range of chironomid $\delta\ ^{18}O$ values will be developed based on 33 lake surface sediment sampled from south-eastern Australia (same sites as the transfer functions). For these lakes, HCs of single taxa will be picked to avoid complications from 'vital effects'. These data will serve to establish the relationship of $\delta^{18}O$ to modern lake temperatures. Deuterium (δD) will be measured concurrently and relationship to climate and environment will be explored based on results obtained. Nine samples so far were analysed and results show that 'vital effects' exist between the two analysed taxa. A close relationship between δ18O of lake water and the HCs was found, however more data are required for further conclusions to be made. There is a possible global correlation between $\delta^{18}O$ of lake water and chironomid head capsules from Australia to Europe and northern North America.

To cross-check the validity of the $\delta^{18}O$ relationship, we will use the transfer function for the chironomids from these lakes. Transfer functions are standard techniques in palaeolimnology and they are generally regarded as accurate and the results from the transfer function can cross-validate the stable isotope results and highlight where, if anywhere, the stable isotope signal is dominated by non-temperature fractionation effects. Once verified, the stable isotope technique will be applied to chironomid head capsules extracted from lake sediment deposits in southern Australia.

6. Conclusion and future work

Limnology and chironomids surveys were conducted through 45 lakes from east and southeast Australia. The findings on land-use, climate change, and limnology nexus and the chironomid-inferred temperature transfer function were published in Chang et al (In Press) and Chang et al (under review) respectively. Future chironomid based lake trophic status transfer function will be developed, validated and applied. Chironomid stable isotope will provide a new proxy for palaeoenvironmental studies in Australia, and if a global correlation exists, this approach should be applicable in chironomids extracted from wetlands elsewhere including the lakes from China.

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Robust recycling of water: The water plant of the future

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Abstract

Our communities, both large and small, have traditionally disposed of waste water to our rivers, lakes and oceans using treatment practices that ensure that particulates, pathogens, chemicals of concern and nutrients are at such a level as to ensure that the receiving water source is, both are not nitrified or immediately hazardous to both environmental and human health. The assumption herein is that dilution to the receiving waters is significant and that the assimilation time is sufficient for natural degradation processes to dominate. Whilst these practices have served us well, population growth and inadequate catchment protection has placed great pressures on our rivers and lakes and many of these water resources are no longer recognisable as a protected water supply for potable input to our communities using conventional water treatment nor as an environmental domain that can sustain biodiversity. In short, the inputs into many of our rivers and lakes are beyond the point where natural ecosystem processes can purify the water and population pressure is such that conventional processing will never recover the situation.

Traditional water treatment systems for potable supply and waste have conventionally been separate processes but in a highly populous domain, where the source and receiving waters are beyond the tipping point, it is appropriate to revisit the concept. In this scenario, water sources are deemed 'unprotected' and need non-conventional treatment and waste water treatment needs to go beyond secondary processing since assimilation rates in receiving waters are too slow. It is of note that the treatment processes required to achieve both goals merge whereby tertiary treatment of waste water to achieve an output with limited or no environmental impact and treatment of waters with pathogens and contaminants beyond those that can be dealt by conventional coagulation and coarse filtration practices, are very similar. Thus, production of waste water for environmental discharge from secondary treated waste water and production of potable water become one and the same.

The water plant of the future needs to deal with the many types of water that our community needs. This includes the needs of recreation, households, industry and the environment. In a highly protected catchment, conventional practices appear appropriate but this is becoming less common with time. Work in our group, in collaboration with Victoria University and a number of water industry service providers, has looked to develop a robust water recycle plant. The plant is designed to not only be low on maintenance, energy (relative to desalination of seawater) and chemical use but be able to deal with a wide variety of source contaminants, to not only produce a potable quality product but a discharge (waste) that is also of very high quality in terms of maintaining environmental values.

Keywords

Water recycling, environmental hazards, human health, water treatment plants, desalination, river catchment

Collaborating organisations

Federation University Australia

Nanjing Institute of Geography and Limnology Chinese Academy of Sciences (NIGLAS)

University of Melbourne

University of Canberra

Charles Sturt University

Griffith University

University of New England

RMIT

Tsinghua University

Northeast Institute of Geography and Agroecology

Beijing Forestry University

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