Literature Review

Quantifying the impact of environmental water on salinity and biotic responses in semi-arid river ecosystems

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Introduction

Globally, environmental degradation is happening at a more dramatic rate and is more widespread than at any other time in history (Crutzen 2002). Most significantly, anthropogenic emissions of carbon dioxide have caused the alteration of global climate, potentially for millennium to come (Crutzen 2002). Such widespread degradation has raised concerns of the sustainability for the Earth's environment into the future (Steffen et al. 2007). So significant and pervasive are the effects of human actions on the environment that many authors have suggested that the Earth has entered into a new geological epoch, one which is defined by human actions, termed the 'Anthropocene' (Crutzen 2002; Steffen et al. 2007; Zalasiewicz et al. 2008). Furthermore, human populations are rapidly expanding, with numbers estimated to reach 10 billion people by the end of this century, placing enormous and expanding demands on the Earth's resources with approximately 30-50% of the Earth's land surface already exploited by humans (Crutzen 2002).

Freshwater is crucial for all life. Globally, freshwater stocks have been, and will continue to be influenced by human demands. Regulation of freshwater systems is now commonplace with humans currently exploiting more half of the world's fresh waters (Crutzen 2002). As a result, global freshwater biodiversity is now gravely threatened (Dudgeon et al. 2006). Fresh waters hold a disproportionate richness of habitats for plants and animals, relative to those available in marine and terrestrial ecosystems (Dudgeon et al., 2006). This is demonstrated by the fact that fresh waters constitute only 0.01% of the world's water and cover around 0.8% of the earth's surface, yet they support 6% of all described species on the globe (Dudgeon et al. 2006; Heino et al. 2009). The crucial value of freshwater ecosystems coupled with the excessive anthropogenic effects that are currently influencing these ecosystems means it is critical for resource managers to carefully consider how to best manage freshwater ecosystems, to sustain biodiversity, thus preserve ecosystem functioning (Garcia-Moreno et al. 2014).

Salinisation is a significant threat to many freshwater ecosystems in particular to ecosystems that are in dry regions (Ghassemi et al. 1995). Understanding the response of salinisation to freshwater biota is crucial to the preservation of biodiversity (James et al. 2003; Jolly et al. 2008; Nielsen et al. 2003b). In Australia, and elsewhere around the world, there has been considerable research into the effects of salinisation particularly for adult life-stages of biota, but there is currently limited understanding of sub-lethal effects versus tolerances for earlier life-stages of biota (Jolly et al. 2008; Nielsen et al. 2003b). This remains a significant knowledge gap, one that should be addressed when considering the management of these ecosystems for the preservation of biodiversity.

The aim of this literature review is to provide an overview of current threats, particularity salinisation, to biota and ecosystem functioning, along with management strategies in wetland ecosystems in semi-arid regions.

Ecosystem function

Ecosystem health and functioning

Ecosystem function has been defined as "… the activities or processes that characterise an ecosystem" (Brinson and Rheinhardt 1996, p.70). Ecosystem functioning occurs as a result of the interaction of physical, chemical and biological processes along with the structural features of an ecosystem (Maltby and Barker 2009). This interaction, and the resultant functioning, is responsible for provision of valuable goods and services, often described as ecosystem services (Maltby and Barker 2009). These ecosystem services are the elements of natural systems that are perceived as beneficial to human society (Cairns and Pratt 1995). As the global human population grows, so too does the demand for ecosystem services, while simultaneously, the sustainable production of these services is either under threat or in decline (Brauman et al. 2007). In an effort to raise the understanding and awareness of human dependence on ecosystem services, the former United Nations Secretary General Kofi Annan called for a global-scale research effort in 2000 that culminated in the Millennium Ecosystem Assessment, which remains the benchmark for ecosystem service research (Brauman et al. 2007; Maltby and Barker 2009). The key findings from this research were published in a synthesis of the work and briefly included: (1) that to meet the need of growing demands over the last 50 years, humans have altered ecosystems at unprecedented levels; (2) these changes have substantially contributed to economic growth although they have come at a cost of degraded ecosystem services; (3) degradation of ecosystem services could rapidly increase in the first half of this century; and (4) improvements to ecosystem services are possible although it would require significant changes in policy, institutions and practices that are presently not underway (MA 2005).

Wetland functioning has been defined as "…the normal or characteristic activities that take place in a wetland or simply the things wetlands do" (Smith et al. 1995, p.21) and the functions and processes that occur in wetlands are demonstrable at global, ecosystem and population levels (Richardson 1994). Wetland functioning provides important ecosystem services which are of benefit to humans and wildlife populations and are important for environmental maintenance (Maltby and Barker 2009). The Millennium Ecosystem Assessment identified four categories in which various ecosystem services fall, including provisioning services (e.g. the provision of timber, fresh water and food), regulatory services (e.g. flood mitigation and climate stabilisation), cultural services, (e.g. spiritual, recreational and aesthetic values), and supporting services, which are the ecosystem processes that underpin the previously stated categories. The services that are provided by the wetlands are directly transferable into economic value (Maltby and Barker 2009); for example, wetlands are able to remove nitrogen from agricultural lands as a result of the nitrogen cycling within the wetland which can be converted into a dollar value associated with the avoided cost of water purification (Hart et al. 1991). Despite this, wetlands globally are often undervalued, sometimes even being thought to be of as having negative value, thus the services that they provide often go unrecognised (Brauman et al. 2007; Maltby and Barker 2009).

