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Management Strategies for Large River Floodplain Lakes Undergoing Rapid Environmental Changes

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1. Introduction

Large river basins are the origin of ancient civilization (Barbier & Thompson, 1998; Sadoff & Grey, 2002). Floodplain lakes, situated adjacent to large river systems, are connected with river channel networks. The connectivity between river channels and wetlands makes the “boom” and “bust” ecology following the drought and flood events that continues to support diverse floral and faunal communities in the floodplains lake systems (Jenkins & Boulton, 2003). Rich biodiversity and occurrence of macro-invertebrate drifts in the Upper Paraguay River-Floodplain-System, parts of the Pantanal (Brazil) Wetland System, and dense microphyte community with regularly supplied allochthonous nutrient inputs and moderation of physical extremes in the billabongs of the Murray-Darling River Floodplain-System Australia are some examples of highly productive floodplains lake ecosystems in the world (Shiel, 1976; Wantzen et al., 2005). Being a productive ecosystem, people living across the large river basins have been greatly benefited from the resources generated by these wetlands for generations (Bright et al., 2010). For example, the indigenous people of the Orinico River Basin, South America, and Murray Darling Basin, Australia have been harvesting the specialised fish community that are adapted to the floodplains wetland systems over several centuries in the past (e.g., Lundberg et al., 1987; Humphries, 2007).

Since the productivity of the large river floodplains lake ecosystems is dependent on naturally occurring riverine flood events, any alternation of the hydrological patterns of rivers can have strong impacts on nutrient dynamics, biological diversity and assemblages of these lakes (Fisher et al., 2000). Over the past few decades the large river systems and its adjacent wetland habitats have undergone rapid environmental changes. Anthropogenic activity across the river basin has increased substantially. River regulations such as construction of dams, irrigation channels, dykes and weirs, and catchment land use activities such as deforestation, agriculture and cattle ranching and introduction of exotic flora and fauna are increased (Power et al., 1996; Kingsford, 2000, Bunn & Arthington, 2002). Rapid climate warming is further intensifying the conditions of ecosystems including the changes in hydrology and water quality of rivers and lakes (Carpenter et al., 1992; Lewis et al., 2000; Palmer et al.; 2008). The coupled human-climate disturbances have led to an increased habitat heterogeneity and complexity of ecosystem processes of majority of floodplains lake systems worldwide (Tockner et al., 2000). Consequently, the people who

have been directly associated with these large river systems for a range of services over generations are influenced by these changes for sustainable living.

One of the critical issues today for majority of river scientists is therefore to understand the large river floodplains lake ecosystems processes that are exposed to a range of coupled human-climate disturbances. Understanding the ecosystem processes and identifying the disturbances altering ecosystem processes can help resource managers to tackle challenges of floodplains lake management and promote healthy and productive ecosystems across the large river basins worldwide. The large river floodplains lake ecosystems are longitudinally modulated by upstream processes, where the main source of organic carbon such as fine particulate matter is transported to downstream environments (Vannote et al., 1980). The use of particulate organic matter is maximized by benthic heterotrophs and microcrustaceans because depositional structures are limited to backwater and nearshore areas (Naiman et al., 1987). However, the role of locally derived ecological processes is unknown in the longitudinal river continuum (Statzner & Higler, 1985) since locally metabolised carbon and the bottom-up control of algal communities are also important for ecosystem processes (Wehr & Descy, 1998). Metabolism and turnover rates of organic carbon in floodplain lakes can vary with the type and nature of the river system from which they are derived. Floodplain lakes associated with blackwater rivers for example have low content of suspended sediments but a high concentration of dissolved organic matter (Meyer, 1990). Metabolism of these floodplain lakes is dependent on allochthonous organic carbon with increased river size despite increases of downstream gross primary production, where riparian swamps are the source of organic carbon (Meyer & Edwards, 1990). Flood pulses in particular are the significant source of carbon for ecosystem structure and functions in large river floodplain lakes (Junk et al., 1989). Accessibility and retention of organic matter are functions of the frequency and duration of flood pulse and extent of inundations (Humphries et al., 1999; King et al., 2003). Apart from the organic matter derived from flood pulses, integration of locally derived autochthonous matter such as phytoplankton, benthic algae and aquatic vascular plants and direct inputs from the riparian zones such as abscised leaves, particulate organic matter, and dissolved organic carbon during flood pulses are also significant source for floodplains lake ecosystem structure and function (Thorpe & Delong, 1994). Autochthonous carbon and direct allochthonous inputs from the riparian zone are labile and maximally utilized by heterotrophs dominant in near-shore leaf-litters and littoral habitats (Thorpe & Delong 1994). As the large river floodplains lakes ecosystems are exposed to coupled human-climate disturbances, scientists have been facing increasingly difficulties to understand the complex ecosystem processes worldwide.

Changes in species richness, diversity and assemblages of biota such as fish, diatoms, macro-and micro-invertebrates across temporal and spatial scales have become useful for understanding rapid environmental changes of large river floodplains lake ecosystems (Ward et al., 1999). Evaluation of the large scale changes in ecosystems as a result of micro-scale changes in environments such as a small rise in surface water temperature or additional inputs of phosphorous concentrations in floodplain lakes over a temporal scale is crucial in relation to changes in biotic assemblages. Interaction between the channel and floodplain systems and between the channel and groundwater aquifers plays a significant role for rapid ecosystem changes in floodplain lakes ecosystems, since the temporal dimensions of these ecosystems are largely integrated and dynamic (Fig. 1, Ward, 1989; Lewis et al., 2000).

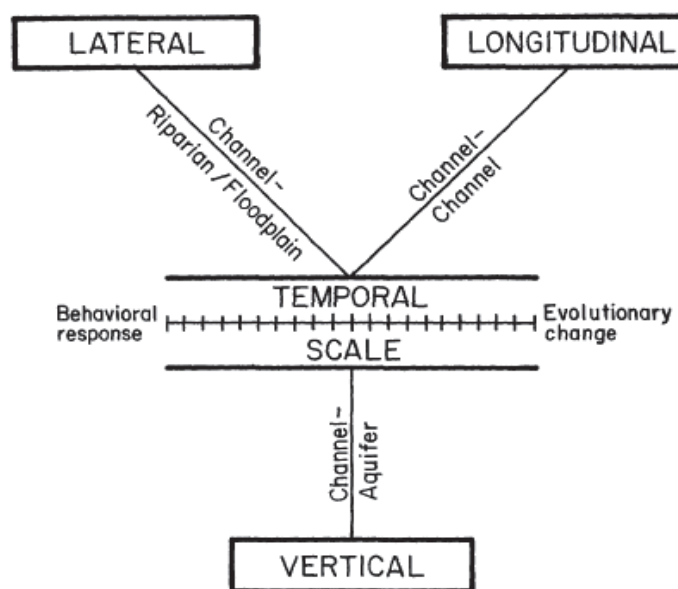


Fig. 1. Four dimensional structures of floodplain lake ecosystem (adapted after Ward, 1989). Triangular interactions (lateral-longitudinal-vertical) determine the spatial and temporal changes of floodplain lakes biota. Lateral influence occurs at riparian zone while longitudinal influence is restricted in river channels. Vertical influence occurs at ground-water aquifer.

