Aquatic biodiversity and the ecological value of on-farm water storages on irrigation farms

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Abstract

Globally, natural wetlands are under threat from water resource development reflecting the need to support a growing population. In the Border Rivers Catchment in Queensland, Australia, a large irrigation industry coupled with a highly variable flow regime has necessitated the building of large on-farm water storages and often associated destruction or isolation of their natural counterparts. With the decline in abundance of natural wetlands, the presence of these storages on the floodplain has raised the question of their suitability as alternative aquatic habitat. This project aimed to investigate the diversity of storages and the structure and function of the aquatic assemblages they support compared with nearby natural wetlands. These results were then used to recommend best management practice for optimising both diversity and ecosystem function in storages.

Initially the physical variety of water storages in the Border Rivers Catchment was described and their morphology and hydrology compared to that of natural wetlands. Storages and natural wetlands formed two distinct groups based on morphology. Storages tended to be large, deep structures with a more regular shape, while natural wetlands were irregular and shallow with large perimeters. Although there was a degree of variability amongst storage sites, most fell into one group and were considered to be a 'typical' storage in this region.

After classifying the storages based on hydrology and morphology, the spatial and temporal biodiversity of the different storage groups was compared with natural wetlands. Spatial patterns (between waterbody types) in fish and macroinvertebrate assemblages composition were explored between August and October 2005 in three waterbody types; natural wetlands, typical storages and a group of storages with similar morphology to typical storages but lacking tailwater ('no tailwater' storages). In total nine fish species were collected comprising seven natives and two exotic species. Only one species, the eel-tailed catfish, *Tandanus tandanus*, was specific to natural wetlands. Although fish species diversity was similar between the two storage types and natural wetlands, there was significant spatial variation in total abundances, with typical storages having ten times the average catch size of that found in natural wetlands. In both storage types catches were dominated by bony bream, *Nematalosa*

erebi, while in natural wetlands there was a more even distribution of species. The percentage of exotic species was much lower in both storage types (< 8%) compared with natural wetlands (> 40%).

There were also significant spatial differences in macroinvertebrate assemblages; 24 samples from natural wetlands were represented by more than 14,500 individuals across 84 different taxa. In comparison, only 34 taxa from 2,142 individuals and 34 taxa from 7,611 individuals were collected in 15 samples from typical storage sites and 12 samples from 'no tailwater' sites respectively. Seventeen taxa were common to all three waterbody types while 45 taxa were specific to natural wetlands. None of the measured environmental variables explained the observed variation in macroinvertebrate assemblages between waterbody types. Temporal patterns in macroinvertebrate and zooplankton assemblages were also investigated comparing natural wetlands and typical storages during the 05/06 and 06/07 cotton seasons. Specifically, macroinvertebrate and zooplankton assemblages from typical storages were less diverse than those found in natural wetlands. Temporal variation in aquatic assemblages was also apparent with macroinvertebrate and zooplankton assemblages of both waterbody types following a similar pattern through time. However, there were no strong associations between the environmental data collected and the observed assemblage composition.

To further investigate the spatial variation observed in aquatic assemblages, two sources of colonisation of storages were examined, namely the aquatic assemblages found in extracted river water and those hatching from the egg bank. Eight different fish species including three exotics were collected during sampling of pumped river water. As was observed in storages, the catch was dominated by *N. erebi* and the contribution of exotic species to the total catch was extremely low (< 1%). Fish with a standard length of over 200 mm survived the extraction process. The eight samples of pumped river water were represented by 22 macroinvertebrate taxa and 332 individuals. This was a reduced diversity than collected from typical storages (34 taxa). In comparison, zooplankton assemblages had similar diversity to those observed in the typical storages, 25 taxa compared with 22 taxa. Sediment samples collected from the floodplain had a more diverse but less abundant egg bank (46,463

individuals from 20 taxa) than those collected from dry typical storages (70,600 individuals from 16 taxa).

The observed spatial differences in morphological and hydrological features and aquatic assemblages suggested that there may also have been variations in aquatic processes between waterbody types. Stable isotope analyses were used to investigate the major sources of energy fuelling the aquatic food webs of natural wetlands and typical storages. In general all components of the food web in typical storages were ^{13}C and ^{15}N-enriched in comparison with natural wetlands. The importance of source carbon to consumers varied greatly between species, sites, and waterbody types. Algae, in the form of biofilm and floating algae, along with zooplankton were the major sources of organic carbon to aquatic food webs, contributing 50% of organic carbon across natural wetlands and 54% across typical storages.

Storages primarily function as water supplies and their associated management makes them mostly unsuitable as ‘replacement’ wetlands. However, given the large numbers of storages across the catchment, if managed effectively, they may provide an additional source of aquatic habitat and help maintain regional biodiversity. To maximise the biodiversity of storages it will be essential to reduce the morphological homogeneity of storages across the landscape and increase habitat diversity within storages. In the future, improved design of new storages and alterations to existing storages and their management could help overcome this problem of low diversity of habitat.

As a group, storages in the Border Rivers Catchment are still fundamentally different to natural waterbodies, with storages being a mostly homogeneous group. If we are to sustain the aquatic biodiversity in the Border Rivers Catchment and other similar irrigation regions it will be necessary to preserve the spatial and temporal variation in habitat evident in natural wetlands.

Statement

I hereby declare that this work has never previously been submitted for a degree or diploma at any University and to the best of my knowledge and belief, the thesis contains no material previously published or written by another person except where due reference is made in the thesis itself.

Susan J. Lutton
May 2009

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I have published work arising from this thesis as principal author. The conceptual model and explanation presented in Chapter 1 were included in the following refereed conference paper which was produced under supervision from Dr Glenn Wilson:

> Lutton, S. (2005) Drivers affecting the aquatic biodiversity and the ecological value of water storages on irrigation properties: a conceptual model. In: *Proceedings of the 9th Annual Environmental Research Event* Hobart, Tasmania, 29 November – 2 December, 2005. (Eds N. Khanna, D. Barton, D. Beale, R. Cornforth, A. Elmahdi, J. McRae, N. Seelsaen and A. Shalav). RMIT University, Melbourne.

The main body of Chapter 3 forms the basis of the following journal manuscript which was produced along with my supervisors Dr Fran Sheldon and Prof. Stuart Bunn:

> Lutton, S.J., Sheldon F. and Bunn, S.E. (In review) On-farm water storages – do they offer a viable alternative aquatic habitat to natural wetlands? *Aquatic Conservation: Marine and Freshwater Ecosystems*.

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Chapter 1 General introduction

1.1 Floodplain wetlands

Floodplain wetlands are an important feature of large river ecosystems throughout the world. Many are intermittently connected with the main river channel especially during flood flows and, following floodwater recession, remain behind as discrete lentic ecosystems. As part of the floodplain ecosystem, wetlands provide flood attenuation, a nutrient sink, water quality amelioration, nutrient recycling, carbon storage and landscape biodiversity as well as recreational facilities and aesthetic value (Crooks *et al.*, 2001; Tockner *et al.*, 2008).

Floodplains and their wetlands offer important protection from flooding by dampening peak flows and dissipating energy, collecting water during the peaks of floods and releasing it slowly back into the river channel. Along Cooper Creek and the Paroo River of the Lake Eyre Basin in central Australia, up to 106,000 km^2 of floodplains become inundated during floods (Kingsford *et al.*, 1998). However, in order to cultivate and take advantage of the rich resources offered by floodplains, many rivers have been cut off from their floodplain and associated wetlands by levees; for example, along the Mississippi, 90% of the floodplain is leveed (Tockner *et al.*, 2008). Instead of fanning out across the floodplain and filling permanent and temporary wetlands, floodwaters are now forced through narrow, funnel like openings between the levees which often leads to a marked increase in peak flow in downstream sections (Rasmussen, 1999; Tockner *et al.*, 2008). As levees continue to be built and floodplains and wetlands are blocked off, flood elevations have risen disproportionately to river discharge (Rasmussen, 1999). Flood damage is expected to continue to increase over the next decades as humans claim new land on the floodplain for farming and homes (Mitchell, 2003).

River flows into wetlands bring fine sediments and nutrients that stimulate the growth of aquatic plants. Wetlands are sometimes considered to be the ‘kidneys’ of the river, trapping and transforming these nutrients and other contaminants that would otherwise be transferred downstream (He and Walling, 1997; Mitsch and Gosselink, 2000). Denitrification activity of floodplain soils is a key process as it constitutes a major sink of nitrate, and as such is involved in the control of nitrogen fluxes along river systems (Pinay *et al.*, 2000). On certain floodplains annual denitrification rates can be up to as much as 70% of the nitrogen deposited during floods (Pinay *et al.*, 1995). Nitrogen removal ranges from 0.5 to 2.6 kg N per Ha per day (Tockner *et al.*, 1999). By temporarily storing water wetlands also retain phosphorus which can enter wetlands with suspended solids or as dissolved phosphorus (Whigham *et al.*, 1988; Rogers *et al.*, 2009). The removal of phosphorus from water in wetlands occurs through use of phosphorus by plants and soil microbes; adsorption by aluminium and iron oxides and hydroxides; precipitation of aluminium, iron, and calcium phosphates; and burial of phosphorus adsorbed to sediments or organic matter(Johnston, 1991; Waldbridge and Struthers, 1993). When full, wetlands can act as a sink for carbon, storing it within the soil. However, when wetlands are drained the decomposition of organic matter is increased releasing carbon dioxide (Turner and Gannon, 2009).

During dry spells, wetlands provide vital refuge for aquatic plants and animals (Shiel, 1995; Walker *et al.*, 1997; Magoulick and Kobza, 2003). Most are shallow and often less turbid than the main channel (Junk *et al.*, 1989; Gell *et al.*, In press), and typically have higher rates of primary production (Tockner and Stanford, 2002; Davies *et al.*, 2008; Tockner *et al.*, 2008). Wetlands are vital to a number of taxa that take advantage of these refugia and food resources during periods of disconnection, including macroinvertebrates (Nielsen *et al.*, 1999; Sheldon *et al.*, 2002), fish (Magoulick and Kobza, 2003; Balcombe and Arthington, 2009) and waterbirds (Kingsford and Porter, 1994; Kingsford *et al.*, 2004). When floodwaters return, these refugia may provide an important source for colonisation of temporary waterbodies (Nielsen *et al.*, 1999) and the adjacent river channel.

The ecosystem goods and services provided by floodplain wetlands are recognised to be of important economic and social value. The value of services provided by floodplains are estimated to be worth US$19,580 per Ha per year, totalling a worldwide value of US$3,920 x 10^9 per year (Costanza *et al.*, 1997). The major services provided by floodplains include flood regulation (37% of total value), water supply (39%) and waste treatment (9%) (Tockner and Stanford, 2002; Tockner *et al.*, 2008), while floodplains along the Danube yield US$6.3 million worth of fish annually (Navodaru *et al.*, 2001).