The Millennium Ecosystem Assessment (2005) clearly demonstrated the ecosystem services provided by wetlands, and aquatic ecosystems more generally, and how crucial those services are to human well-being. In response to this reliance, the health and consequent functioning of aquatic ecosystems has become a topical issue for scientists and natural resource managers alike (Xu et al. 1999; Maltby 2009; Fairweather 1999a; Boulton 1999). Furthermore, Fairweather (1999a) argued that there is a need to adopt a whole-system viewpoint when considering environmental assessments.

Indicators of ecosystem functioning

The health of an ecosystem has been defined in terms of the system's organisation, vigour and resilience (Burkhard et al. 2008). Ecosystems are referred to as healthy if they are resilient and the provision of ecosystem services is sustainable (Burkhard et al. 2008), implying that the ecosystem has the ability to maintain its structure (organisation) and function (vigour) and is resilient to stress over time (Rapport et al. 1998; Fairweather 1999a; Burkhard et al. 2008). Ecosystem health potentially includes all abiotic and biotic components of the landscape, including humans as a key element, particularly regarding the management of ecosystems (Fairweather 1999a). Many environmental managers now consider the protection of ecosystem health as a primary goal (Xu et al. 2001).

Because ecosystem health cannot be directly measured or observed, surrogate measures, i.e. indicators, must be developed (Burkhard et al. 2008). Indicators provide evidence of, or a signal about, the state of conditions being measured; thus they are used to gain insight into the whole ecosystem by only examining a small section of that ecosystem (Bertram et al. 2005). Suitable indicators of ecosystem health should consider both the structural and functional components of ecosystems at a range of spatial and temporal scales (Burkhard et al. 2008). Indicators should be measured routinely and be able to be collected at low cost by non-specialised fieldworkers (Fairweather 1999a; Boulton 1999). It is also crucial that indicators are validated to ensure that the interpretation of the data collected is clear (Fairweather 1999a; Boulton 1999). In order to be effective, indicators should alert managers to imminent events that will have a deleterious effect on an ecosystem with sufficient time to take preventative action (Boulton 1999). A pertinent point to acknowledge when considering indicators of ecosystem health is that they should not replace ongoing, comprehensive work, rather they should be informed by that work (Fairweather 1999b).

Historically, indicators of ecosystem health have measured particular species or components of an ecosystem (Xu et al. 1999). Although this does provide valuable information, Fairweather (1999a, p.445) explains that such measures are not sufficient to reveal all that may be wrong with an ecosystem and that such an approach represents a "very static view of ecology". Just measuring components fails to measure the dynamic functioning of an ecosystem (Fairweather 1999a). An alternative to measuring such structural or static components of an ecosystem is to measure indicators that represent important ecological processes (e.g. decomposition or primary production) from which conclusions regarding the whole system can be drawn (Burkhard et al. 2008). Using ecological processes is still a relatively new concept (Lester et al. 2011) and is often neglected in environmental assessments, despite the well-known and accepted concept that ecosystem processes often shape the structure of ecosystems (Fairweather 1999a).

Traditional ecological indicator programs that have focused on measuring structural components of an ecosystem only provide a snapshot of a particular metric (e.g. species composition, abundance or richness) then presume to relate this to an important ecological process (Fairweather 1999a). Bunn and Davies (2000) stress the need for caution when interpreting results obtained from measures of structural components, pointing out that changes in the ecological patterns of structural components are often a result of naturally-occurring changes in ecological processes rather than changes that may have been induced anthropogenically. Furthermore, marked changes in ecological process do not always result in changes in structural patterns.

Developing indicators of ecological process would have many advantages over traditional indicators, allowing greater insight into how an ecosystem functions in relation to its structure over large temporal and spatial scales (Fairweather 1999a; Fairweather 1999b). Measuring ecosystem processes often integrates responses of multiple species (Lester et al. 2011), requires less technical expertise when compared to traditional structural measures (Fairweather 1999a) and allows for the relatively rapid assessment of trends in an ecosystem thus fulfilling the requirement of an indicator to provide an early warning signal (Fairweather 1999a, Boulton 1999).