Micro-crustaceans are one of significant indicators of rapid environmental changes of large river floodplains lakes over a range of temporal and spatial scales. Microcrustaceans prefer littoral benthic and pelagic habitats and they have wide optima and tolerance to a range of environmental variables. Micro-crustaceans such as cladocerans are one of significant components of the large river floodplains lake ecosystems. Cladocerans emerge rapidly following the inundation, feed principally on phytoplankton, bacteria and detritus and actively transfer energy across the food webs (Reid & Brooks, 2000; Jenkins & Boulton, 2003). Cladoceran exoskeletons and their ephippia are archived in floodplain lake sediment being useful indicator for a long term environmental changes (Kattel, 2011). The use of modern and sub-fossil assemblages of micro-crustaceans such as cladocerans can help floodplains lake ecologists and river scientists to understand complex ecosystem processes and develop effective management strategies for these ecosystems worldwide. In this chapter, we have identified a range of issues of rapid environmental changes of large river floodplains lake ecosystems worldwide. We have then highlighted the use of the micro-crustaceans, such as cladoceran zooplankton to improve management practices of the vulnerable ecosystems of floodplains lakes in the large river basins.

2. Materials and methods

This chapter is based on a range of case studies on large river floodplains lake ecosystems worldwide. The case studies were varying in nature either focusing on theoretical models being developed over the past decades on large river floodplains lakes ecosystem processes, or highlighting the impacts of global environmental changes on these floodplains lake ecosystems. The theoretical models were reviewed mainly on river continuum concept

(RCC), flood pulse and riverine productivity (RPM) models, where most of these models were tested in North America, Europe and Australia for understanding ecosystem processes of the large river systems (Vannote et al., 1980; Naiman et al., 1987; Junk et al., 1989; Thorpe & Delong, 1994). The case studies on critical management issues of rapid environmental changes of the large river basins were collated from various continents including the Yangtze River System, Asia (Yang et al., 2007; Chen et al., 2011), the Mississippi River System, North America (Wren et al., 2008), the Orinoco, Salado and the Paraguay River Systems, South America (Lundberg et al., 1987; Claps et al., 2009), Orange-Vaal River System, South Africa (Ashton et al., 1986), Erbo River System, Europe (Gallardo et al., 2007) and the Murray Darling River System, Australia (Humphries et al., 1999; King et al., 2003). Following the identification of critical management issues of the large river systems, prevailing conditions of biotic assemblages in changes in large river floodplain lakes were reviewed from the case studies of Europe, North America and Australia (Fisher et al., 2000; Lewis et al., 2000). Then a comprehensive review was undertaken on the use of microcrustaceans to understand the complex ecosystem processes and configure effective management strategies when they are exposed to a range of external disturbances including climate change over temporal and spatial scales.

3. Management issues for large river floodplains lake ecosystems

At least four major disturbing factors have been identified for alteration of ecosystem processes of the large river floodplains lakes worldwide. Resource managers are increasingly concerned about management issues of river regulation, land use activity, introduction of exotic species and rapid climate warming which have been considered to make significant impacts on large river floodplains lake ecosystem processes worldwide in recent decades.

3.1 River regulation

Water abstraction, diversion and regulations are one of critical management issues of floodplains lake ecosystems of river basins. Naturally occurring physical structures strongly support biodiversity of lakes. Natural flows are important for succession of food web structure and dynamics (Fig. 2). Physical structures enhance water quality, energy budget and flow regime of rivers. Improved water quality maintains the health of floodplain lakes. Interaction between channel morphology and river discharge in up-streams is important for structure and function of the downstream river ecosystems (Bunn & Arthington, 2002). For example, braided river channels in arid and semi-arid climates are characterised by a network of constantly shifting low sinuosity water courses while meandering river channels are influenced by channel width and depth with steady discharge (Johnston et al., 1997). The alteration of these river channels influences morphology consequently the habitat of a wide range of biota in the reaches which are significant source of energy transfer across the food web. For example, fish of the Orinoco River use the main channel primarily for migration and dispersal depends largely on floodplains for growth and subsistence (Lewis et al., 2000). Diversity and length of the food chains increase in natural flow regimes of many large river systems (Maddock et al., 2004). However, construction of dams, weirs and irrigation channels can have substantial implications for physical structures and riparian ecosystems (Walker, 1985). Such consequences can be immediate and obvious or gradual and subtle depending on the nature of regulation and the occurrence, diversity and composition of biota (Power et al., 1996).

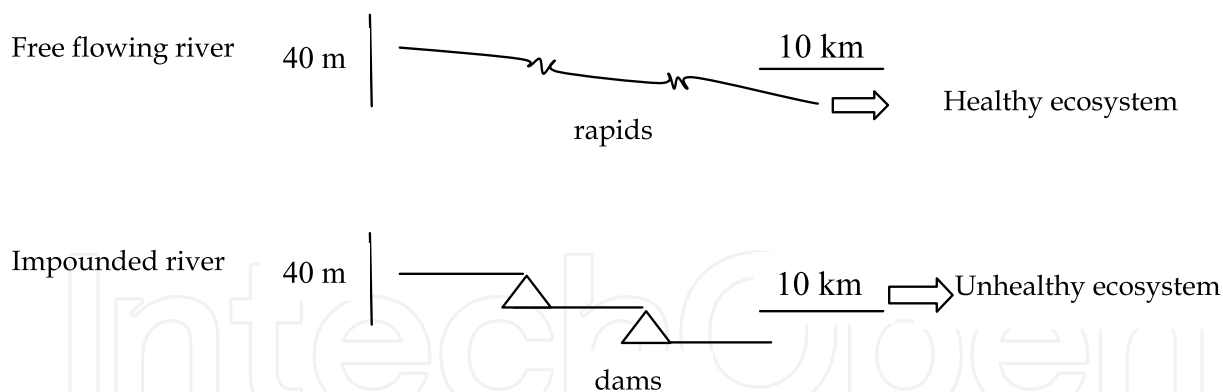


Fig. 2. Free flowing rivers consist of series of rapids and slow flowing stretches. Rapids are important for succession of food web structure and dynamics. Naturally occurring physical structures enhance water quality, energy budget and flow regime of the river. Improved water quality maintains the health of riverine floodplain lake ecosystems. In impounded rivers, rapids have been removed by erected dams. Dams can have direct implications on hydrology reducing the downstream flow variation. Dams hinder the upstream migration of biota, alter thermal environment, nutrient movement and sediment loading and predator-prey interaction in downstream food webs (adapted after Nilsson & Berggren, 2000).