1.2 Decline of natural floodplain wetlands

With an ever increasing world population the security of water for consumption, agriculture, electricity production, recreation and tourism has become a major international issue (Postel *et al.*, 1996; Fischer and Heilig, 1997; Vörösmarty *et al.*, 2000; Lake and Bond, 2007; Likens *et al.*, 2009). Unfortunately, an increase in water security is reflected in an increase in water resource development and a consequent loss of natural wetlands (Ligon *et al.*, 1995; Thomas, 1996; Lemly *et al.*, 2000). Across the globe, the regulation of large rivers for water supply has seen a decrease in the number and size of floodplain wetlands (Kingsford, 2000), such that since 1900, 50% of the world's wetlands have been lost (OECD, 1996). In Europe 95% of the original floodplain area has been converted for other uses and the remaining area has lost its natural function (Tockner *et al.*, 2008). For example, over 75% of the floodplains along the Danube and its major tributaries have been isolated from the river channel by levees and are functionally extinct (Nachtnebel, 2000; Tockner *et al.*, 2008). North America has lost about 50% of its wetland cover, primarily of riverine floodplains (Tockner and Stanford, 2002). The loss of floodplain wetlands around the world will not only have a huge economic cost (see Section 1.1) but is threatening the survival of endemic human societies that inhabit these areas (Tockner and Stanford, 2002). Predictions suggest the pressure to drain land for agriculture will intensify (OECD, 1996), with drastic consequences for remaining floodplain wetlands. Despite their considerable ecological and economic value, floodplains are expected to remain amongst the most threatened systems and to disappear faster than any other wetland type (Tockner and Stanford, 2002; Tockner *et al.*, 2008).

The situation in Australia mirrors the global trend. Dams, diversions and river management have all reduced the frequency, magnitude and duration of flooding in floodplain wetlands (Kingsford, 2000; Southwell, 2002; Cullen, 2007), while clearing, draining, filling and damming have led to large scale wetland loss, especially in southern Australia (Lukacs and Pearson, 1996). The loss and degradation of wetlands in Australia has been ongoing since European settlement 230 years ago (Bunn *et al.*, 1997). It is estimated that more than 89% of wetlands in the Murray-Darling Basin and 75% of wetlands in coastal New South Wales have been destroyed (Bunn *et al.*, 1997). On the Swan Coastal Plain (Western Australia), 75% of wetlands have been filled or drained and freshwater marsh wetlands in Victoria have been reduced by over 70% (Bunn *et al.*, 1997). Australia is also prone to frequent and severe droughts, on average every 18 years (BOM, 2008). These dry periods can be long lasting with the most recent drought beginning in 1996 and persisting in large parts of southern and eastern Australia for over a decade (MDBC, 2008).

This trend has continued in the drier regions of the Australian continent. In the Border Rivers Catchment in the northern Murray-Darling Basin, many natural wetlands associated with the Macintyre, Barwon, Weir, Dumaresq, Severn, Beardy and Mole Rivers have contracted in size (DWR, 1995) or disappeared completely (Kingsford, 1999) as a result of water resource development and agricultural expansion. This region supports a large irrigation industry growing cotton, wheat, lucerne, peanuts, corn, stone and pome fruits, vegetables, wine and table grapes on about 60,000 hectares of irrigated land (BRFF, 2007). In the Border Rivers region, high variability in annual rainfall (DWR, 1995) has necessitated the construction of impoundments which reduce the problem of an unreliable water supply and allow the storage of water for use at a later date. The low topographical relief and lack of suitable sites for major impoundments on the river main stem has led to a reliance on a large number of on-farm water storages. Direct abstraction and diversion of flood flows into these on-farm storages has resulted in reduced magnitude, duration and frequency of flow and flood events along the Macintyre River (Southwell, 2002; Thoms *et al.*, 2005). This anthropogenic activity combined with a prolonged drought has produced the driest conditions on record (e.g. Bond *et al.*, 2008). Consequently, while natural wetlands are declining in response to reduced flows and floods, these artificial wetlands have become a significant part of the floodplain landscape within the Border Rivers Catchment and other irrigation regions.

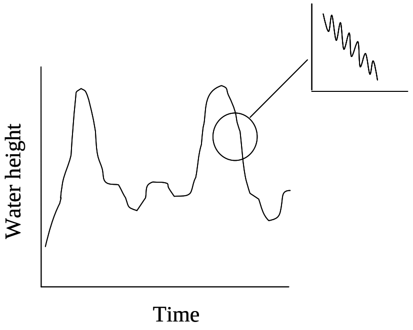
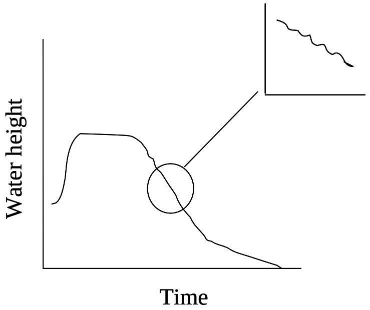
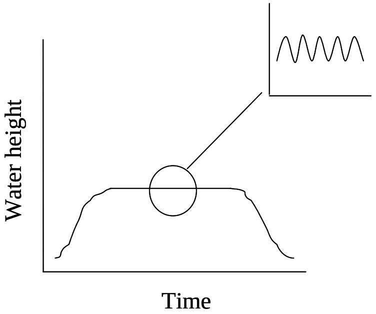
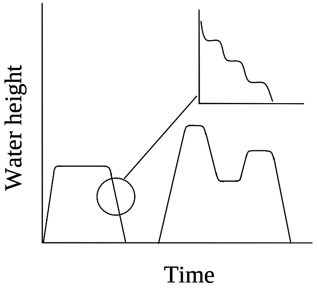
1.3 On-farm storages as alternative aquatic habitat

In some intensive agricultural regions of Australia, including the Border Rivers Catchment, water is diverted away from natural wetlands and river waterholes and is retained in on-farm water storage dams or ring tanks (herein referred to as “storages”) for the purposes of irrigation (refer to Appendix 1 and 2 for example photos). While some of the ecosystem services described in Section 1.1 may never be replicated by storages, e.g. flood attenuation, fishery resource and aesthetic value, given their increasing prevalence on the landscape, they may off-set some of the losses of natural wetlands in the region. There is the potential that, if thoughtfully designed and managed, storages may provide suitable wetland habitat and refugia for aquatic taxa. Effective management to maximise the ecological value of these storages will require an understanding of how they compare to natural aquatic habitat, such as floodplain rivers, natural floodplain wetlands and true lakes (Table 1.1).

Floodplains are defined as ‘areas of low lying land that are subject to inundation by lateral overflow water from rivers or lakes with which they are associated’ (Junk and Welcomme, 1990). Natural floodplain systems with unregulated flows and no channel or river bank engineering consist of the main river channel, major tributaries and distributaries and their floodplain area (including minor anabranches, backwaters, oxbow lakes or billabongs, and other inundation zones) (Leigh, 2008). In their natural state, riverine floodplains are characterised by high biodiversity and productivity (Tockner and Stanford, 2002). They are disturbance dominated systems, typified by a high level of habitat heterogeneity and a diverse biota adapted to high spatio-temporal heterogeneity (Tockner and Stanford, 2002). Unpredictable river flows create increased habitat diversity on the floodplain compared with the main river channel (Power *et al.*, 1995; Ward *et al.*, 1999).

Rivers can be viewed as downstream arrays of hydrogeomorphic patches resulting from shifts in hydrological and geomorphic conditions (Thorp *et al.*, 2006). The biotic communities and ecosystem processes within each patch are shaped by its unique physical characteristics in terms of size, vertical and lateral shape, structural/hydrological complexity and habitat characteristics such as mean substrate size (Thorp *et al.*, 2006). This theory is consistent with the widely acknowledged concept that the main drivers of biotic composition, structure and function in floodplain rivers are the variable flow and flood pulses that link river channels to floodplain wetlands laterally (Junk *et al.*, 1989; Amoros and Bornette, 2002; Kingsford *et al.*, 2004; Thorp *et al.*, 2006; Nhiwatiwa *et al.*, 2009). Flow is a major determinant of physical habitat in streams, which in turn is a major determinant of biotic composition (Bunn and Arthington, 2002). Davis and Barmuta (1989) noted that disturbance by flood events is one of the most important drivers of temporal and

Table 1.1: Comparison of drivers and processes in rivers, floodplain wetlands, true lakes and storages (Junk *et al.*, 1989; Power *et al.*, 1995; Ward *et al.*, 1999; Jackson *et al.*, 2001; Amoros and Bornette, 2002; Tockner and Stanford, 2002; Kingsford *et al.*, 2004; Tonn *et al.*, 2004; Marshall *et al.*, 2006; Thorp *et al.*, 2006).

	Floodplain Rivers	**Floodplain Wetlands**	**True Lakes**	**Storages**
Disturbance: flow	 HIGH	 MODERATE	 LOW	 MODERATE
Spatial heterogeneity	HIGH High habitat heterogeneity Macro – river channel Meso – backwaters, billabongs, anabranches Micro – vegetation, snags, substrata	MODERATE Meso – billabongs, oxbow lakes Micro – vegetation, snags, substrata	LOW Micro – vegetation, substrata	LOW Micro – limited; some vegetation but mostly bare bank.
Temporal heterogeneity	HIGH Disturbance from flood events	MODERATE Variable flow affecting hydrological connectivity	LOW Low levels of hydrological disturbance	VERY LOW Storage walls maintain shape despite temporal fluctuations in water level
Diversity control	ABIOTIC Flow Physical characteristics (shape, size) Habitat characteristics & diversity	ABIOTIC/BIOTIC When connected – flow When disconnected – nutrients, competition, predation	ABIOTIC/BIOTIC Physical characteristics (depth, area) Competition, predation	BIOTIC Competition, predation Connectivity & colonisation pathways
Ecosystem process e.g. food webs	Diverse Complex - algal based and allocthonous	Diverse Complex - algal based and allocthonous	Nutrient Limited Simple - algal based	Nutrient limited Simple – algal based autocthonous

spatial variability of benthic communities in streams, while flow velocity can influence algal biomass and taxonomic composition in floodplain rivers (Ryder *et al.*, 2006).

Floodplain wetlands, or billabongs as they are referred to in Australia, exist for much of the time as isolated ecosystems, only linking to the river system in times of high flows (Nielsen *et al.*, 1999). In this respect, natural floodplain wetlands may be more similar to lakes during these periods of disconnection. However, similar to floodplain rivers, the main drivers of processes in the waterbodies of riverine floodplains are related to variable flow and differences in the nature and intensity of hydrological connectivity (Table 1.1); i.e. distance of the waterbody from the river and permanent versus temporary connections to the river (Power *et al.*, 1995; Amoros and Bornette, 2002; Marshall *et al.*, 2006). Hydrology is likely to be the single most important determinant of the establishment and maintenance of wetland structure, processes and functioning (Mitsch and Gosselink, 2000). Hydrology modifies a wetlands physical and chemical conditions which in turn influence the biodiversity of the wetland (Jin, 2008).