Functional assessment procedures

Based on the evidence above, there is an obvious need to develop a methodology that can assess the functioning of ecosystems like wetlands (Maltby and Barker 2009). Efforts to address this need were initially conducted in Northern America and attempted to treat a wetland as a single functional unit. It was realised that the evaluation of an entire wetland was often not possible as they were too complex, thus a methodology based on assessing a sub-area, known as an assessment area, was adopted (Maltby 2009). More recently a methodology for assessing wetland functioning in Europe has been developed (Maltby, 2009). The Functional Assessment Procedures (FAPs) are based on the identification of sub-areas of a wetland called hydrogeomorphic units which are defined as “areas of homogeneous geomorphology, hydrology and/or hydrogeology, and under normal conditions, homogeneous soil/sediment” (Maltby et al. 1998).

FAPs were designed with the notion that they could be carried out by both experts and non-experts alike in a relatively rapid manner, would not require highly specialised equipment and should first steps in a multi-tiered approach for natural resource managers wanting to assess the value of a wetland (Maltby 2009). Within the FAPs there are three main stages: firstly, HGMUs are delineated within an assessment area in the wetland; secondly, the ecological process are identified and evaluated within the HGMUs; and finally, an assessment is made (Maltby and Barker 2009). The delineation of the HGMU within a given assessment area is carried out by identifying site-specific characteristics using either aerial photographs or direct field observations (Maltby 2009). The functioning of the HGMUs is assessed by the occurrence and performance of the ecological processes. This is done by answering process-specific questions that are based on observable characteristics, then assigning these processes a score based on their performance (Maltby 2009). The final stage uses the data that have been collected in the previous stages to firstly assign a score to the processes, then to functions and finally for the entire HGMU (Maltby 2009). Once all the HGMUs within an assessment area have been assessed their scores can then be collated to evaluate either, a specific function across HGMUs within the assessment area, all functions occurring in an individual HGMU or all functions that are occurring across the entire assessment area

(Maltby & Barker 2009). Developing a tool similar to the FAPs in Australia to enable natural resource managers effectively assess wetland functioning would be highly valuable.

Semi-arid and arid environments

Semi-arid and arid regions are defined as areas that annual rainfalls of below 500 mm, where the potential for evapotranspiration far outweighs precipitation (Jolly et al. 2008) and are referred to as dryland regions. Dryland regions encompass approximately 47% of Earth's land area (Williams 1999). As of 1990, these regions were inhabited by more than 400 million people, although this figure was predicted to rise, placing ever-increasing strain on the freshwater resources that exist within them (Williams 1999). These regions are characterised by extreme climatic and hydrological variability which occurs seasonally, inter-annually and over larger timeframes (Jolly et al. 2008). This variability includes severe and frequent periods of drought accompanied by infrequent but significant flooding events (Jolly et al. 2008). Thus, this climatic and consequent hydrological variability is a key ecological driver in these regions (Jolly et al. 2008).

A paradoxical situation exists when considering dryland regions and the aquatic systems that exist within them. Namely, some of the world's largest rivers, for example the Colorado, Nile and Murray-Darling Basins, run through these relatively low-rainfall regions although they originate in areas with more mesic climates (Williams 1999). Several of the world's most important wetlands including the Kafue Flats (Zambia), Okavango Delta (Botswana), Prairie Potholes (North America) and the Coorong and Lakes Alexandrina and Albert (Australia) are also situated in dryland regions.

Wetlands found within dryland regions provide crucial habitat for unique biota in otherwise dry environments (Jolly et al. 2008) and the faunal diversity can equal, or in some cases exceed, that of temperate or tropical wetlands (Williams 1998).

Salinity

Salinisation in dryland regions

Salinisation is defined as the process by which the concentration of total dissolved solids is increased in water and soils, either by natural or anthropogenic means and also encompasses the ecological and economic consequences of phenomenon (Bailey et al. 2006; Ghassemi et al. 1995). Naturally-occurring salinisation is a result of extended periods of regional salt accumulation (Ghassemi et al. 1995). This salt can have multiple origins including rock weathering, aquifers, deposition of salt present in rain water, connate water in sediments and aeolian clays (Ghassemi et al. 1995; Jolly et al. 2008). Natural increases in soil and water salinity are referred to as primary salinisation.

The salt that is stored in groundwater and the soil profile via the aforementioned processes can also be mobilised through the landscape as a result of anthropogenic land use which raise the water table close the soil surface. Then, as water is evaporated, the salt is left behind, resulting in soil salinisation (Ghassemi et al. 1995). This salt can then move laterally through the landscape resulting in salinisation of rivers, wetlands, floodplains and riparian zones, as these systems tend to occupy the lowest points in the landscape (James et al. 2003). The process of salinisation as a result of human-induced change is known as secondary salinisation (Ghassemi et al. 1995) and is most commonly encountered in dryland regions of the globe (Williams 2001). Ionically, $NaCl$, is the dominant salt that effects wetlands in dryland regions (Bailey at al. 2006).