Regulation of the Murray River, Australia over the past 50 years has resulted in considerable implications for ecosystem structure and functions. For example, construction of dams in the Murray River has reduced downstream flows as well as obstructed upstream migration of biota including thermal environment, nutrient movement and sediment loading and predator-prey interactions (Gehrke & Harris, 2000). Since flooding generates biogeochemical processes, the major impact of dams is the interruption of the exchange of energy between river and riparian zone during flood events (Sam et al., 2000). Low flows events are critical for lowland fish assemblages and plant community structure (Capon, 2003). Increased water residence time increases crustacean biomass (Humphries et al., 1999). However, alternation of natural low flow patterns can influence diadromous fish populations which utilize crustaceans as their major diet. Fish species such as Murray cod (*Maccullochella peelii peelii*) and silver perch (*Bidyanus bidyanus*) which do not require special flood events in the Murray River Australia is able to utilize low-flows events for spawning (King et al., 2003). However, the growth of larvae of Murray cod (*M. peelii peelii*) is significantly influenced by construction of dams and irrigation channels across the MDB. Larvae are consistently stranding in the dam when drifting (Koehn & Harrington, 2005). In contrast, recruitment of other fish species requires floodplain inundation and increased water volume (King et al., 2003).

Alteration of riparian vegetation can influence nutrient sources of wetland biota. Composition and diversity of naturally occurring riparian forests such as river red gum trees (*Eucalyptus camaldulensis*) in MDB have declined as a result of river regulation (Robertson et al., 2001). For example, stable isotope ratios of oxygen reveal that river red gum (*E. camaldulensis*) forests are efficient for utilizing water at varying salinity gradients (Mensforth et al., 1994) through reduced transpiration rates (Costelloe et al., 2008). However, continued low flows occur as a result of a rise in ground water salinity. Absence of natural floods influences recharge of naturally occurring groundwater salinity levels and will have detrimental effects on floodplains riparian biota (Jolly et al., 2001). Die back in

river red gum (*E. camaldulensis*) communities can occur at electrical conductivity as high as 40dSm^{-1} (Mensforth et al., 1994). Substantial buffering of catchment soils as a result of river regulation and subsequent release of sulphur in floodplains following the European settlements has influenced on diatom communities in the Lower Murray River (Gell et al., 2007). Irrigation of soils with low permeability is also causing saline groundwater to rise. Salt accumulates in the top soil as water continues to evaporate. Partial drying of previously inundated floodplains reduce nutrient availability such as total nitrogen (TN) and total phosphorous (TP) in the system causing negative effects on ecosystem functioning (Baldwin & Mitchell, 2000). Irrigation dams in the previously fertile Indus River floodplain (Pakistan) are also reported to have caused a massive salinity problem. Extensive abstraction of water from the Amu Dar'ya and Syr Dar'ya, the two largest tributaries of the Aral Sea has caused 80% reduction of the water volume in the Aral Sea within the last four decades resulting in a four-fold increase in salinity concentrations of the floodplain lake consequently limiting ecosystem structure and functions (Aladin & Plotnikov, 1993).

3.2 Land use

Increased land use activity across the catchment of the large river system is other significant management issue. Ecological attributes of large river floodplain lakes have been constantly modified by industrial and cultural developments. Modern farming practices have made implications for physical and hydrological features of floodplain wetlands including the changes in water quality and sediment processes. Wren et al. (2008) reported that the sediment accumulation rates of the Sky Lake in the Mississippi River system, USA has increased to 50-folds following the clearing of forests began by humans in 300 years ago. In natural flood pulse concept, river floodplains are regularly flooded and dried (Bayley, 1995). Catchment organic matter generated across spatial and temporal scales is transported to river floodplains. A high turnover rate of organic matter and nutrients are predicted to occur as a result of natural flood events. During flood events, nutrients dissolve with flood waters consequently accelerating primary production. However, under dry conditions, decomposition processes of floodplain lakes would increase relative to production. Intensification of land use including waste disposal, agriculture, grazing and forest clearance in catchments all have considerable implications for natural flood pulse events (Jansen & Robertson, 2001).

Large river floodplain wetlands are species rich habitats which connect distant ecosystem not only through the migration of river biota but also from the transport of water, sediments, nutrients and contaminants (Sparks, 1995; Fisher et al., 2000, Chen et al., 2011). The integrity of floodplain lakes, which is maintained by hydrological dynamics, biological productivity and river connectivity are significantly impeded by land use activity. Alteration in riparian vegetation in particular is detrimental for changes in species diversity and ecosystem functioning of floodplain lakes. For example, alteration of the natural riparian vegetation by humans has modified the ecosystem processes of the wetlands and its catchments across the Sacramento, USA. Modification of wetland landscape has already been noticed as a result of cultivation, soil erosion and sedimentation to down-streams and in many cases loss of productivity has also occurred (Alpert et al., 1999).

Application of nitrogen and phosphorous has increased for agriculture across the large river basins worldwide. An alteration in global nitrogen cycle has occurred in recent years by widespread use of N-fixing crops, fertilizers, habitat change and burning of fossil fuels.

Continuous use of nitrogen and phosphorous as fertilizers in agriculture and urban landscapes lead to leaking of mobile inorganic nitrate ions in the system (Turner et al. 2003). As a result of algal blooms and low dissolved oxygen at nutrient rich environment, wetland ecosystem health has reduced substantially. Widespread release of phosphorous into the Yangtze River floodplain lakes over the past decades, for example, has caused a regime shift, where a transformation has occurred in large number of lakes with macrophyte dominated states to algal dominated states (Yang et al., 2007). Although some disturbances are beneficial to habitat heterogeneity and species, the lack of disturbance events have negative impacts on these lakes. For example, the Oxbow Lake of the Middle Erbo River (Spain) and Bottle Bend Lagoon of the Murray River (Australia) are reported to have undergone increased salinisation and eutrophication followed by a loss of biodiversity as a result of land use change (Lamontagne et al. 2006; Gallardo et al., 2007).

Land use activity has also exacerbated the release of a range of toxic substances in large river floodplain lakes. Trace metals (e.g., Hg, Pb, Zn), persistent organic pollutants (POPs), and organometallic compounds are detrimental for ecosystem health. Polycyclic aromatic hydrocarbons (PAHs), polychlorinated biphenyls (PCBs), polychlorinated dibenzo-p-dioxins (PCDDs) and polychlorinated dibenzofurans (PCDFs) as well as many other organochlorine pesticides (e.g. DDTs; taxophene) and brominated flame retardants (BFRs; including polybrominated diphenyl ethers (PBDEs) can be lethal for wetland biota if their concentration is high in the system. The organometallic compound, such as methylmercury (MeHg) is the most toxic compound (Leung et al., 2007). Following the industrial revolution in Europe (1800 AD), wetland contamination by organochlorine compounds increased substantially. The DDT concentrations for example in lake sediments were the highest during the 1950s. Boating activity in floodplain lakes has influenced substantially for estuarine biota in the large river mouths as well as mollusc communities in freshwater environments due to increased organometallic toxicity (US-EPA, 2003). Methylmercury, for example although present in a small concentration (0.1-5.0 percent) of total mercury, it represents 90-100% in invertebrates and fish. Increased POPs, PCBs and PAHs toxicity can cause endocrine disruption in fish and crustaceans (Matthiessen & Johnson, 2007). The tributyl tin (TBT) can cause reproductive failure in *Daphnia* at 400 ng L⁻¹ and 380 ng L⁻¹ TBT levels (Brooke et al., 1986). Macrophyte density in lowland shallow river floodplain wetlands in the UK substantially reduced in the sixties as a result of recreational boating followed by TBT pollution (Sayer et al., 2006).