In comparison, true lakes do not experience high levels of disturbance resulting from highly variable flows and as such have lower spatial and temporal heterogeneity compared with rivers and floodplain wetlands (Table 1.1). The Geomorphic-Trophic Model suggests that biogeochemical and biotic processes in floodplain lakes can be largely understood on the basis of morphometric and hydrological factors (Gawne and Scholz, 2006). On the Orinoco floodplain, variations in placement and morphometry cause lakes to pass through successional changes, associated with the cessation of flushing by the river, at different rates (Lewis *et al.*, 2000). Therefore biogeochemical (nutrient concentrations, organic carbon in suspension) and biotic (plankton abundance, fish community composition) variables differ among lakes (Lewis *et al.*, 2000). Several geomorphic constraints such as lake depth and surface area limit fish distribution in arctic lakes (Hershey *et al.*, 1999). Temporal variations of aquatic assemblages in lakes are also likely to be governed by interactions between species such as competition, parasitism and predation (Jackson *et al.*, 2001; Tonn *et al.*, 2004). Food webs are likely to be less complex in lakes than floodplain river systems (Table 1.1). Power *et al.* (1995) found that food webs were more complex within temporally fluctuating environments rather than environments with stabilised water levels.

Despite being physically located on the floodplain, storages are most likely acting more like true lakes than natural floodplain wetlands. However, because of their position and their hydrological regimes, they will have aquatic assemblages that are accustomed to life in either the river channel itself or in natural floodplain wetlands. This biota is likely to have a variety of feeding and life history adaptations for surviving and exploiting highly variable flow regimes associated with these natural habitats and their high spatio-temporal heterogeneity (Junk *et al.*, 1989). Storages are closed systems in that, once water has entered the storage, via either extraction from the river channel or overland flow, fully aquatic taxa and life stages within the storage have no direct access to the floodplain. Only mobile taxa and life stages will be able to move between storages and natural wetlands and the exchange of nutrients and organic matter between storages and the river channel will be limited (Amoros and Bornette, 2002). It is therefore likely that storages will have less diverse food webs than the counterparts in neighbouring natural wetlands (Power *et al.*, 1995).

1.4 Conceptual model comparing storages and natural wetlands

In exploring the role of storages as viable aquatic habitat, an important step was to conceptualise how storages differ from their natural floodplain wetland counterparts (Fig. 1.1). This hypothetical model depicts potential drivers that may be affecting the biodiversity of storages compared with natural wetlands and sets the framework for the current study.

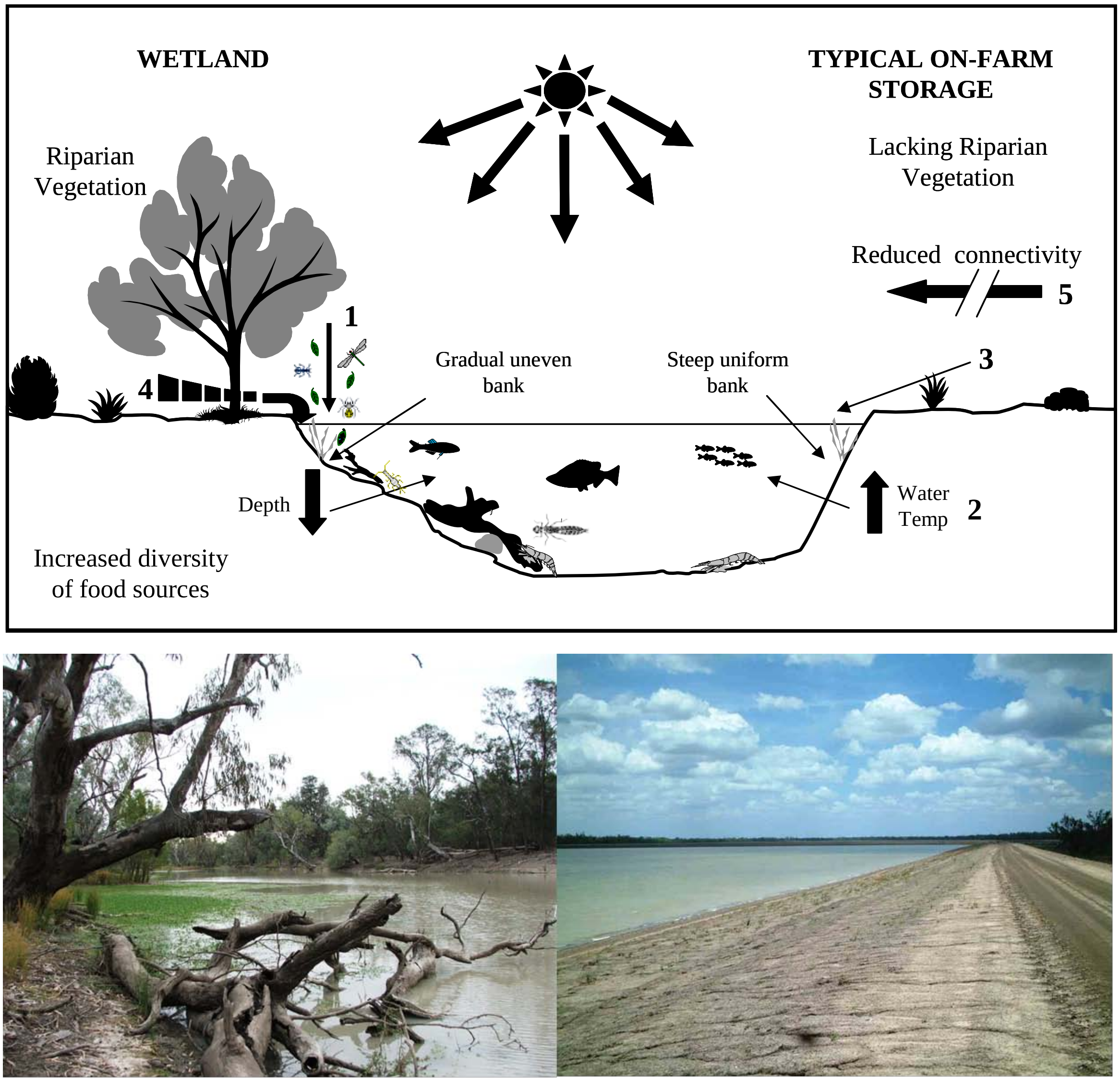


Fig. 1.1: Conceptual model comparing potential drivers affecting biodiversity between natural wetlands and on-farm water storages. 1. Inputs of leaf litter and terrestrial invertebrates. 2. Rise in water temperature due to lack of shading. 3. No source of logs and branches. 4. Filtration of sediments and nutrients (adapted from Bunn *et al.*, 1999). 5) Levee banks that surround the majority of irrigation farms form a barrier between storages and the floodplain. Photos: Susan Lutton.

1.4.1 Morphology

In wetlands and rivers, surface area, volume, perimeter, channel complexity and bank slope are thought to affect the overall biodiversity of the ecosystem (Olden *et al.*, 2001; Bunn and Arthington, 2002). Thoms *et al.* (2006) suggested that increased channel complexity of dryland rivers increased the surface area available for organic matter accumulation, a potential food source and habitat for lower order aquatic organisms. It is well known that the biomass of both fish and invertebrates in lakes is concentrated around the edge, or littoral zone (Keast and Harker, 1977), with the complexity of the shoreline influencing diversity (Jackson and Harvey, 1993).

Surface area, volume and shoreline perimeter of lakes have been positively correlated with habitat diversity which in turn affects species diversity (Jackson *et al.*, 2001). Increases in the diversity of zooplankton (Dodson *et al.*, 2000), fish (Browne, 1981) and molluscs (Browne, 1981) have been reported as lake surface area increases. In lakes on the Orinoco floodplain, variation in fish communities appears to be determined by the optical environment which in turn is strongly related to lake shape (Lewis *et al.*, 2000).

Benthic algae have been found to be extremely important contributors to waterhole (Bunn *et al.*, 2003) and lake (Hecky and Hesslein, 1995) food webs. In dryland river waterholes, as found in the Border Rivers, the water is very turbid restricting photosynthesis of benthic algae to a narrow photic zone around the bank. The steeper the slope of the bank, the smaller the width of this photic zone (Fellows *et al.*, 2009). Depth is also likely to affect productivity with shallow lakes usually being more productive (Hamilton and Lewis, 1990; Perin *et al.*, 1996).

It is likely that differences in morphology between natural wetlands and artificial storages, such as bank slope and depth, will in turn lead to differences in biodiversity and production.

1.4.2 Connectivity

The theory of island biogeography attempts to establish and explain the factors that affect the species richness of natural communities (MacArthur and Wilson, 1967). Although the theory was developed to explain the diversity of actual islands it has since been extended to include any ecosystem that is surrounded by unlike ecosystems, such as mountain tops, game preserves and lakes; or in this case, floodplain waterbodies. It is proposed that the number of species found on an undisturbed island is determined by immigration, emigration and extinction (MacArthur and Wilson, 1967). During a flood event, floodplain wetlands are filled from overland flow, connecting channels that link them to the main river channel or groundwater. At this time, aquatic biota can move freely between the river and wetlands via the floodplain (CEC, 2000; Arthington *et al.*, 2005; Balcombe *et al.*, 2007). In contrast, storages are separated from the floodplain by an embankment around the storage itself and by levees that are built around the majority of irrigation farms to protect crops from floodwater and prevent the release of contaminated water into the river system (see Section 2.7.3). Therefore, water is pumped from the river channel either directly into the storage or via a supply channel. Any aquatic fauna in the extracted water will pass through at least one pump before entering the storage. Due to their size, it is more likely that fish will be injured or killed as they go through the pump. These physical barriers and reduced connectivity of storages to the river and floodplain will influence colonisation and emigration and immigration from and into these waterbodies. In turn, these factors are likely to reduce the species richness of artificial storages.

1.4.3 Source of water

Unlike natural wetlands, storages have access to tailwater. When fields are irrigated, any excess water is collected in the tail drain and known as tailwater. In many cases, tailwater is then recycled back to the storage dam where it can be used for further irrigations. As tailwater is excess irrigation water, which has flowed along the crop rows, it is likely that it will contain chemical residues from recent spraying, either in solution or adsorbed onto soil particles (Crossan, 2002; Rose, 2006). Potential contaminants include herbicides, insecticides, defoliators, wetting agents, conditioners and fertilisers. Some of these chemicals are likely to reduce algal diversity and biomass in storages (Johnson, 1986; Okamura *et al.*, 2002) which are an important base of food webs in dryland river catchments (Hamilton *et al.*, 1992; Bunn *et al.*, 2003; Delong and Thorp, 2006).

1.4.4 Riparian vegetation

The majority of storages are intrinsically different from natural wetlands due to their constructed nature and therefore lack many of the critical attributes that drive the diversity of native aquatic taxa, such as coarse woody debris, riparian vegetation and macrophytes. Natural wetlands have a multilayered band of vegetation along their edge (Lovett *et al.*, 2003), whereas the majority of storages are devoid of riparian vegetation. The importance of linkages between the riparian zone and the aquatic ecosystem has been well documented (Bunn, 1993; Schulze and Walker, 1997; Pusey and Arthington, 2003). Riparian vegetation adds organic carbon to the aquatic food web in the form of litter (Schulze and Walker, 1997; Bunn *et al.*, 1998). Terrestrial invertebrates falling from riparian vegetation also add energy and nutrients to the aquatic food web (Bunn *et al.*, 1998). Lack of riparian vegetation on constructed storages is likely to lead to a reduction in the amount of energy and nutrients available to the aquatic food web therein. The general absence of trees also means that there is no source of coarse or large woody debris which provide important habitat for fish, plants and macroinvertebrates (Bunn *et al.*, 1998; Sheldon and Walker, 1998).