For many countries, salinisation and, in particular, secondary salinisation, poses the most significant threat to water resources (Ghassemi et al. 1995; James et al. 2003; Jolly et al. 2008; Williams 2001). Globally, an area of 160 million hectares is currently affected by secondary salinisation spanning numerous countries including Argentina, Australia, China, Commonwealth of Independent States, India, Iran, Iraq, South Africa, Thailand and the United States of America (Ghassemi et al. 1995; Williams 1999).

Wetlands in dryland regions experience natural variability in salinities due to inherent environmental conditions including irregular rainfall, groundwater inflows and high evaporation (Jolly et al. 2008). Salinisation resulting from anthropogenic land-use changes can be avoided naturally in wetter regions, as higher rainfall allows the leaching of substances, including salt, through the soil (Bailey et al. 2006). Dryland regions rarely experience drenching rain and thus secondary salinisation poses a major problem (Ghassemi et al. 1995). Testament to this, the Australian and African continents contain the largest dryland regions and also have the largest area of land affected by salinization in the world (Ghassemi et al. 1995).

Salinisation in Australia

In Australia, 5.7 million hectares are either 'at risk' or 'affected' by dry land salinity and thus salinisation is now considered the most serious issue facing land and water, especially as this figure is predicted to rise to ~17 million hectares by 2050 (James et al. 2009).

Much of the research into salinisation of freshwater ecosystems in has occurred in Australia as a result of the extent of salinisation in the country (Nielsen et al. 2003b), although it is likely that there will be similarities between the Australian experience and dryland environments elsewhere (Kingsford 2006).

Implications of salinisation for freshwater biota

There is large variability in the characteristics of aquatic ecosystems to which the biota that live within must adapt. In order to maintain the required balance between water and dissolved ions in cells and tissues freshwater biota have developed an array of physiological mechanisms and adaptations (Hart et al. 1991). Increasing salinity in the environment disturbs the ability of freshwater biota to maintain this balance, resulting in either a deficiency of water or surplus of ions in the cells often resulting in toxic effects including mortality (Hart et al. 1991).

Macrophytes

Macrophytes are crucial to the function of freshwater ecosystems playing many important roles, for example they are important to the maintenance of water quality by affecting nutrient inputs and suspension of sediments (Nurminen 2003). Macrophytes also influence epiphyton, phytoplankton and invertebrate populations, thus also influence fish communities (Nurminen 2003).

Increased stress is placed on freshwater macrophytes by rising salinity as plants require extra energy or specialised mechanisms to cope (Keddy 2010). (Nielsen et al. 2003b) proposed that freshwater macrophytes are generally not tolerant of increasing salinities, finding that macrophytes observed in the field have an upper salinity tolerance of approximately 4000 mg L^{-1}. At this point, halophytic species such as *Ruppia* spp. tend to replace non-halophytic species such as *Myriophyllum* spp. (Nielsen et al. 2003b). This finding aligns with those of other authors (Brock and Lane 1983; Hart et al. 1991) although more recently (Bailey et al. 2006) and (James et al. 2009) have suggested that this value should be slightly higher, at 5000 mg L^{-1}.

Increased salinity generally results in reduced vigour in macrophytes as their ability to extract water from there surrounding medium is disabled (Hart et al. 1991, James et al. 2009). Adverse effects are experienced by some macrophytes as salinities exceed 1000 mg L^{-1} (Nielson et al. 2003). One such effect is a reduction in growth rate, including the reduction of root and leaf development (Hart et al. 1991, Nielsen et al. 2003). Along with a reduction in plant growth rate and size, (James and Hart 1993) demonstrated that salinity above 1000 mg L^{-1} also inhibits sexual and asexual reproduction. Increasing salinity reduces the formation of below-ground tubers which are necessary for plant growth in the following year, at salinities of 6000 mg L^{-1}. For example, Warwick and Bailey (1996, cited in Nielson et al. 2003b) measured a two-thirds decrease in storage tubers in the soil of the macrophyte *Potamogeton tricarinatus.*

Invertebrates

Invertebrates exist in nearly all freshwater environments and are extremely diverse (Covich et al. 1999). They are important to the function of freshwater ecosystems as they have many important roles in ecosystem processes (Covich et al. 1999). Invertebrates are essential for decomposition so as to release nutrients that are used by aquatic angiosperms, bacteria, algae and fungi (Covich et al. 1999). They also provide a crucial food sources for higher trophic level consumers (e.g. fish and birds) while some invertebrates are predators and control numbers and distribution of their prey (Covich et al. 1999).

Freshwater invertebrates are hyper-osmotic regulators, thus are unable to maintain body fluid solute concentrations lower than of the water in which they live (Hart et al. 1991). Once salinity in the water exceeds the threshold at which invertebrates can maintain a constant ionic concentration, cellular function is lost as result of loss of water from the cells and an individual dies (Hart et al. 1991). James et al. (2003) found that, at salinities above 2000 mg L^{-1}, adverse effects of salinity will be experienced by some macroinvertebrates. Macroinvertebrates that are structurally simple, often lacking an impermeable exoskeleton (e.g. Hydra, Molluscs and Flatworms) are likely to exhibit the greatest sensitivity to salinity (Hart et al. 1991).