3.3 Introduction of exotic species

Introduction of exotic flora and fauna is another significant management issue of the large river floodplain lake ecosystems worldwide. Displacement of habitats and subsequent extinction of native populations are reported as some of the foremost impacts of introduced species in many large river floodplain lakes. River basins of the Northern Hemisphere inhabit the highest number of non-native fish species (Leprieur et al., 2008). More than 50% of the biota of the Hudson River in the USA comprises introduced species mostly from Europe where 10% of those populations have significant ecological impacts on native populations (Nilsson & Berggren, 2000). Human activities are blamed to facilitate the establishment of non-native species by disturbing natural landscapes and by increasing propagule pressures on native populations (Leprieur et al., 2008; Simões et al., 2009).

Large river floodplain lakes are increasingly sensitive to biological invasion. The extended river networks often have recurrent disturbances and enhanced invasion (Elvira, 1995; Mills et al., 1996). Dispersal of seeds and eggs are rapid at landscape level through river channel networks. Disturbance regime and floodplain productivity also enhance invasion (Chapin III et al., 2000). Non-native species once introduced in large river systems can spread rapidly (Koehn, 2004). Favouring wide ranging climates, flexible in habitat selection and increased physiological adaptation are characteristic features of non-native species for a successful colonisation in a new environment (Mooney & Cleland, 2001). Whilst the impacts of invasion on native populations has been increasing, what condition is necessary for invasion, the way the invasion progresses through space and time and the properties of invasive biota is yet to be understood fully. Under regulated environments these patterns have become pronounced, and the nature of the invading species in susceptible habitats is also becoming unpredictable (Bunn & Arthington, 2002). For example, water regulation in one of the South African large river systems (Orange-Vaal River System) has stabilized the natural flow regimes favouring the alien aquatic vegetation (e.g. *Myriophyllum* sp, *Azolla* sp) consequently reducing the water movement, light penetration and oxygenation followed by displacing the native vegetation bed (Ashton et al., 1986). Introduced fish species such as European perch (*Perca fluviatilis*) and common carp (*Cyprinus carpio*) in the Murray River Australia have successfully established populations following the European arrival causing retarded growth and development of native fish populations (Koehn, 2004). Some endemic species including Macquarie perch (*Macquaria australasica*) Murray hardyhead (*Craterocephalus fluviatilis*) and Murray cod (*M. peelii peelii*) have become critically endangered or vulnerable in recent decades (Hutchison & Armstrong, 1993).

3.4 Climate change

Rapid rate of climate warming in recent decades has become one of important management issues of large river floodplain lake ecosystems. Climate change can cause floodplain lakes ecosystems through a variety of ways such as alteration of flood events, channel morphology, nutrient dynamics and growth and reproduction of wetland and riparian biota.

Floods are essential for nutrient dynamics, primary and secondary production and growth and development of native plant and animals (Harris & Gehrke, 1993). Regular inundation provides water for riparian vegetation and continuation of ecosystem processes (Nilsson & Berggren, 2000). Runoff with organic rich nutrients create potential for the establishment of a new community. Recovery of riparian catchments after flood or drought is rapid and the diversity and abundance of flora and fauna increase substantially within a short period (Jenkins & Boulton, 2003). However, climate warming reduces annual inflows and runoff volume of the large river systems. Climate change also alters river channels, erosion, nutrient and sediment transports influencing terrestrial vegetation, soil moisture and evapotranspiration processes in large river floodplains lakes (Palmer et al., 2008). Holocene records of floodplains in the USA show that magnitude of floods is intense in arid regions resulting in channel widening which often have sparse riparian vegetation (Carpenter et al., 1992). In Murray River, a rise of 1° C in recent decades is predicted to have caused approximately 15% reduction in the annual flows (Cai & Cowan, 2008). Since 1950s, the MDB has experienced warming of around 0.8° C with declining rainfall as low as 10 mm per decade resulting in degraded water quality across the region. Important flood-cued native

fish populations such as golden perch (*Macquaria ambigua ambigua*) are significantly altered as a result of climate-induced low flow events in the MDB (Humphries et al., 1999).

Climate change influences nutrient concentrations in floodplain lakes (Spink et al., 1998). Elementary nitrogen level and biogeochemical cycles in sediments can vary with climate warming (Catalan et al., 2002). In drought phase, sulphur stored in the upper areas of the littoral zone can re-oxidise causing lakes and river floodplains in down-streams to re-acidify (e.g., Yan et al., 1996; Dillion & Lazerte, 1992). Rising temperature, longer dry spells and runoff distribution in the MDB for example have intensified vegetation patterning and concentration of dryland salinity in recent decade (Hughes, 2003). When soil with rich sulphides (or 'black ooze') characteristic of dark and soft are disturbed and oxygenated, they react rapidly resulting in environmental hazards floodplains systems (Lamontagne et al., 2003).

Climate warming can influence growth and reproduction as well as phenology of wetlands biota directly (Hughes, 2003). Increased concentration of atmospheric CO₂ intensifies photosynthetic processes of riparian trees. The leaf stomatal conductance of these trees decreases and the plant-water use efficiency increases in elevated CO₂. However, productivity of plant biomass at high level of CO₂ is short-lived resulting in changes in energy balance in the system (Dunlop & Brown, 2008).

Whilst we have identified some key management issues of the large river floodplains lake ecosystems, the next step is to find appropriate solutions for these problems. There are a range of management options available, our aim is however, how we can understand and best interpret the ecosystem processes of the floodplain lakes that are exposed to anthropogenic and climatic variability and can guide resource managers using the best management strategy. Understanding of changes in assemblages and diversity of wetlands biota, particularly micro-crustaceans along temporal and spatial scales is one of potential tools that provides prevailing conditions of changing large river floodplains lake ecosystems.

4. Prevailing conditions of biotic assemblages in changing large river floodplain lakes

The structure of micro- and macro- invertebrate communities is dependent on factors such as water quality (e.g. nutrients salinity, pH), food resources and habitat availability. Oligotrophic (low nutrient) conditions may limit primary production, thereby limiting a key food resource (i.e. phytoplankton) for some functional groups of invertebrates (Jeppesen et al., 2000). Eutrophic (high nutrient) conditions, high temperature and stable (e.g. stratified and poorly mixed) water bodies may favour key phytoplankton groups (i.e. cyanobacteria) that are a poor quality food resources for micro-invertebrates (Jeppesen et al., 2000). Furthermore, some functional groups of invertebrates feed exclusively on either phytoplankton or macrophytes. Consequently, shifts between macrophyte and phytoplankton dominated states will lead to shifts in the composition of micro- and macro-invertebrate communities (Jeppesen et al., 2002). Changes in water column salinity may also substantially influence the composition of the micro-and macro-invertebrate community due to differences in species-specific salinity tolerances. Furthermore, it is considered highly likely that changes in soil salinity (Brock et al., 2005) or soil pH (Hall & Baldwin, 2006) may severely impact the viability of invertebrate seed banks within wetland/floodplain soils. It is also recognised that the viability of invertebrate seed banks decreases with time (Nielsen

et al., 2007). Long (i.e. >10 years) dry periods may exceed the viability period, severely compromising the invertebrate community that will hatch from soil seed banks during subsequent floods (Williams, 1985).