Riparian vegetation also provides shading during the day, moderating fluctuations in water temperature in the littoral zone. Where riparian vegetation is absent, water temperatures are likely to rise affecting the assemblage composition of algal and aquatic plant communities and their contribution to energy production (Arthington, 1995). The reduction in shade and increase in water temperature may drive a shift in aquatic production from dominance by microalgae, such as diatoms, to larger, less palatable species, such as green algae and macrophytes (Bunn *et al.*, 1999).

1.4.5 Habitat availability and substrate type

The amount and type of habitat available is often considered a primary limiting factor in population and community recovery in degraded ecosystems (Bond and Lake, 2003). Availability of habitat is believed to be important to fish and macroinvertebrate communities for a number of reasons. Habitat can provide shelter from the elements and refuge from predators (Everett and Ruiz, 1993; Fischer and Eckmann, 1997; Olden and Jackson, 2001; Lewin *et al.*, 2004) while providing suitable spawning sites for certain species (Rowland, 1983; Rust *et al.*, 2002) and in some cases can also be a source of food (Grenouillet and Pont, 2001). Species abundance and diversity have both shown positive correlations with the abundance of coarse woody debris (CWD) present at a site (Benke *et al.*, 1984; Newbrey *et al.*, 2005; Smokorowski *et al.*, 2006; Helmus and Sass, 2008), with CWD providing a refuge from predation for epibenthic or juvenile fish and invertebrates (Everett and Ruiz, 1993; Schindler and Scheuerell, 2002; Sass *et al.*, 2006). Fish abundance can increase with the addition of CWD to streams (House and Boehne, 1985) and lakes (Newbrey *et al.*, 2005). Likewise the complexity of habitat available is positively correlated with macroinvertebrate richness, with the addition of grooves to artificial snags increasing macroinvertebrate diversity (O'Connor, 1991). In lakes, ponds and ditches, which are more comparable to storages, habitat preference of macroinvertebrates has been found to be influenced chiefly by vegetation, where there was a definite relation between vegetation and the distribution of macroinvertebrates (Korinkova, 1971; Dvořaki and Bestz, 1982; Scheffer *et al.*, 1984).

Substrate diversity may also affect food availability in aquatic habitats. Even when there is little stony substrate present, hard substrate such as woody debris can be the site of the majority of biotic production (Benke *et al.*, 1984). In dryland river waterholes, double the number of diatom species were found on hard substrates (woody debris) compared with soft substrates (sediments) (McGregor *et al.*, 2006). The absence of woody debris and rocks is likely to affect the abundance of diatoms which could reduce the algal biomass available to aquatic food webs in storages.

1.4.6 Diversity of food sources

As outlined in the previous sections diversity of food sources may be limited in storages as a result of reduced amount of riparian vegetation, lack of hard substrate, steep sloping banks and presence of chemicals. Biodiversity is known to both increase and decrease with productivity and to be unimodally related, such that peak biodiversity occurs at intermediate productivity (Waide *et al.*, 1999; Kassen *et al.*, 2000). More recently, multivariate models predicted that both disturbance and productivity together interact to effect biodiversity (Kondoh, 2000; Worm *et al.*, 2002).

Aquatic macrophytes have been used as a strategy to increase biodiversity in reservoirs (Thomaz *et al.*, 2006). Two main factors are generally used to explain higher biodiversity in vegetated areas, namely food availability and shelter (Rozas and Odum, 1988). High prey densities are typically associated with macrophytes (Patterson, 1993; Grenouillet and Pont, 2001). However, different macrophyte types can offer contrasting morphological characteristics (e.g. size, number and orientation of leaves and stems) (Chick and McIvor, 1994) and plant architecture is likely to influence both invertebrate and fish distributions (Dionne and Folt, 1991). Although debate continues (Worm *et al.*, 2002), lack of resource diversity at storages may reduce animal species richness at these sites (Novotny *et al.*, 2006).

1.5 Research hypotheses

This thesis explores the ecological value of storages on irrigation farms within the Border Rivers Catchment. From the conceptual model presented in Fig. 1.1 it is hypothesised that:

1. Natural floodplain wetlands in the Border Rivers Catchment will have more complex morphology than storages.
2. The aquatic communities of storages will be less diverse and abundant than those found in neighbouring natural floodplain wetlands as a result of less complex morphology and reduced connectivity.
3. Given the adverse effect of chemicals on aquatic communities, the storages with more exposure to chemical contaminants are hypothesised to support a less diverse animal community than other storages.
4. Due to the limited access to energy sources, the food web of storages will be more dependent on autochthonous food resources than allochthonous food resources.

1.6 Thesis outline

This introductory chapter has reviewed past research on floodplain wetlands and sought to predict how the assemblages and processes of storages might compare with natural wetlands. The conceptual model and hypotheses were first presented at the 2005 9th Annual Environmental Research Event (Lutton, 2005).

In Chapter 2 of the thesis, physical, hydrological and climatic characteristics of the Border Rivers Catchment are described, together with an outline of the irrigation industry in the catchment and a description of a typical irrigation farm. Chapter 3 outlines the morphological and hydrological features of storages in the catchment and how these compare with floodplain wetlands. Results are used to classify storages and select study sites for the remaining investigations in this thesis. This chapter forms the basis of the following journal manuscript:

Lutton, S.J., Sheldon F. and Bunn, S.E. (In press) On-farm water storages – do they offer a viable alternative aquatic habitat to natural wetlands? *Aquatic Conservation: Marine and Freshwater Ecosystems*.

Chapter 4 and Chapter 5 describe spatial and temporal variations in aquatic assemblages of storages and floodplain wetlands. This work, conducted in the spring of 2005 and over the course of the 2005-2006 and 2006-2007 summers, also investigated the effects of different management options applied to storages on the aquatic assemblages therein. Using the results from an egg bank mesocosm experiment and sampling of extracted river water, Chapter 6 discusses the likely sources of biota found in storages in the Border Rivers Catchment. Chapter 7 presents analyses of the food web structure of five storage sites and five floodplain wetland sites using stable isotope techniques. Chapter 8 summarises the major findings from the previous chapters and discusses them with relation to the long term management of storages and how their value as alternative aquatic habitat within the Border Rivers Catchment could be improved.

Chapter 2 Study area

2.1 Location

Drylands occur where climatic, topographic or oceanographic factors prevent moisture-bearing weather systems reaching that zone (North, 2003). These arid and semi-arid regions cover over one third of the global land area including parts of Australia, southern Africa, the Americas, China, India and the Mediterranean (Walker *et al.*, 1995; North, 2003). Rivers draining these areas are characterised by their highly variable and unpredictable flow regimes (Walker *et al.*, 1995; Puckridge *et al.*, 1998) causing them to fluctuate between being highly fragmented and highly connected (Sheldon *et al.*, 2002). This temporal variation in conditions creates systems with high diversity of biota and ecological processes (Kingsford *et al.*, 1998). Large Australian dryland rivers, including the Cooper and Diamantina, might be considered the most variable in the world (Puckridge *et al.*, 1998; Bunn *et al.*, 2006b), existing for much of the time as a string of disconnected waterholes (Bunn *et al.*, 2003). However, when they do flow, they can occupy vast floodplains, often for months at a time (Bunn *et al.*, 2006a), refilling numerous, temporary and permanent, floodplain wetlands (Tronstad *et al.*, 2005).

The Border Rivers Catchment is one such dryland area in Australia (Fig. 2.1). Occupying an area of approximately 49,470 km^2, this catchment spans the border between north west New South Wales (NSW) and southern Queensland (QLD), Australia, encompassing roughly equal areas of the two states (DLWC, 1999). The Great Dividing Range forms the eastern boundary of the catchment with elevations of up to 1350 m above sea level (ASL), dropping to approximately 150 m ASL at Mungindi, the catchment's most western town (Boddy and Bales, 1996). The Border Rivers Catchment is approximately 200 km from north to south and 450 km from east to west at its widest points (BRFF, 2002).

The major rivers in the catchment are the Macintyre and Severn Rivers in the south-east, the Dumaresq in the east, the Weir River in the north-west and Macintyre Brook in the north (McCosker, 1996). The Macintyre officially becomes the Barwon River

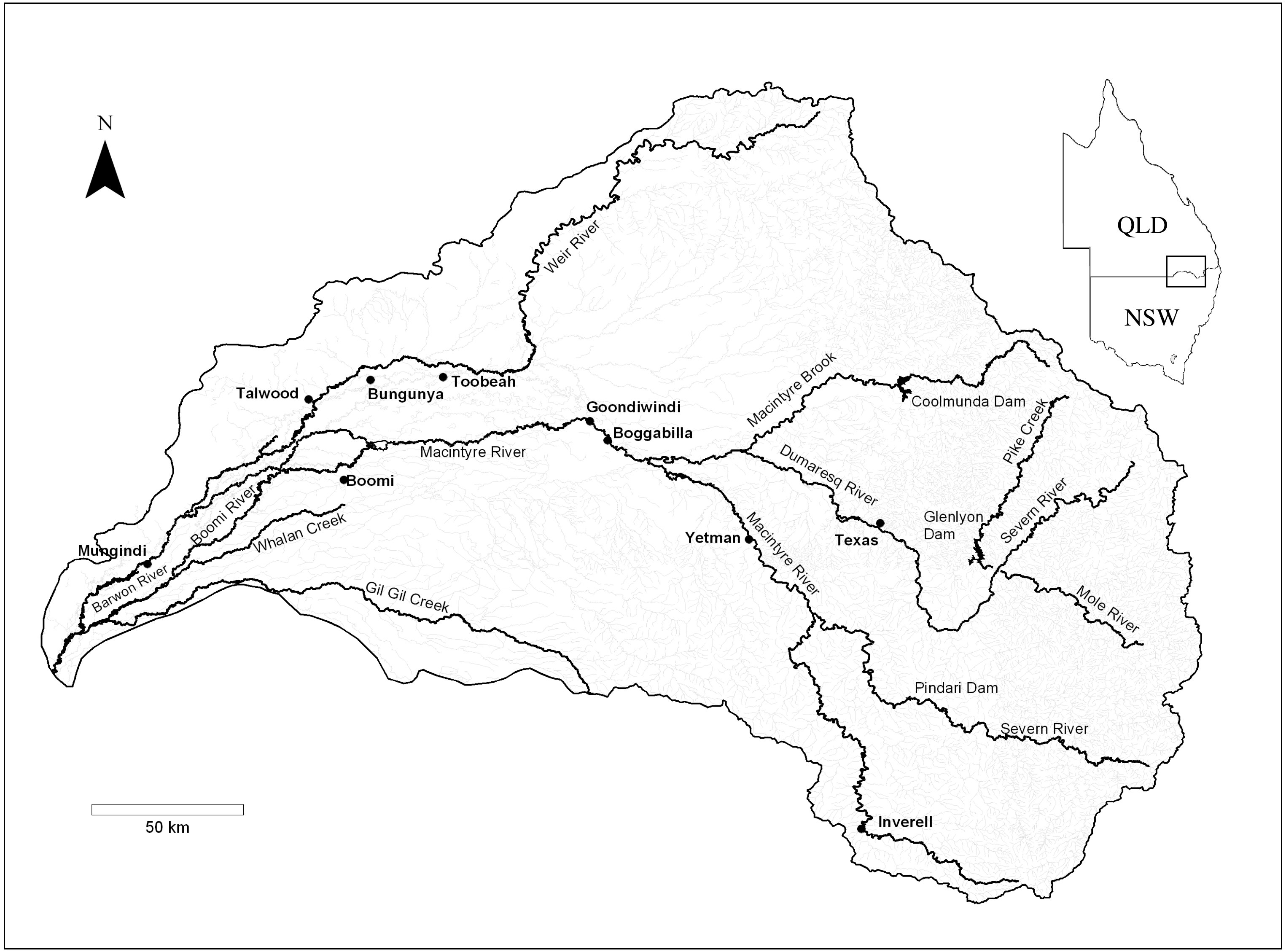


Fig. 2.1: Map showing the location of the Border Rivers Catchment within Queensland and New South Wales (inset).