Relative to other biotic groups, there is more information on the salinity tolerances of macroinvertebrates as they are often used for biomonitoring of health in aquatic ecosystems

(Nielson et al. 2003, Boon et al. 2002). Despite this, very little is currently known of the early life-stages of biota (Nielson 2003).

Effects of salinisation on early life-stages of biota

Stress that alters the condition of an organism without causing mortality is referred to a sublethal stress. Generally, adult life-stage biota are far more tolerant to salinity than their earlier life-stages (James et al. 2003) and the observation of any given life-stage at a particular salinity does not necessarily infer that particular taxon will be able to complete their life cycle at that salinity (Kefford et al. 2004a). One reason for the greater intolerance to increasing salinity of the smaller, early life-stage biota is that they have a higher surface area to volume ratio, resulting in higher potential for water and ion transfer (Kefford et al. 2004b). In Australia, there has been considerable research into the salinity tolerance of freshwater biota in response to the threat of salinisation, although the majority of data collected has been for adult life-stages of biota (Hart et al. 1991, Boon et al 2002). In comparison, information regarding the salinity tolerances of earlier life-stage biota is poorly understood (Nielsen et al. 2003a; Nielsen et al. 2003b; Nielsen et al. 2007; Kefford et al. 2004a; Brock et al. 2005). The lower tolerance of early life-stages has clear implications when considering the management of freshwater ecosystems as purely managing for the salinity thresholds of adult life-stage biota could underestimate the effects of salinity in an ecosystem thus potentially resulting in a reduction in recruitment and thus a decline in aquatic populations through time (James et al. 2003; Kefford et al. 2005; Kefford et al. 2007; Jolly et al. 2008). James et al. (2003) point out the importance of addressing this short fall in the knowledge, contending that understanding how increased salinity will effect early life-stage biota and the recruitment and reproductive processes could be the key to successfully preserving biodiversity in aquatic ecosystems.

In dryland regions, dormant life-stages including egg, seeds, spores and asexual propagules form the seed-bank and are important biodiversity reservoirs (Nielsen et al. 2003a). The seedlings and juvenile zooplankton that emerge from the seed-bank are potentially less tolerant to salinity than the adult life-stages (Jolly et al. 2008). (Kefford et al. 2004a) provide evidence that early life-stage biota can be more sensitive to that of their older life-stages when they compared the salinity tolerance of 12 invertebrate taxa (10 from Australia and two from South Africa). While the authors found the early life-stages of some taxa have a similar salinity tolerances to that of their older life-stage counterparts, for example the South African decapod *Caridina nilotica* and the Australian Plecopteran *Plectrocnemia* sp. (Table 8. Kefford et al. 2004a), across all taxa studied they found that the tolerances of early life-stage of biota were almost half of that of the older life-stages.

Emergence of seedlings and zooplankton in response to altered salinity regime was investigated by Brock et al. (2005) and Nielson et al. (2003) using sediments collected from the Macquarie Marshes, a large temporary wetland in the Macquarie river catchment, NSW. The experiments involved outdoor mesocosm experiments in which the authors manipulated salinity and water regime. The authors found, in both instances, that when sediment were submerged in water that was maintained at constant salinities above 1000 mg L^{-1}, a reduction of plant and zooplankton emergence was observed and at that above 5000 mg L^{-1}, emergence and survival was restricted to only a few taxa (Brock et al. 2005; Nielsen et al. 2003a). Further to this, Nielsen et al. (2007) were able demonstrate that emergence of seedlings and zooplankton from the seed bank was not adversely effected by short pulses of high salinity followed by a return to freshwater. Nielson et al. (2008) also demonstrated that assemblages of macrophytes and zooplankton that emerge from sediments are influenced by a gradual increase in salinity (over 6 months) which affected emerging macrophytes and zooplankton in the same way as a sudden increase in salinity.

Though the evidence suggests that salinity will have greater effects on early life-stages of biota, a recent study by Kefford et al. (2007) points out that it is difficult to produce a simple rule of thumb pertaining to the comparative salinity tolerance early life-stages and older life-stages biota of salinity tolerance. This study reported considerable diversity in the comparative salinity tolerances of early and older life-stages of biota with early life-stage tolerances ranging from 4 – 88% of the older life-stages, while some taxa did not show any difference in salinity tolerance. Both Kefford et al. (2004a) and Kefford et al. (2007) found that, generally, insect and mollusc early life-stages were more sensitive, whereas early life-stages for Crustacea were generally as tolerant as older life-stages.