However, the variety of conditions among floodplains lake biota at any given time is largely dependent on system processes of the river basin. Abundance and diversity of microcrustaceans can change with the initial condition of the basin morphometry, where setpoints are determined by flood inundation (Fig. 3). The ecosystem of large river floodplain lakes adjacent to the large river is primarily influenced by its position, how far the lake is situated from the river, and how long the inundation is lasted for (Lewis et al., 2000).

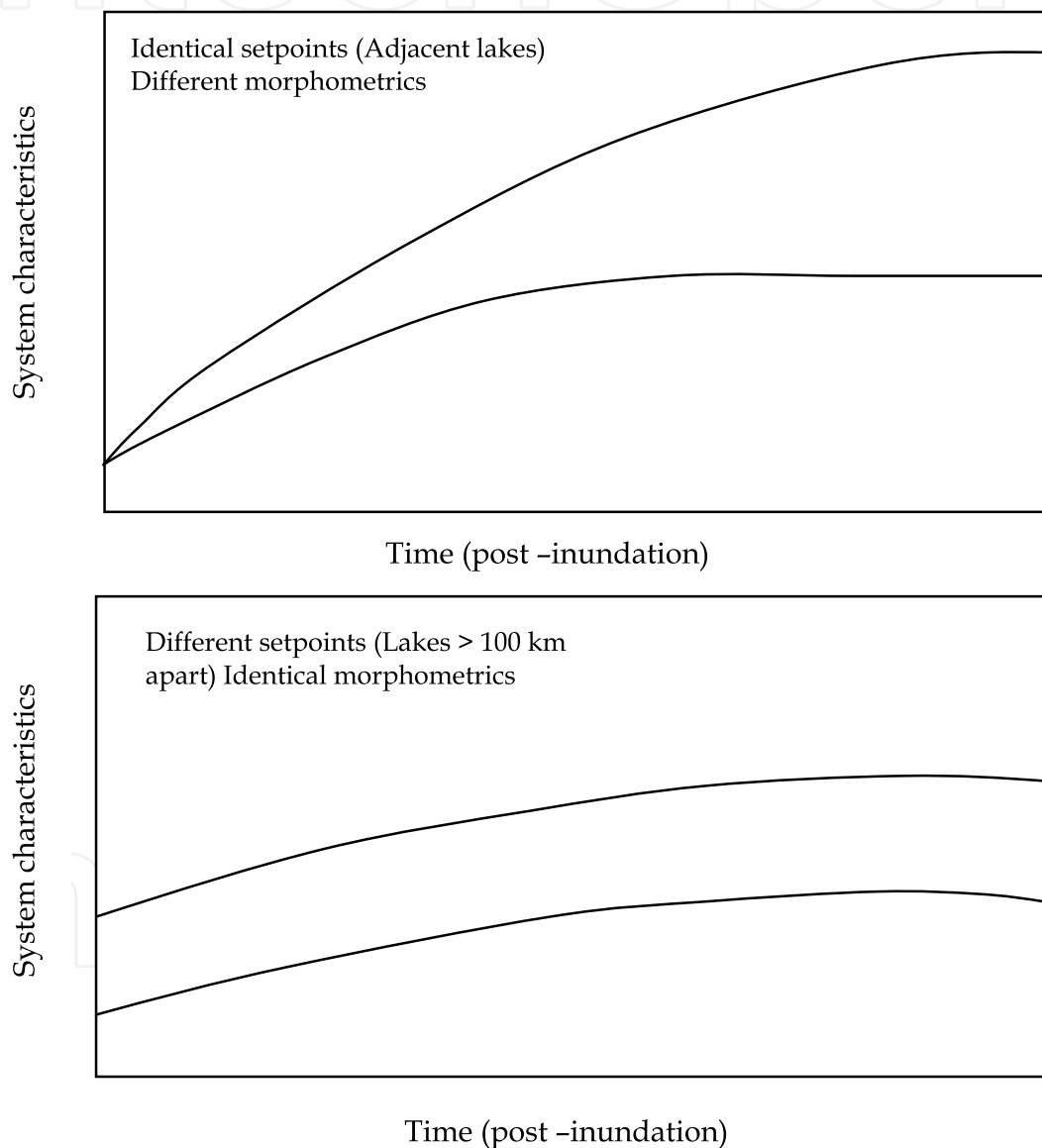


Fig. 3. Causes of the variation in the assemblages of wetland biota including microcrustaceans, and abiotic composition amongst large river floodplain lakes. Setpoint at the time of inundation (upper diagram) is determined by position of adjacent lakes distributed within the c. 600 km of the floodplain; setpoint following the inundation (lower diagram) is determined by basin morphology (adapted after Lewis et al., 2000).

As a result, water quality of the adjacent wetlands will change following the flood inundation, consequently diversifying the biotic and abiotic assemblages across the wetland (Lewis et al., 2000). For example, in Missouri River floodplain wetlands, alteration of river corridor is reported to have reduced flood pulses significantly. As a result of the absence of flood pulses, micro-crustaceans such as copepod and *Bosmina* showed a strong sensitivity to basic habitat characteristics during and after the flood events within the naturally functioning section of the river (Fisher et al., 2000).

A comprehensive understanding of the large river floodplains lake ecosystems can only help configure effective management strategies. Information regarding diversity and assemblages of micro-crustaceans across temporal and spatial scales is useful for understanding degraded floodplains lake ecosystems and water quality. Changes in assemblages of micro-crustaceans at particular time can provide disturbances caused by external forces such as climate change, invasive species and anthropogenic release of nutrients into the systems and help resource managers to mitigate these problems.

5. Configuring management strategies of large river floodplain lake ecosystems: Role of micro-crustaceans

Assemblage structure of micro-crustaceans such as cladoceran zooplankton across temporal and spatial scales of large river floodplains lakes can help resource managers for understanding the drivers of ecosystem changes and configuring a range of management strategies. Below how the information obtained from micro-crustaceans are useful to manage floodplains lake ecosystem is comprehensively discussed.

5.1 Management of food web

Understanding of temporal and spatial changes in diversity, composition and abundances of micro-crustacean assemblages are useful for sustainable ecosystem management in large river floodplain lakes. Seasonal production of autochthonous carbon (algae, macrophytes) and inputs derived from the riparian catchments help functioning of floodplains lake ecosystems (Thorpe & Delong, 1994; Lewis et al., 2000). The carbon derived from the riparian system is assimilated by micro-invertebrates supporting the higher trophic levels in food web. However, the degree of energy assimilation by micro-crustaceans in lacustrine food web is less understood. The physical transport of materials to biological transformation to carbon in floodplains lakes varies substantially due to alteration of river flows (Walker et al., 1995). Micro-crustaceans serve as an important role during energy transfer across the trophic levels. For example, in an arid river, Rio Grande (New Mexico, USA), recruitment of some fish occurred during high flows (spring), whereas other fish recruited during low-flows (late summer). Micro-habitats with low current velocity and high temperature were vital nursery grounds for the Rio Grande fishes. Stable isotope analyses of carbon revealed that the Rio Grande fish larvae would obtain carbon predominately from algal production in early summer, but would use organic carbon derived from emergent macrophytes when river discharge would decrease in mid-summer. The shift in carbon assimilation was facilitated by micro-invertebrates that reduced edible algae switching to macrophytes in mid-summer (Pease et al., 2006).