23 km upstream of Mungindi at the Weir River junction (DWR, 1995) though many growers in the Talwood area refer to it as the Barwon River. Other important tributaries include another Severn River, Pike Creek, the Beardy River and the Mole River (McCosker, 1996).

There are many significant wetlands in the Border Rivers, particularly downstream of Boggabilla. A number of effluent streams diverge from the Macintyre River near Boggabilla and Goondiwindi (DWR, 1995) to form some of the streams that fill these waterholes and floodplain lagoons (Medeiros, 2004). The wetlands include lagoons or waterholes in flood channels (DWR, 1995) and numerous anabranch channels and associated billabongs along the Macintyre River channel itself (Southwell, 2002).

2.2 Climate

Summer rainfall patterns dominate this area of Australia, with 55% of the annual rainfall occurring from November to March (DWR, 1995). Annual median rainfalls range from greater than 800 mm at Tenterfield in the east, to 500 mm in the extreme west of the catchment near Mungindi (DWR, 1995). However, interannual variability is very high, for example, records from Boggabilla show that annual rainfall has ranged from a low of 174 mm in 1902 to a high of 1,041 mm in 1950 (DWR, 1995).

In contrast to rainfall, temperatures increase from east to west across the catchment. On the Western Plains, average daily maximum and minimum temperatures in January are approximately 35 ^{o}C and 20 ^{o}C respectively and in July these drop to 20 ^{o}C and 5 ^{o}C. At higher elevations in the east, temperatures are 27 ^{o}C and 14 ^{o}C in January and 13 ^{o}C and 1 ^{o}C in July (DWR, 1995). During summer, evaporation exceeds average rainfall throughout the catchment, ranging from 1200 mm a year in the east to 1750 mm in the west (DWR, 1995).

2.3 Geomorphology and vegetation

Based on landforms, geomorphology, soils and vegetation, the Border Rivers Catchment can be divided into three distinct zones; the upland zone, billabong zone and riverine plains zone (Ring *et al.*, 1984; McCosker, 1996).

The upland zone on the Tablelands east of Texas is characterised by hilly to rugged landscapes of granite and basalt rocks (McCosker, 1996). The terrain is mostly steep and rivers flow quickly through well defined valleys and riverbeds (Ring *et al.*, 1984). Water is generally confined within the banks of the streams, which tend to rise rapidly after rain but fall just as quickly afterwards. Widespread clearing in this area of the catchment has left only remnants of the original eucalypt woodland (Boey *et al.*, 1997). However, sections of the steep hill country still support a rich diversity of plant species, including mountain red gum (*Eucalyptus pulverulenta*), silver leafed ironbark (*E. paniculata*), white and grey box (*E. albens* and *E. microcarpa*) with an understorey of tea trees (*Leptospermum brachyandrum*), wattles (*Acacia* sp.) and mountain oaks (*Allocasuarina torulosa*) (DWR, 1995). A number of vulnerable plant species have been recorded in this area of the catchment, including the shrub, *Allocasuarina rupicola*, the small tree, *Callistemon pungens* and the grass, *Dichelachne parva* (McCosker, 1996).

The billabong zone runs from Texas to approximately 20 km downstream of Goondiwindi. Here the slope of the land decreases and floodplains become much more extensive. This zone has undulating topography and the river is bordered by numerous intermittent and semi-permanent billabongs (DWR, 1990; McCosker, 1996). Intensive grazing and clearing of trees has considerably altered the composition of the understorey in this area of the catchment. However, tracts of original mountain red gum, silver leafed iron bark and smooth barked apple (*Angophora costata*) woodland are still intact (Boey *et al.*, 1997). The billabongs in this zone support a range of submerged, floating and emergent macrophytes, including pondweeds (*Potamogeton* spp.), water primrose (*Ludwigia peploides*), water couch (*Paspalum distichum*), spiny mudgrass (*Pseudoraphis spinescens*) and tall spike rush (*Eleocharis sphacelata*) (McCosker, 1999). The rare spike rush (*Eleocharis blakeana*) has been found in this zone along with the giant waterlily (*Nymphaea gigantea*) (Sainty and Jacobs, 1993).

The riverine plains zone begins 20 km west of Goondiwindi and extends to the end of the basin. This zone consists almost entirely of recent alluvia on fairly flat topography. For example, from Boggabilla to Mungindi, some 300 km downstream, the river falls by only 60 m (DWR, 1995). In this area, flows of medium size spread out over large areas of the floodplain and several effluent streams and anabranches run water away from the Macintyre River (DWR, 1995). Virtually no semi-permanent water bodies exist away from the main river channels in this zone (Ring *et al.*, 1984) and many of the upper and mid-catchment plant species give way to drier, floodplain species, such as river red gums (*Eucalyptus camaldulensis*) and coolibah trees (*E. coolabah*) (Boey *et al.*, 1997). River red gums dominate the fringing riparian vegetation along active channels while coolibahs become dominant on the floodplain and banks of ephemeral anabranches (McCosker, 1999). The floodplain is home to several flood dependent species with a canopy of coolibahs, an understorey of river cooba (*Acacia stenophylla*) and a third stratum of lignum (*Muehlenbeckia florulenta*). Ground cover is dominated by Warrego summer-grass (*Paspalidium jubiflorum*). The composition of plant species has been significantly affected by irrigation and pastoral activities (Boey *et al.*, 1997) and Boddy & Bales (1996) reported a reduction in aquatic macrophytes in this zone.

In many areas, introduced species now predominate (Boey *et al.*, 1997) including weeping willows (*Salix babylonica*), which have spread extensively throughout the catchment (McCosker, 1996), and lippia (*Phyla canescens*), which is present on the banks of the Macintyre and Dumaresq Rivers (McCosker, 1999).

Storages and floodplain wetlands selected for this study were located within the billabong and riverine plains zones (Fig. 2.2).

2.4 Hydrology

Encompassing the headwaters of the Murray-Darling Basin, the Border Rivers Catchment has a mean annual inflow of approximately 1,200,000 megalitres (ML) (BRFF, 2002); approximately half originating in the NSW portion of the catchment (Boddy and Bales, 1996). Under natural conditions, mean annual end of system flow, after channel losses and natural processes, would be 580,000 ML (derived from natural flows at Mungindi through the daily flow model IQQM: (BRFF, 2002)). The remainder would water the floodplain and stream banks, fill wetlands, recharge groundwater and would also be transpired by in-stream and on-bank vegetation and evaporate.

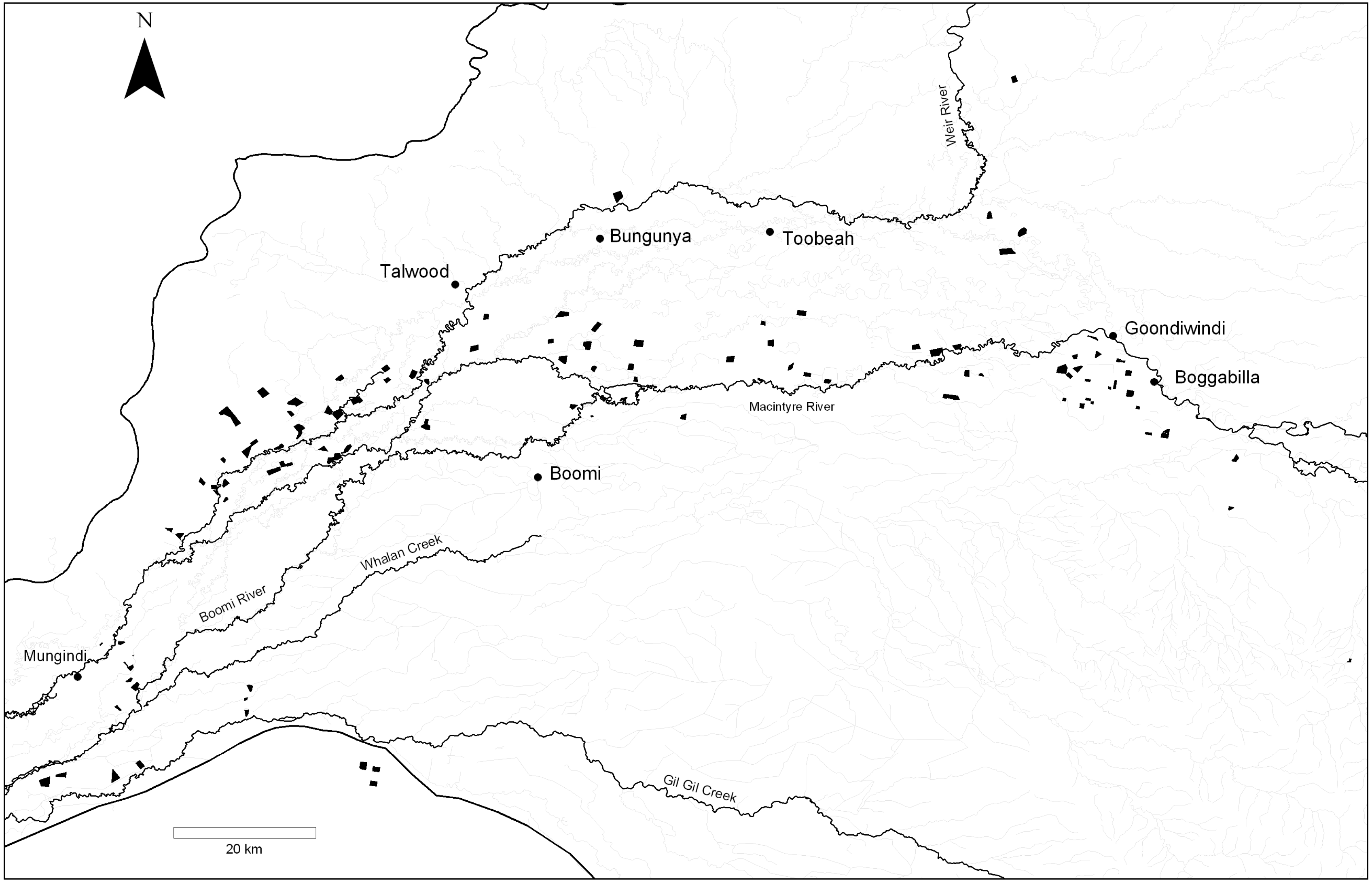


Fig. 2.2: Map showing the location of the storages (back objects) studied within the Border Rivers Catchment.