Effects of salinisation in combination with other stressors

In salinised wetlands, salt is often not the only perturbation that applies stress to biota as salinisation can affect a range of water column physicochemical variables including dissolved oxygen, water density, waterlogging, temperature and pH (Bailey et al. 2006; James et al. 2003; Jolly et al. 2008). This compounding effect of combined stressors has been rarely studied and further knowledge would be highly valuable to water resource managers.

pH can have strong influences on freshwater biota, in particular low pH. Generally, values below a pH of 5.5 have been associated with deleterious effects to freshwater biota (Heino et al. 2009). Low pH, when in combination with high salinity, is believed to be extremely challenging to osmoregulation (Blinn et al. 2004). Bailey et al. (2006) cite unpublished work by K. James, B. Hart, P. Bailey and N. Warwick carried out in 1991 in which they investigated the effects of increased salinity on aquatic biota in combination with a range of other physicochemical stressors in Rafertys swamp (Central Victoria). The authors used nine mesocosms, artificially increasing salinity in six, three to 600 mg L^{-1} (low salinity) and three to 1800 mg L^{-1} (high salinity) leaving the remaining three mesocosms as controls at 140 mg L^{-1}. pH in the mesocosms started at pH 6.9. As the enclosures experienced evaporation, salinity levels tripled while pH decreased to pH 5.5 and pH 3.8 in the mesocosms elevated to 600 mg L^{-1} and 1800 mg L^{-1} respectively. The control mesocosms only fell to pH 6.4. This is a good example of how changes in salinity will affect pH and act on wetland biota synergistically. This effect could either act as a press or ramp disturbance, depending on the rate of change of the parameters (Bailey et al. 2006).

Zalizniak et al. (2009) recently investigated the effect of pH on the salinity tolerance of five freshwater invertebrate species including two insect species, *Notalina fulva* (Leptoceridae) and *Centroptilum* sp. (Baetidae), one mollusc; *Physa acuta* (Physidae), one cnidarian; *Hydra oligactis* (Hydridae) and one Ciliophora; *Paramecium caudatum* (Parameciidae). The experiment was a laboratory-based tank experiment which involved subjecting individuals from each invertebrate species to multiple combinations of pH and salinity, recording the lethal and sublethal responses (sublethal responses only measured for *Hydra oligactis and Paramecium caudatum)*. Unexpectedly, the authors found that low pH did not have a detectable effect on the salinity tolerances of the invertebrate, and furthermore the authors suggested that high salinity may even counteract the effect of low pH thus potentially explaining why pH did not affect the salinity tolerances of the invertebrates (Zalizniak et al. 2009).

Management of aquatic ecosystems

Flow regime and environmental-flows

Flow regime is a critical determinant of the ecology of rivers and their associated wetlands and alterations to the flow regime are often claimed to be the most serious and continuous threat to these ecosystems (Bunn and Arthington 2002). In a review by Bunn and Arthington (2002), the authors identified and discussed the ecological implications of altering flow regimes by outlining four ‘basic principles’ by which aquatic biodiversity is influenced by altering flow. Briefly summarised, the four principles are as follows; 1) aquatic communities are determined by the physical habitat which can be altered by alterations in flow; 2) aquatic biota life histories have evolved in response to natural flow regime; 3) alterations in flow restricts longitudinal and lateral connectivity within stream and wetlands habitats; and 4) altered flow regime can facilitate the invasion and success of exotic species (Bunn and Arthington 2002).

As alteration of flow affects freshwater biota in a number of ways, it is difficult to identify which attributes of altered flow are directly responsible for particular observed impacts (Bunn and Arthington 2002). For example, would a decline in a particular macrophyte species come as a result of altered substrate composition due to reduced flow velocity or as a result of the establishment of an exotic species of fish, such as carp that increased water turbidity as a result of feeding, or both?

Science needs to play a major role if the effects of flow alteration are to effectively mitigated, although Bunn and Arthington (2002) pointed out that science currently has a limited ability to predict and quantify the effects of altered flow. This issue is compounded by the fact that there is now greater competition for water than ever before. In the long term, to convince water resource managers that the water should be allocated to the needs of the environment, rather than for the needs of agriculture, industry and domestic uses, there needs to be demonstrable impacts of that water on freshwater biodiversity (Bunn and Arthington 2002; Davies et al. 2014).

The idea that river ecosystems, including their wetlands, need adequate water to sustain ecological process in not new, appearing in literature as early as the 1940s (Arthington et al. 2006). As previously discussed in this review, the ecological processes occurring in aquatic environments are what provide ecosystem services and thus their preservation is paramount. Since this initial recognition, there has been increasing concern of the impacts of flow regulation on freshwater biota and thus this research area has received a lot of attention from scientists who have investigated how managers can provide water for the environment (Arthington et al. 2006; Tharme 2003). The provision of water for the environment has been given numerous titles such as, ecological flows, environmental water allocation (EWA), in-stream flow and environmental flows (Davies et al. 2014). Although there are numerous names, each of these titles convey essentially the same concepts (Davies et al. 2014). This review will use the term 'environmental flows' which have been defined as "the quantity, timing, and quality of water flows required to sustain freshwater and estuarine ecosystems and the human livelihoods and well-being that depend on these ecosystems" (Brisbane Declaration 2007, p.1).