Some species of cladocerans have responded to flood events in the Orinoco River floodplain lakes in Venezuela by showing a varying birth, death and population rates (Twombly & Lewis, 1989). In these floodplain lakes, birth rates increase at a time of flood inundation

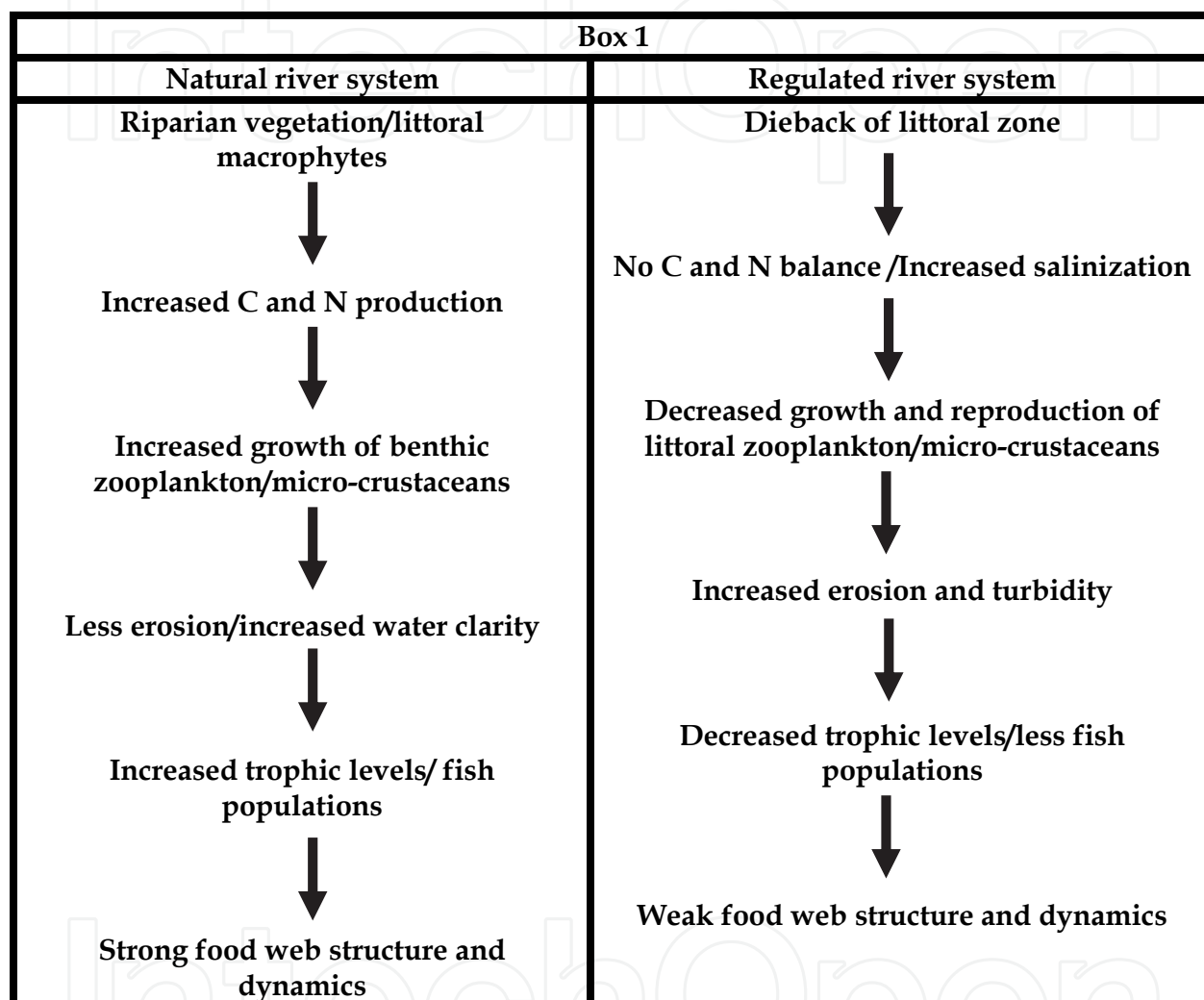
while mortality increases when fish and invertebrate predations are high (Twombly & Lewis, 1989). In a lowland river system, fish can have size selective predation leading to small sized zooplankton dominating the system, consequently the preservation of the small-sized zooplankton such as *Bosmina* in the system. Mean size of cladoceran mandibles, remains of *Daphnia*:*Bosmina* ratios and the length of the carapaces and mucros of *Bosmina* can infer past fish assemblages in floodplain lakes and help understanding any changes in food web over time (Kattel, 2011). The cladocerans display morphological variability (cyclomorphosis) in food web. Vertebrate predation pressure on *Bosmina*, for example can result in variation in size of the mucro (Hann et al., 1994). In temperate arid Australia, *Daphnia carinata* show a cyclomorphic behaviour with seasonal changes in body size. Increase in *D. carinata* size indicates a low seasonal water temperature and can help infer the condition of the micro-habitat climates for growth and reproduction of these animals (Mitchell, 1978).

Prolonged drought can lead to cessation of crustacean populations and functioning of floodplain ecosystems. Intensity of floodplain drying also increases changes in algal composition and diversity intensifying the top-down predation and competition (Schneider & Frost, 1996). However, the dry-wet cycles in floodplains recharge the system contributing to the emergence of endangered species of micro-crustaceans through regeneration of egg banks (Boulton & Loyid, 1992). The ephippia of cladocerans in floodplain lakes are viable for several decades. Hatching of resting eggs through genetically advanced technology can help restoring endemic populations (Jeppesen et al., 2000). For example, *Daphnia* ephippia as old as 40 years derived from a lake in South Australia is reported to have been able to hatch in the laboratory environment (Barry et al., 2005). Recently Jeppesen et al. (2002) have successfully reconstructed the catch per unit effort (CPUE) of the planktivorous fish inferred by *Daphnia* ephippia size in a European lake. In the Murray Darling River, the patterns of micro-crustacean distribution and ecosystem processes have been altered by alteration of littoral vegetation and zooplankton egg banks (Jenkins & Boulton, 2007). Unlike in natural condition, where riparian vegetation and littoral macrophyte communities are intact, C and N production is cyclical in nature promoting the growth of littoral macrophytes and micro-crustaceans stabilizing food web structure and dynamics through improvement of the water quality (Box 1), the dieback of littoral vegetation in impounded rivers can alter entire ecological processes including the C and N balances followed by increased salinisation in the region. Zooplankton to phytoplankton ratios serves as a good indicator for grazing intensity of fish and provides the insight for food web structure and dynamics of floodplain lakes (Jeppesen et al., 2001).