Natural flows tend to be higher between the months of October and February. The majority of the runoff is generated in upland areas to the east where rainfall is highest. At Boggabilla, average flow is about 910,000 ML per year which is only around 7% of the average rainfall for the catchment (DWR, 1995). However, under natural conditions, actual flows rarely match the average. Two significant features of the natural flow regime of the Border Rivers Catchment are its inability to sustain flow during periods of drought and its highly variable discharge, between 1% and 500% of the yearly average (Keyte, 1993) (Fig. 2.3). Prior to construction of Glenlyon dam, flow at Boggabilla completely ceased in an average of one year in four (DWR, 1995). The coefficient of variation of monthly flows from hydrographs of four sites in the Border Rivers Catchment ranges from 2.06 to 3.42 (Table 2.1).

It was considered that such inter-annual variation in flow would not sustain the present water requirements of domestic and industrial consumption; much less large scale irrigated agricultural production. Therefore, construction of storage facilities was essential for maintaining continuity of water supplies.

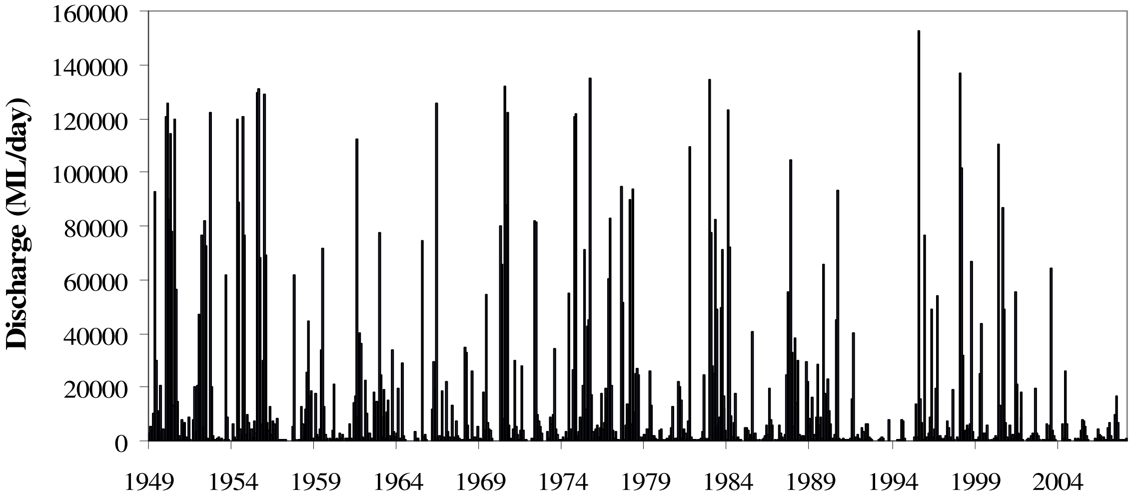


Fig. 2.3: Hydrograph of maximum daily discharge per month for the Macintyre River recorded from 1949 to 2008 at the Goondiwindi gauging station (416201A)
(source: http://www.nrw.qld.gov.au/watershed/precomp/416201a/416201a.htm).

Table 2.1: Summary statistics of monthly flow records (ML) for four gauging stations in the Border Rivers Catchment (source: http://www.nrw.qld.gov.au/watershed).

	Macintyre River at Goondiwindi	**Macintyre River at Goondiwindi Weir**	**Weir River at Talwood**	**Callandoon Ck at Carana Weir**
Gauging Station	416201A	416201B	416202A	416203A
Period of Record	1916-2006	1916-2006	1948-2006	1995-2006
Number of Values	741	741	530	126
Mean	82156	81209	12227	4613
Standard Deviation	169144	167102	41829	13310
Coeff. of Variation	2.06	2.06	3.42	2.89
Coeff. of Skewness	4.55	4.61	5.52	5.04
Serial Correlation	0.363	0.361	0.286	0.37

Pindari Dam is the only major dam in the NSW portion of the catchment and is situated on the Severn River. Its capacity was enlarged from 37,000 ML to 312,000 ML in 1995 and 100% of the water is allocated to NSW licence holders (BRFF, 2007). Glenlyon Dam on Pike Creek has a capacity of 254,000 ML and the water is shared between QLD and NSW in the ratio of 43%:57%, while Coolmunda Dam, on the Macintyre Brook, has a capacity of 74,000 ML and 100% of the water is allocated to QLD licence holders (BRFF, 2007). In addition to the three main dams, there is also a major re-regulating weir at Boggabilla, built in 1991, with a capacity of 5,850 ML (DWR, 1995). With the construction of dams, downstream flows in the river systems can be regulated by flow releases to increase the reliability of supply. The regulated rivers in the Border Rivers Catchment are the Severn, Macintyre and Barwon Rivers below Glenlyon and Pindari dams and as far downstream as the town of Mungindi; Macintyre Brook below Coolmunda dam and the Dumaresq which is regulated by Glenlyon and Coolmunda dams (DNR and DLWC, 2000). There are also numerous smaller weirs along the system built by governments and landholders to provide a more permanent water supply for urban, stock, domestic and in some cases irrigation purposes. These smaller weirs have little or no regulatory function (BRFF, 2002).

Dams provide regulated flows for irrigation, households, stock, town water supplies and industrial purposes throughout the catchment, capturing water during floods for later release downstream. This regulation affects the flow regime of large stretches of river within the catchment, reducing downstream flow variability and almost eliminating periods of no flow (McCosker, 1996) (Plate 2.1 and 2.2).

Plate 2.1: Macintyre River looking west from the old bridge at Goondiwindi during the 2002 drought, after construction of the Goondiwindi and Boggabilla weirs. Photo: Glenn Wilson.

Plate 2.2: Macintyre River looking west from the old bridge at Goondiwindi during the 1940 drought before construction of the Goondiwindi and Boggabilla weirs. Photo: John Elsley.

Within the Macintyre irrigation region, the total volume of stream flow has been reduced due to water extraction for the irrigation of cotton and other crops. On average, about 40% of the total Border Rivers inflows are extracted (GHD, 1992), with irrigation using 95% of total diversions (MDBC, 2002). However, with the extreme variability of inflows from year to year, this percentage figure will change dramatically depending on whether it is a dry or a wet year.

2.5 Land use

Despite the growing irrigation industry, dryland farming is still the main land use in the Border Rivers Catchment (Keyte, 1993). Of the area developed for farming, approximately 10,000 km^2 is used for growing crops and 54,000 km^2 is forestry and grazing land (Wylie and Greggery, 1995) supporting around 85,000 head of cattle and 5,500,000 sheep (McCosker, 1996). Over the last 40 years, a large area of the catchment has been sown to improved pastures, now covering an area of 3,300 km^2 (McCosker, 1996).

Intensive animal industries include cattle feedlots and piggeries. There are nine cattle feedlots throughout the catchment with a total operating capacity of 140,000 head and 20 properties with more than 500 pigs (Wylie, 1995). Cereal cropping commenced in the area after World War II and now around 5,000 km^2 are sown annually to wheat and barley and 560 km^2 to grain sorghum (Wylie, 1995). Established in the early 1900's, the horticultural industry is flourishing on 45 km^2 in the uplands zone near Stanthorpe (Wylie, 1995).

In the upland zone, irrigation is restricted to small areas of tobacco, lucerne, wheat, sorghum, improved pastures and fodder crops next to the river and on the alluvial flats (DBBRC, 1988). Large scale irrigation occurs below the junction of the Dumaresq and Macintyre Rivers (DBBRC, 1988) growing cotton, sorghum and wheat.

Alluvial mining for tin and sapphires has occurred in the Tenterfield-Emmaville and Inverell-Glen Innes areas respectively (Boey *et al.*, 1997). Tin, silver, copper and lead have also been mined in the catchment (Keyte, 1993).

2.6 Aquatic communities in the Border Rivers Catchment

2.6.1 Fish

The Border Rivers Catchment is home to 14 species of native fish (Moffatt and Voller, 2002) (Table 2.2) including the popular recreational angling species golden perch (*Macquaria ambigua*), silver perch (*Bidyanus bidyanus*), Murray cod (*Maccullochella peelii*) and the eel tailed catfish (*Tandanus tandanus*) (Swales and Curran, 1995). Populations of golden perch, silver perch and Murray cod have suffered significant declines in the last 100 years (Farragher and Harris, 1993) which may be due to over fishing (Swales and Curran, 1995). Bony bream (*Nematolosa erebi*) is abundant in the billabong and riverine plains zones forming huge populations (Briggs and McDowall, 1996). It is found in turbid waters and can tolerate a wide range of water quality and habitat conditions (Briggs and McDowall, 1996). Spangled perch (*Leiopotherapon unicolor*) is also found predominantly in the billabong and riverine plains zones, potentially inhabiting any body of water within these areas (Moffatt and Voller, 2002). Purple spotted gudgeon (*Mogurnda adspersa*) and olive perchlet (*Ambassis agassizii*) are less common in the catchment and are considered rare by Wager and Jackson (1993).

Changed flow regimes in the catchment have aided the establishment of three exotic fish species (Table 2.2); European carp (*Cyprinus carpio*), goldfish (*Carassius auratus*) and mosquito fish (*Gambusia holbrooki*) (BRICC, 2000). In the Barwon and Macintyre Rivers, *C. carpio* are the main exotic species, comprising between 20 to 30% of the total fish population (DPI, 2003). The spread of all three exotic species has been enhanced by their ability to withstand a wide range of environmental conditions, including high temperatures and low oxygen levels allowing them to inhabit areas not tolerated by native species (McCosker, 1996).

Table 2.2: Fish species found in the Border Rivers Catchment.