Presently, more than 200 methodologies have been developed world wide for determining environmental flows (Tharme 2003). Early methods were concerned with providing the minimum flows to sustain fish populations (Arthington et al. 2006; Davies et al. 2014; Poff et al. 1997). As previously discussed, flow regimes are naturally variable both temporally and spatially, and, the flow regime for a particular system defines riverine ecosystems structure and function along with the many adaptations of aquatic biota (Bunn and Arthington 2002; Dudgeon et al. 2006; Poff et al. 1997). Therefore, sustaining only 'minimum' flows is not sufficient to maintain riverine ecosystem functioning (Arthington et al. 2006). Instead what is needed is environmental flows that are able to mimic the natural variability within these systems which is commonly referred to as 'the natural flow regime paradigm' (Poff et al. 1997). As aquatic biota are influenced via interrelated spatial and temporal mechanisms, it is in fact critical that the environmental flow are designed to mimic the natural flow regime to be effective (Arthington et al. 2006; Bunn and Arthington 2002; Poff et al. 1997). Designing environmental water provisions that mimic natural flow regime rather, as opposed to targeting the needs of a particular species, is beneficial as it is then more likely that the entire ecosystem, or at least the section of an ecosystem that is receiving the flows, will be sustained (Arthington et al. 2006).

Flow regulation is now widespread through Australian riverine ecosystems. In order to secure water for drinking, agriculture and mining, water control schemes were instigated almost immediately after European colonisation in Australia (Arthington and Pusey 2003). Construction of structures designed to regulate flow (e.g. dams, weirs, locks) has been extensive since then, with Australia now holding the highest per capita water storage in the world (8.8 Ml x10^7) (Arthington and Pusey 2003). Most of the stored water is held in a small number of large dames (i.e. approximately 50% of the total capacity is held within the ten largest dams) (Arthington and Pusey 2003) This regulation has significantly altered the characteristics of many of Australia's riverine ecosystems; with some once-permanent water bodies becoming intermitting dry and conversely some naturally ephemeral water bodies becoming permanently inundated (Arthington and Pusey 2003; Kingsford 2000; Thoms and Sheldon 2002). The alteration of flow in freshwater ecosystems has had numerous negative ecological impacts in Australian riverine ecosystems many of which are outlined in (Arthington and Pusey 2003). One of the most conspicuous impacts was a 1000 km long toxic cyanobacteria bloom which occurred in the Barwon-Darling river in 1991 (Arthington and Pusey 2003). Another very notable impact of flow alteration in rivers is the wide spread loss of their associated wetlands, for example the up to 90% of floodplain wetlands have been lost (Arthington and Pusey 2003).

The Murray-Darling Basin (MDB) covers 1,061,500 km^2 and contains Australia's three longest rivers, the Murray, Darling and Murrumbidgee (Docker and Robinson 2013; Pittock and Finlayson 2011). There are approximately 30,000 wetlands within the MDB, of which 16 are listed as wetlands of international importance under the Ramsar Convention (Docker and Robinson 2013). Only 2% of the land within the MDB is irrigated although approximately

90% of water diversions supply irrigation (Pittock and Finlayson 2011). There was a large effort during the 20th Century to regulate the system to facilitate irrigation and to guard against drought (Pittock and Finlayson 2011) and the MDB is now the most developed basin in Australia (Kingsford 2000). Until the 1990s, water was increasing withdrawn and the health of the system was degraded (Pittock and Finlayson 2011). This degradation prompted Australian Governments to place a cap on surface water withdrawals at 1993-1994 levels. Interventions to reduce salinity and cyanobacteria blooms and to boost native fish populations by provided environmental flows were initiated but were not sufficient to curb the decline in the ecosystems health (Pittock and Finlayson 2011). In 2007, the Australian Government took control of the management of the MDB and established the Murray-Darling Basin Authority (MDBA) for the purpose of creating a plan for managing all water resources across the Basin, something that previously the four state governments (Queensland, New South Wales ,Victoria and South Australia) had not achieved (Docker and Robinson 2013). The Plan, often referred to as 'the Basin Plan' outlines a sustainable level of water that can be withdrawn from the system so as to sustain the environmental values of the Basin (Docker and Robinson 2013). The Australian Government has since been acquiring water (via direct market purchases along with investments to irrigation efficiency) in order to ensure that there is enough water to of the need of the environment (Docker and Robinson 2013).