5.2 Management of water quality

Micro-crustaceans are used for assessing water quality of the large river floodplain lakes extensively (Gannon & Stemberger, 1978). These organisms are classified according to their preferences for nutrient enrichments (e.g. eutrophic, mesotrophic or oligotrophic), chemistry (e.g. alkaline, acidic or saline) in water. Phosphorous (P) and nitrogen (N) are two key nutrients significant for wetland ecosystems. Phosphorous is commonly the growth limiting nutrient in freshwaters exerting a strong control on species composition and primary productivity. Nitrogen can also be a limiting or colimiting nutrient with phosphorous. Anthropogenic influences especially from sewage effluences and agricultural fertilizers can enrich P and N concentrations substantially reducing the water quality (Boucherle & Züllig, 1983). For example, dramatic rise of nitrate concentrations in the Michigan River wetland

system was a result of land use intensification following the European arrival. An increased grazing pressure by zooplankton on large algae resulted in increased smaller phytoplankton populations in the Michigan River wetland (Turner & Rabalais, 2003). By keeping a constant zooplankton:phytoplankton (Zp:Ph) ratio in off-shore zone has helped resource managers to maintain clear water quality of the Michigan River wetland system in recent decades (Turner & Rabalais, 2003).



Box 1.

5.2.1 Phosphorous (P) and nitrogen (N) management

Phosphorous compounds are measured as total phosphorous (TP) and soluble reactive phosphorous (SRP) and nitrogen is measured as total nitrogen (TN), ammonia (NH₃⁺), nitrate (NO₃⁻) and nitrite (NO₂). Understanding the dynamics of P and N is essential whilst managing water quality. In shallow enriched lakes internal cycling of P can result in highly variable TP concentrations, often a strong seasonal variation occurs as well as this can usually be high in the summer when P is released from the sediment under anoxic conditions. Nitrogen concentrations however in summer are low in shallow temperate lakes due to an increased assimilation by algae. The high algal biomass leads to oxygen depletion and loss of biodiversity and fish mortality. Understanding of the environmental

perturbations of P and N and toxic algal blooms has been advanced through the use of the PCR-DNA technique on cladoceran eggs. Weider et al. (1997) reported that there is a link between changes in allozyme allele and eutrophication caused by increased nitrogen concentrations and algal bloom in the European lowland wetland systems. External nutrient loading in some wetlands from the point source has been controlled by external sewage treatment. However, recovery of the shallow floodplains wetland ecosystems has been delayed due to internal phosphorous loading. Cladoceran-inferred transfer function for TP has been developed in north Europe in order to examine the relationship between zooplankton assemblages and P-induced eutrophication in lowland lake system (e.g., Brodersen et al., 1998). Some benthic chydorid cladocerans are reported to have been predominantly occurring in lowland lakes with relatively high algal productivity. Ecological effect of pollution in interconnected shallow floodplain lakes of the River Erewash system in the UK suggest that significant P and N enrichment in the catchment over the past decades have resulted in a switch from submerged macrophytes to phytoplankton dominant system which have altered macro-and-micro-invertebrates communities over a range of time scales in the past (Sayer et al., 1999; Sayer & Roberts, 2001). Evaluation of P and N concentrations in lowland large river floodplain lakes at temporal and spatial scales using the micro-crustacean assemblages provides a crucial understanding of the land use activity and ecosystem change.

5.2.2 Controlling acidification

Unlike upland lakes the effects of acidification on water quality of large river floodplain lakes is relatively less studied. The impacts of acid deposition on upland rivers and lakes of Europe are shown to have influenced negatively on ecosystem structure and function as a result of sulphur-induced acid rain in the past. In order to improve water quality, attempts were made to reconstruct acidity (lake water pH) inferred by a range of biological proxies archived in lake sediment (Battarbee, 2000). Using modern remains of zooplankton, cladoceran-based pH transfer function was developed. Cladocerans responded very well to acidification of a range of lakes (N=22) distributed across Germany and Austria over the past. The reconstruction of pH using a sediment core derived from the Lake Großer Arbersee shows that a severe decrease in pH in this lake from about 6 to values of about 4.8 over the past decades (Krause-Dellin & Steinberg, 1986). Some acidobiontic species of cladoceran such as *Alonella exigua* preferring pH less than 5.5 are reported to have survived. Information regarding changes in micro-crustacean assemblages and diversity provides the timing of catchment modification of large river floodplain lakes by humans and help resource managers control acidification. Some endemic zooplankton species of copepods and cladocerans in Australian rivers and wetland systems are reported to have been associated with low water pH, which in turn is regarded as zooplankton preferring habitats with dominant granites and soil types (Tayler et al., 1996). Sulphidic acidification in the Murray Darling Basin is rapid (Baldwin & Mitchell, 2000). Sulphur present in floodplain sediments are exposed to reduce sulphide due to prolonged drought and river regulations enhancing acidification (Hall et al., 2006). A range of ecological effects of sulphidic acidification has been documented in the Murray River Basin. However, the timing for water quality change has not yet been tested using micro-crustaceans. Sulphidic sediment influence hatchability of micro-crustaceans and reduce diversity of acid-sensitive taxa. The use of these animals can help identifying habitat types that are exposed to sulphidic process and reconstructing acidification over various time scales.

5.2.3 Controlling salinisation

Salinity is becoming an increasingly challenging issue for managing water quality and ecosystems of many lowland riverine floodplain wetlands worldwide. Riverine floodplains of coastal zones are frequently inundated by saline water as a result of sea level rise (Schallenberg et al., 2003). Micro-crustaceans can be utilized to manage water quality in wetland since increased salinity in wetlands cause physiological stress in zooplankton resulting from limited osmoregulatory function influencing feeding rate, growth, reproduction, body size, life span and survival capacity. Cladocerans such as *Sida* and *Simocephalus* show optima very close to the mean value of salinity ranging between 0.2 and 17.4‰ (Aminsick et al., 2005). Amongst chydorids, *Acroperus harpae*, *Graptoleberis testudinaria*, *Alonella nana* and *E. lamellatus* prefer low salinity ranges while *Oxyurella* and *Leydigia* prefer high salinity ranges (Aminsick et al., 2005). Transfer function weighted averaging (WA) models for salinity show that cladoceran assemblages are excellent proxies for reconstructing salt concentration in wetlands and help identifying the timing of the release of salt into the system (Bos et al., 1999).

Salinity in arid and semi-arid rivers is influenced by prolonged drought, river regulation, periodic low flows and intensive land use activities in river catchments (Nielsen et al., 2003a). Unlike lowland coastal zones, arid and semi-arid rivers receive salts from groundwater and terrestrial materials via the rock weathering or from the transboundary pollutants from the atmosphere. During low flows, the combination of evaporation and groundwater intrusions assist to increase the natural salinity levels (Jolly et al., 2001). In Murray Darling River, Australia, however, the natural processes have been significantly altered by humans following the European arrival (Jolly et al., 2001). There have been noticeable differences in species richness of micro-crustaceans in low-flows and high salinity periods (Nielsen et al., 2003b, 2007). Zooplankton sampled from longitudinal gradients of the South American arid rivers such as the Salado River (Buenos Aires Province, Argentina) indicates that they have species-specific variations in salinity optima and tolerances (Claps et al. 2009). Hatching of resting eggs of zooplankton is reported to have reduced in wetland with high salinity levels (Skinner et al., 2001). Variance partitioning of benthic cladocerans response to lake water salinity in Kenya suggest that salinity explained more than 51% of the observed variations (Verschuren et al., 2000). Recently Barry et al. (2005) assessed the hatching response of *Daphnia ephippia* to the diatom inferred salinity levels of a sediment core collected from a lake in southwest Victoria, Australia, where significant differences in ephippial densities and hatching were observed with respect to varying salinity levels. Given the increased sensitivity to salinity by cladocerans these organisms can be used to quantify a threshold of salt that are appropriate for a healthy floodplain wetland ecosystems.