Species	Common Name
Native fish	
Nematolosa erebi	bony bream
Galaxis olidus	mountain galaxias
Retropinna semoni	Australian smelt
Tandanus tandanus	eel-tailed catfish
Craterocephalus stercusmuscarum fulvus	fly-specked hardyhead
Melanotaenia fluviatilis	crimson spotted rainbowfish
Ambassis agassizii	olive perchlet
Macquaria ambigua	yellowbelly or golden perch
Maccullochella peelii	Murray cod
Bidyanus bidyanus	silver perch or black bream
Leiopotherapon unicolor	spangled perch
Gadopsis marmoratus	river blackfish
Mogurnda adspersa	purple spotted gudgeon
Hypseleotris spp.	carp gudgeons
Exotic fish	
Cyprinus carpio	European carp
Carassius auratus	goldfish
Gambusia holbrooki	gambusia or mosquito fish

2.6.2 Macroinvertebrates

Boddy & Bales (1996) reported a total of 270 different macroinvertebrate taxa throughout the Border Rivers from sampling over three years between 1993 and 1995. They believed that this was a conservative number, as all specimens could not be identified to species level. The orders Coleoptera, Diptera, Hemiptera, Odonata and the phylum Mollusca accounted for over 85% of the taxa collected (Boddy & Bales, 1996). They found that the macroinvertebrate community in the Border Rivers was similar to or slightly more diverse than other rivers in northern NSW and that there was considerable spatial and temporal variation in average number of taxa and individuals. Upstream of the Macintyre-Dumaresq confluence, in the Mole, Dumaresq and Macintyre Rivers and Macintyre Brook, most sites had a high diversity and abundance of macroinvertebrates (Boddy & Bales, 1996). Number of taxa and individuals declined at sites downstream of the confluence, which were dominated by Diptera (Boddy & Bales, 1996). Choy & Thomson (1999) recorded the highest number of taxa in the upper unregulated reaches of the Severn River. Macroinvertebrate communities were found to improve with good flow conditions (Boddy and Bales, 1996; Choy and Thomson, 1999).

2.7 Irrigation industry

2.7.1 History of cotton in Australia

More than 100 countries worldwide grow cotton (ICAC, 2007), including the USA, China and Australia. Cotton is a summer crop preferring hot summers with low humidity and maximum amount of sunshine (CA, 2008). Almost all parts of the cotton plant are used in some way; fibre for clothing, linters for bandages and x-rays, cotton seed can be pressed for oil or fed whole to livestock and poultry, the stalk is used in ethanol production or as mulch and crop stubble is left on the field to help prevent erosion and return nutrients to the soil (CA, 2008). Along with Egypt, Australia produces the best quality cotton in the world (CA, 2008).

Dryland cotton growing in Australia dates back to the early 1800’s with small quantities grown in QLD as early as 1857. However, irrigated production did not take off until the 1960’s with the completion of Keepit Dam on the Namoi River in NSW and Burrendong Dam in the Macquarie Valley (CA, 2008). Today, two thirds of Australia’s cotton is grown in NSW and the remainder in QLD (CA, 2008). In 2005/06, 84% of the crop was grown under irrigation (ACIC, 2008). Irrigated cotton is generally planted in September or October and picking can begin as early as March. Depending on the season, the initial irrigation at planting is usually followed by four to five further irrigations from mid December to late February. In 2004-05 cotton used 18% of the irrigation water in Australia while the majority was used for irrigation of grazing pastures (28.7%) (Trewin, 2006). On average 6.7 ML of water is needed per hectare of cotton grown compared to rice at 12.3 ML per hectare and sugar cane at 5.5 ML per hectare (Trewin, 2006).

Australia is a relatively small producer of cotton, yielding 1.3 million bales from 1,420 km^2 in 2006/07, compared with 31 million bales from China and 20 million bales from the USA (ICAC, 2007). However, the cotton industry contributes on average $1.5 billion per year to the national economy (CA, 2008) and is one of Australia’s highest rural export earners (ABARE, 2006). Combined with this, the production of cotton creates numerous jobs on the farm itself, as well as for distributors of farm machinery, farm consultants, itinerant workers, picking contractors and ginning staff. It is estimated that the Australian cotton industry employs 10,000 individuals (ACIC, 2008). Despite its high economic value, irrigated agriculture accounts for only 0.5% of Australia's farming area (CA, 2008). However, the position of irrigation farms on the floodplain will compound any effect they have on aquatic systems and the organisms within. In the Border Rivers/Gwydir/St George catchment, for example, the percentage of irrigation farms with stream or river frontage was 91.2% (Doyle and Coleman, 2006).

2.7.2 History of irrigation in the Border Rivers Catchment

Irrigated farming in the Border Rivers Catchment actually began in the early 1870's. However, prior to 1976, irrigation was largely confined to the upper parts of the catchment, with fruit and vegetables grown on the Granite Belt and tobacco as the major crop elsewhere (BRFF, 2002). Compared to today's standards, the volume of water used was very small.

Cotton growing in the Border Rivers began soon after the first settlement in NSW but was not a prominent land use until the 1960's. After completion of Glenlyon Dam in 1976, both QLD and NSW governments encouraged landholders to take up licences and put them to productive use (BRFF, 2002). The first broadacre flood irrigation of cotton was developed between Boggabilla and Goondiwindi on the NSW side of the river in the late 1970's (BRFF, 2002). Since then development has increased significantly with 600 km^2 of land in the catchment developed for broad acre irrigation (BRFF, 2007). As well as cotton, other irrigated crops include wheat, peanuts, corn, chickpeas and sorghum. Of the irrigated crops grown in 1993/94, 85% of these were cotton crops (DBBRC, 1994). In a good year an estimated 3000 jobs are generated by the irrigation industry in the Border Rivers Catchment producing a cotton crop worth $350 million (BRFF, 2007). The majority of irrigation in the catchment is found in the billabong and riverine plains zones, extending along the Macintyre and Barwon River systems from Boggabilla to below Mungindi (Boey *et al.*, 1997; Davies *et al.*, 2008).

Maintaining irrigated agriculture in a semi-arid environment with variable rainfall, as observed in the Border Rivers Catchment, is associated with significant issues of water security (Deng *et al.*, 2006). Therefore, in many irrigation regions of Australia on-farm water storages have been built. These storages are large, raised earth structures, designed to allow water to be collected during high flow events, stored and used at a later time when flows and rainfall are low. The majority of on-farm storage occurs on the floodplain west of Boggabilla (Fig. 2.2). Wylie (1995) estimated that in 1995 the storage capacity of private storages across the catchment was 247,000 ML. By 2008, this had grown to 510,000 ML (T. Napier, pers. comm., 2008), or 42.5% of the average annual inflow.

2.7.3 Features of a typical broadacre irrigation farm

The majority of broadacre irrigation farms in the Border Rivers have been built on large flat areas of predominantly clay soils on the floodplains of the Macintyre-Barwon River. The average size of irrigation farms in the Border Rivers/Gwydir/St George area is 135 km^2 which is more than double the average size of cotton farms throughout Australia (Doyle and Coleman, 2006). The majority of these properties are mixed enterprises with, on average, only 18 km^2 developed for broad acre irrigation. The remainder of the farm is assigned to dryland cropping and grazing (67 km^2), native vegetation (20 km^2) and unspecified land use (30 km^2) (Doyle and Coleman, 2006).

Development of a broadacre irrigation property requires extensive planning to establish the position of fields, storages and water channels. A number of features are commonly found on broadacre irrigation properties in the Border Rivers:

1) **Pump site on the riverbank:** This pump station is used to extract water from the river into a supply channel (Plate 2.3).
2) **Supply channel:** This channel is located so that the water pumped into it from the river will flow across the floodplain to the development where it is pumped or gravitates into a storage (Plate 2.4).
3) **One or more storages:** These are used to store water until it is needed for irrigation purposes (Plate 2.5).
4) **Network of farm supply channels, head-ditches and tailwater returns:** These allow water to be moved from the storages to the fields and excess water to be moved back again. They also allow the transfer of water between storages (Plate 2.6 and 2.7).

5) **Other pump stations:** These are used to pump water into the storage from the supply and drainage system and also out to the supply channels (Plate 2.4b).
6) **Large rectangular fields:** These are cleared of trees and laser-levelled so that water runs from the head ditch end of the field to the tailwater end (Plate 2.7).
7) **Levee banks:** These are built around the irrigation development for flood protection and containment of excess irrigation water and storm water runoff (Plate 2.8).

Irrigation fields are furrowed and water is applied using siphons, which transfer water from the head-ditch onto the field (Plate 2.7). Fields are laser-levelled so that water applied will flow from one end of the field to the other to allow rapid and complete runoff and prevent water-logging. This also ensures more efficient use of water during irrigation. Cotton plants can only tolerate being flooded with water for a short period; hence efficient drainage is essential. Once water is flowing from the tailwater end siphons are stopped. It is now common practice for this tailwater to be returned to the storages via channels so that there is less chance of tailwater entering nearby rivers and also to increase water use efficiency in the industry (CA, 2008). Depending on the type of application system and management, the average amount of tailwater produced during surface irrigations is 22% of the water that is applied but this can vary from 0 to 56% with a median figure of 14% (G. Harris, pers. comm., 2006). On average only 85% of this tailwater is recycled due to seepage and evaporation losses (R. Jackson, pers. comm., 2007).

In many cases, the natural flood pattern and path of effluent streams will have been diverted away from the irrigation fields by levee banks either into channels or around the developed area. These levee banks protect the development from flooding but also help to retain storm water runoff, especially if this occurs immediately after a chemical application (Plate 2.8).

2.3a: River pump site

Permanent pump sites extract water from the river.

2.3b: Pipes in the river

Permanent metal pipes in the river are connected to a pump site to harvest water when permitted.

2.3c: A weir in the river

Some properties may have built weirs to collect unregulated flows and pump from these. Others do not have weirs and pump directly from the river under the conditions of their licence agreements.

Plate 2.3: Infrastructure associate with pumping from river channel. Photos: Susan Lutton.

2.4a: Supply channel

Water from the river is pumped into the supply channel and flows across the farm.

2.4b: On-farm pump site

Once water has flowed across the farm via the supply channel it reaches the on-farm pump site and is pumped into one of the storages.

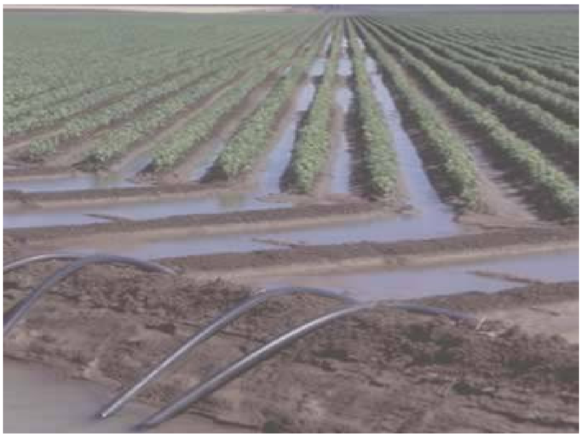
Plate 2.4: Infrastructure associated with moving water across the farm and into a storage. Photos: Susan Lutton.

Water is stored in one or more storages until needed for irrigation purposes.

Plate 2.5: A storage. Photo: Susan Lutton.

Water is released from the storage into a main supply channel. Gravitational force may be sufficient for this or the water may need to be pumped, particularly when the depth of water in the storage is low.

Plate 2.6: Main supply channel. Photo: Susan Lutton.



Water is released from the main supply channel into a head-ditch via a gate in the channel. Water can then be siphoned from the head-ditch onto the field.

Plate 2.7: Head-ditches and field. Photo: Susan Lutton.