Coorong, Lower Lakes and Murray Mouth

The Coorong, Lower Lakes and Murray Mouth (CLLMM) is one of the 16 Ramsar-listed wetlands in the MDB. It is a wetland complex at the terminus of the Murray-Darling Basin in South Australia. It is comprised of lakes, streams, lagoons and other wetlands (Kingsford et al. 2011). The CLLMM supports an exceptional array of biota, providing crucial habitat for breeding and feeding of waterbirds and native fish (Kingsford et al. 2011). The ecological importance of the CLLMM was formally recognised in 1985 by its listing on the Ramsar

Convention for Wetlands of International Significance, with the region meeting eight of Ramsar's nine possible nominating criteria, making the region a biodiversity 'hot spot' (Phillips and Muller 2006). The high level of biodiversity in the region is largely due to an historical salinity gradient that simultaneously includes freshwater lakes and the estuarine, marine and hypersaline Coorong (Lester et al. 2011). The 'Lower Lakes' refers to Lakes Alexandrina and Albert. The Lower Lakes receive the majority of freshwater inflows via the River Murray but are also fed by smaller streams including the Finniss River and Currency Creek (Kingsford et al. 2011). The Lower Lakes are separated from the Coorong and Murray Mouth by barrages that were built in the 1930s. Both lakes are large, shallow and are surround by extensive fringing vegetation (MDBA 2010). Lake Albert lies to the south-east of Lake Alexandrina, the two being joined by a narrow opening called the Narrung Narrows. Lake Albert is a terminal lake and thus flow of water into or out of the lake is determined by lake level in Lake Alexandrina (MDBA 2010). Lake Albert is typically more saline than Lake Alexandrina and water levels and quality, in particular salinity, in Lake Albert cannot be managed independently of Lake Alexandrina (MDBA 2010).

As previously discussed, the MDB has been highly modified upstream of the CLLMM region and thus the wetland complex has received far less fresh water from the River Murray than it would have naturally. Modelling indicates that current levels of diversion have reduced the mean annual flow to the Lower Murray by 71% with flows of 10,764 GL per annum simulated under without development conditions compared with 3,075 GL per annum under scenarios with current levels of development (Kingsford et al. 2011). This large reduction in flow coupled with the recent "Millennium Drought" saw the entire CLLMM region heavily degraded (Kingsford et al. 2011). Evaporation in the Lower Lakes was excessive and salinity was increased to unprecedented levels (Kingsford et al. 2011). The increased salinity in the previously freshwater Lower Lakes resulted in a variety of effects to freshwater biota.

Freshwater plant and fish species declined with estuarine and marine biota invading the once freshwater Lakes (Kingsford et al. 2011). Most notably estuarine tubeworms *(Ficopomatus enigmaticus)* attached themselves to freshwater mussels and turtles, causing mortality (Kingsford et al. 2011). As the Lower Lakes continued to dry, acid sulphate soils which occur naturally in the region were exposed, with a low pH of 2.8 being recoded (Kingsford et al. 2011). Low pH has serious implications for to freshwater biota as it can inhibiting gas exchange, osmoregulatory function and damage gills (Kingsford et al. 2011).

Conclusion

Salinisation as a result of human activity is having a demonstrable impact on freshwater ecosystems around the globe. The ecological values of wetlands in drylands such as semi-arid regions are currently being degraded by salinisation. While there has been significant research into the implications of salinisation on freshwater biota, there remains a gap in the knowledge regarding the effects on early life-stages of biota when compared to their older life-stage counterparts. In particular, there is little known of the sub-lethal effects to freshwater biota. This lack of knowledge may result in the overestimation of the tolerance of freshwater ecosystems to salinisation. Further knowledge in these area will be crucial to the effective management of wetland ecosystems in semi-arid regions. Furthermore, the goal of management of ecosystems generally, but in this instance freshwater wetland ecosystems, is shifting from that of managing for particular components of an ecosystem to a more holist approach, aiming to preserve the health of the entire ecosystem. This has been made possible by using the performance of ecological processes as indicators for ecosystem function. A method to do this has been developed for Europe, but this method is not simply transferable to regions with drier climates, such as semi-arid regions. There is much scope for further research into the application of ecological processes as indicators for wetland ecosystem functioning. In particular identifying indicators of the ecological process of biotic responses to salinity could potentially be a very useful tool for managers as a result of the significant implications that salinisation will have on freshwater ecosystems

I propose a research strategy that prioritises four key areas for further research using Lakes Alexandrina and Albert as a case study region. These are: (1) investigate the effects of salinity on early life-stages of biota by measuring the effect of salinisation on the emergence of macrophytes and invertebrates from the seed/egg bank; (2) investigate sub-lethal effects of salinisation on macroinvertebrates by measuring responses in growth and movement; (3) investigate the use of biotic responses to salinity as a potential indicator of ecosystem function; and (4) use the above knowledge gained from the investigation above to validate current management strategies within Lakes Alexandrina and Albert, including those pertaining to environmental flows.

This research will provide an increased understanding of how increased salinity will affect wetlands in semi-arid regions and it will also contribute to the current and future management tools within the region, and other semi-arid wetlands ecosystems.

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