5.2.4 Management of toxic substances

Pollution caused by toxic substances is becoming a major threat to diversity, composition and abundances of biota in large river floodplain lakes. Most trace metals have natural mineral origins and it is essential to understand the amount of mineral inputs into wetlands. Records of anthropogenic lead pollution in European lakes are reported to have determined by ratios of $^{206}\text{Pb}/^{207}\text{Pb}$ in sediment. Natural ratios of the isotope ($^{206}\text{Pb}:^{207}\text{Pb}$) are generally higher than those of anthropogenically induced lead pollution and can be determined by analysis of floodplain lake sediments. Recently pyrite pollution has become one of major issues of the organic metallic toxicity across the Murray Darling Basin, Australia due to the

exposure of sulphur contained sediments following the river regulations and prolonged drought. The processes controlling the FeS pollution in the Murray Darling Basin floodplain lakes is unknown. Establishing a macrophyte colony tolerant to sulphur-induced acidification can be useful. Engelhardt & Ritchie (2001) examined the role of aquatic macrophytes diversity in ecosystem functioning. Greater species richness and biomass of macrophytes tend to lower the chemical activities by filtering the particulate elements from the water and assisting ecosystem functioning and enhancing the wetland management practices. Phytoplankton zooplankton such as *Eurycerus* and *Graptoleberis* (Quade, 1969) are proven to be useful for reconstructing past macrophyte cover in some billabongs in Australia (Ogden, 2000). Information regarding macrophyte cover in the past can help elucidating organometallic toxicity in lakes over time. Earliest records of POPs in lake sediments are generally limited, but the PAHs are produced from the combustion of organic matter, and generally have a long term record of past events (e.g. forest fires) in sediment. Sedimentary ratio of 1,7-dimethylphenanthrene and 2,6-dimethylphenanthrene has been used as indicator of wood combustion (Fermández et al. 2000). Recently Kattel and Sirocko (2011) have used cladoceran subfossils to identify the range of past anthropogenic regimes including the alteration of forest catchments in a European maar lake.

5.3 Management of invasive species

The endemic floodplain lake ecosystems of the North America were invaded by exotic flora and fauna soon after their introduction (e.g., Mooney & Cleland, 2001). The invading microcrustaceans, *Daphnia lumholtzi* also colonised the Upper Paraná River floodplain lakes of South America soon after their introduction. Favourable temperature, water transparency and decreased nutrient concentrations supported the expansion of *D. lumholtzi* in South American wetland system (Simões et al., 2009). The actual effects of alien species on microcrustacean assemblages are not known, but microcrustacean assemblages are useful for understanding the impacts and timing of invasion on endemic ecosystems. Less *Daphnia* ephippia are deposited in sediments derived from introduced plants such as *Plantago* and *Pinus* in the Murray Darling River floodplain wetlands in Australia (Reid et al., 2007). Caudal remains of exotic zooplankton *Bythotrephes* sp. in sediment of a Canadian lake were useful to track the energy flow toward the higher trophic level as *Bythotrephes* sp. consistently reduced endemic crustacean populations that were important diet of fish (Hall & Yan, 1997). The timing of geographic distribution pattern of exotic *Daphnia* in North America such as *D. galeata* is unnoticed as a result of extensive hybridization with native *Daphnia*. Allozyme analysis of *Daphnia* ephippia in Europe and North America have become useful for reconstructing timing of invasion (Taylor & Hebert, 1993) and a genetic analysis of cladoceran fossil ephippia have advanced further the knowledge of global distribution patterns and impacts of exotic species on endemic ecosystems (Hairston et al., 1999).

5.4 Mitigation of climate change

Climate change exacerbates the ecological effects of large river floodplain lakes by altering the dynamics of nutrients, pH, salinity and organic toxic compounds such as PAHs and POPs. Mitigation is an action to reduce the risk and hazards of climate associated impacts on ecosystems (IPCC, 2007). Microcrustacean assemblages are useful for understanding these impacts on large river floodplain lakes ecosystems and help configuring appropriate mitigation strategies. Cladocerans show variation in temperature optima and tolerance

ranges. Subfossil cladocerans assemblages can help identifying climate change in a range of time scales in the past (Battarbee, 2000). Climate change such as amount of rainfall causes enlargement and contraction of wetland habitats leading to distinct variations in the relative abundances of littoral and planktonic cladoceran assemblages (Alhonen, 1970). The ratio of littoral:planktonic (L:P) cladocerans serves as significant indicator of climate-induced hydrological regime shifts in shallow floodplain lakes (Ogden, 2000). Cladoceran assemblages and resting eggs have responded to the termination of the last glacial maximum (LGM) and the Holocene sea level rise in coastal regions (Kattel & Augustinus, 2010). Development of a cladoceran-inferred calibration model for temperature is useful to understand the impacts of climate change on ecosystems over a range of time scales in the past and help developing effective management strategies to reduce vulnerability on time (e.g., Lotter et al., 1997; Kattel et al., 2008).

6. Conclusion

Management of large river floodplains lake ecosystems have become increasingly challenging in recent decades as a result of coupled human-climate disturbances. A range of theoretical models being developed in large river systems, have become useful to understand floodplain lake ecosystems processes and develop effective management strategies for restoration of these lakes. However, unprecedented impacts such as river regulation, land use activity, introduction of exotic species and rapid climate warming in recent decades on floodplains lake ecosystems together have intensified the effects and made the ecosystem processes complex to understand. The use of micro-crustaceans particularly the cladocerans are increasingly useful indicator to infer the changes occurring in large river floodplain lakes. Cladocerans play an invaluable role in food web structure and dynamics and they have a wide range of optima and tolerances to temperature as well as other environmental perturbations in floodplains systems. The use of cladoceran subfossils and their ephippia has further reformed our understanding of ecological processes of floodplains lakes of large river system. A long term investigation of the changes in a range of abiotic and biotic assemblages including micro-crustaceans is important to achieve conservation and management goals of large river floodplain lakes ecosystems effectively. Appropriate quantitative and qualitative assessments of these ecosystems can help understanding the past changes and developing future prediction models that provide appropriate information of risks of environmental vulnerabilities and enhances mitigation measures. However, such effort can only be achieved through wider collaborations amongst scientific communities, governments and international organisations.

7. References

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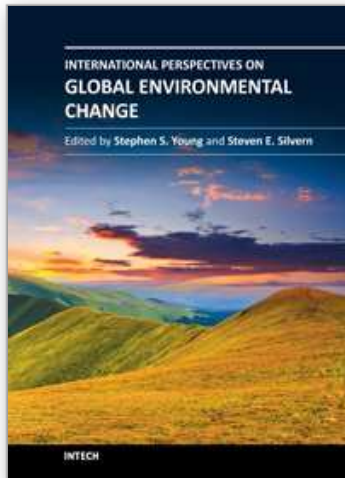
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