A levee bank protects the crops (left of picture) from flood waters on the surrounding floodplain (right of picture). Levee banks also help retain tailwater during heavy rainfall events.

Plate 2.8: Levee bank. Photo: Susan Lutton.

Chapter 3 Morphological characteristics of on-farm water storages and their similarity to natural waterbodies

3.1 Introduction

Maintaining irrigated agriculture in a semi-arid environment with variable rainfall is associated with problems in securing access to water (Deng *et al.*, 2006), so in many irrigation regions of Australia, storages have been built to overcome the problem of an unreliable natural water supply. These storages are large, raised earth structures, designed to hold water for irrigating crops such as cotton, sorghum and wheat. Storages allow water to be harvested from the floodplain or pumped from the river channel during high flow events, stored and used at a later time when flows and rainfall are low. The importance of artificial wetlands for biodiversity in agricultural areas has already been recognised in Europe (Céréghino *et al.*, 2008a; Céréghino *et al.*, 2008b) and the USA (Knutson *et al.*, 2004). With the destruction and decline of natural wetlands, storages in irrigation areas of Australia have been suggested as alternative aquatic habitat for amphibians (Knutson *et al.*, 1999; Hazell *et al.*, 2001; Hazell *et al.*, 2004), waterbirds (Brouwer, 1995) and other aquatic species (Williams *et al.*, 2003; Markwell and Fellows, 2008). However, their morphology and physical habitat characteristics may make them unsuitable as replacement floodplain wetlands.

Floodplain wetlands exist in many shapes and sizes and waterholes with different physical characteristics vary in their biological productivity and in turn, the organisms which they support (Davis *et al.*, 2002). Hydrological connection history has been shown to influence invertebrate assemblage composition of natural waterholes (Sheldon *et al.*, 2002; Marshall *et al.*, 2006) and wetlands (Timms, 2001; Jenkins and Boulton, 2003) while the availability of physical habitat at a range of scales can also influence assemblage composition in rivers (Sheldon and Walker, 1998). Due to this variation, classification systems are well developed for natural wetlands. Wetlands have been grouped using features such as hydrology and geomorphology so as to allow generalisations about each class in terms of management or conservation potential (Cowardin and Golet, 1995).

As with natural wetlands, storages can be morphologically and hydrologically diverse and also potentially support diverse faunal and floral assemblages. An important step in understanding how effective storages are as alternatives to natural wetlands, is to understand the variety of storage types and how their hydrological and morphological characteristics compare to those of natural wetlands. This is a vital step in identifying both pattern (assemblage composition) and process attributes of storages and is critical for understanding the conservation value and recommending best management practices to landholders.

3.1.1 Aims

Due to the number of storage sites on the floodplain of the Border Rivers Catchment and their potential as aquatic habitat, it is essential to achieve some understanding of the characteristics of storages and how they compare morphologically with natural wetlands. This chapter explores the degree of variation in the physical and hydrological characteristics of storages in the irrigation area of the Border Rivers Catchment and uses this classification to comment on how effective storages may be as alternative habitat for aquatic fauna. Specifically it is hypothesised that:

1) Morphological and hydrological spatial variability will be limited amongst storages in the Border Rivers Catchment.
2) The majority of storages will be managed, such that they contain tailwater.
3) When compared with natural wetlands, the majority of storages will have a very distinctive morphology.

3.2 Methods

3.2.1 Data collection

A sample of water storages in the Border Rivers Catchment, totalling 99 storages distributed across 41 properties were identified as study sites using industry representatives, aerial photographs and maps (See Fig. 2.2; refer to Appendix 1 for examples of sites). Hydrological, morphological and management data were obtained via site visits, Geographical Information System (GIS) mapping and landholder interviews. Site visits were conducted between 22nd December 2004 and 21st February 2005. GPS coordinates were taken at the corners of each storage and subsequently incorporated into a GIS map, which was then used to calculate variables describing aspects of size and shape for each storage site (Table 3.1). Landholders were either interviewed in person or completed a questionnaire about each of the storages on their property (Appendix 3). This provided management information, hydrological and morphological data that could not be obtained from site visits or the GIS map. The complete data set of 99 storages was used for the classification of storage sites.

A total of 70 natural floodplain wetlands, including lagoons or waterholes in flood channels, anabranch channels and associated billabongs, were selected from aerial photographs of the Border Rivers Catchment and GIS layers for the Murray Darling Basin (refer to Appendix 2 for examples of sites). Only wetlands covering an area of greater than 20,000 m^2 were included. Variables describing the position on the floodplain and the size, shape, surface area, perimeter and length of each natural wetland were generated from the GIS map (Table 3.1) with five measured transects along each wetland used to calculated the average width. Landholders provided the average depth for 20 of these natural wetlands and the values were used to estimate the bankful volume. Landholders were unable to provide length and width measurements for two of the storage sites, therefore a subset of 97 storages with complete length and width measurements was used for comparison with natural wetlands.

Table 3.1: A description of the variables calculated as a measure of storage and natural wetland shape.

Name	Symbol	Description	Data Source
Depth (m)	D	Height of embankment around edge of storage Average depth of the natural wetland when full	Landholder Landholder
Surface Area (m^2)	A	Generated from Arcview	GIS Map
Perimeter (m)	P	Generated from Arcview	GIS Map
Length (m)	L	Measured using Arcview	GIS Map
Width (m)	W	Average bankfull width measured using Arcview	GIS Map
Length:Width ratio	LW	Length of waterhole divided by width	GIS Map
Capacity (m^3)	C	The amount of water held by the storage when full Estimated volume of water held by the natural wetland when full	Landholder GIS Map
Distance from River (km)	Dis	The shortest distance from the waterhole to the Barwon-Macintyre River	GIS Map
Circularity Index	CI	The circularity index was calculated using a formula to determine the shape of drainage basins (Miller, 1953) $F=4\Pi A/P^2$ Where A is the surface area of the waterhole and P is the perimeter of the waterhole. As F approaches 1 the waterhole will be more circular in shape, as F approaches 0 the waterhole will become more linear in shape	GIS Map
Horton’s Form Factor	HFF	Horton’s form factor is another calculation normally used to obtain a measure of drainage basin shape (Horton, 1932) $F=A/L$ Where is A is the surface area of the waterhole and L is the length of the waterhole	GIS Map
Elongation Ratio	ER	Elongation ratio was calculated for each of the wetlands (Schumm, 1956) ER=diameter of a circle with the same area as the waterhole/length	GIS Map

Table 3.2: A description of the morphology variables recorded for each storage.

Name	Description	Data Source
Age (Years)	Using the year the storage was built, age was calculated at time of analysis.	Landholder
Surface Area (m^2)	Generated from Arcview	GIS Map
Perimeter (m)	Generated from Arcview	GIS Map
Depth (m)	Height of embankment around edge of storage	Landholder
Capacity (m^3)	The amount of water held by the storage when full	Landholder
Distance from source river (km)	The distance from the river that water is pumped from during allocated and unregulated flows*	Landholder

*Where access to more than one river was available, the river that was closest to the storage was used.

Table 3.3: A description of the hydrology variables recorded for each storage.

Name	Description	Source
Groundwater	Presence/absence of groundwater.	Landholder
Allocated flows	Presence/absence of water from regulated flows.	Landholder
Unregulated flows	Presence/absence of water from unregulated flows.	Landholder
Overland flow	Presence/absence of water from overland flow.	Landholder
Tailwater	Presence/absence of tailwater.	Landholder

3.2.2 Classification of Storages

Storages were classified using both morphology and hydrology variables, with morphological variables including data relating to size and shape as well as age of the storage (Table 3.2) and hydrology variables reflecting the source of water used to fill each storage (Table 3.3). Five different sources of water for irrigation purposes were identified; (i) “allocated” (supplemented) flows, those supplied by releases from dams and supplemented by tributary flows; (ii) “unregulated” (supplementary) flows, those supplied by natural drainage and tributary flows; (iii) “overland” flow (floodplain harvesting), that harvested predominantly from out of bank flows during floods; (iv) “groundwater”, supplied from both alluvial and artesian underground sources; and (v) “tailwater”, any excess water that runs off an irrigation field while being watered. One storage may contain any combination of these five sources of water.

3.2.3 Data analysis

Variations in morphology between waterbody types were explored using frequency of occurrence histograms and scatter plot matrices. Initial analyses included all 70 wetlands and 97 storage sites but had a reduced number of variables because depth and capacity were not available for all of the former and so were omitted from the analyses. A second set of analyses, comparing depth and capacity only, were conducted using the subset of 20 natural wetland sites for which these measurements were available.

Both spatial variability (morphology) and hydrological variability were used to classify the storage sites into meaningful groups. Dendrograms for both the morphological dataset and the hydrological dataset were generated from the raw data using hierarchical agglomerative cluster analysis (UPGMA) in the Primer 5 software package (PRIMER 5.2.9; Clarke and Gorley, 2001). Normalised Euclidean distance was used as the measure of similarity, with group averaging used to generate the clusters. The hydrology and morphology dendrograms were then compared using a manual two-way cluster analysis which separated the storages into groups based on both hydrology and morphology.

While classification provided us with a mechanism for grouping storages based on hydrology and morphology, it was essential to further explore the morphological and hydrological attributes driving the differences. Principal Component Analysis (PCA) was used to isolate those variables contributing most to the difference between storages. PCA was conducted using the default settings in Primer 5; normalised data and the maximum number of components set at 5 (PRIMER 5.2.9; Clarke and Gorley, 2001).

3.3 Results

3.3.1 Comparison of storages with floodplain wetlands

Floodplain wetlands in the Border Rivers Catchment were very different from storages with respect to morphology. Storages tended to be bigger than natural wetlands with a greater area and larger capacity (Table 3.4, Fig. 3.1 and 3.2). There were also differences in shape. Although natural wetlands tended to cover a smaller area (Fig. 3.2b and Fig. 3.3b), they had longer perimeters than storages (Fig. 3.2c and Fig. 3.3c), implying that natural wetlands were more irregular in shape.

The histograms (Fig. 3.2) and box plots (Fig. 3.3) for L:W, CI, HFF, ER and depth illustrate clear differences between storages and natural wetlands. Natural wetland length ranged from 185 m to 8959 m while the depth varied from 0.8 m to 3 m. By comparison, storages were considerably shorter (Fig. 3.2d and Fig. 3.3d), varying from 320 m to 2030 m, but were generally deeper (Fig. 3.2a and Fig. 3.3a), varying from 1.5 m to 8 m. Natural wetlands tended to be long (mean length of 1974 m) and meandering (mean L:W ratio of 21.41 and mean ER of 0.33) compared with storages (mean length of 893 m; mean L:W ratio of 2.00 and mean ER of 0.86) (Fig. 3.2d, f, k and Fig. 3.3d, f, k). The circularity index (CI) and Horton’s form factor (HFF) suggested storages were more likely to resemble circles (mean CI: 0.738; mean HFF of 497.80) while floodplain wetlands were more likely to be linear in shape (mean CI of 0.119; mean HFF of 119.61) (Fig. 3.2i, j and Fig. 3.3i, j). On the floodplain; storages occurred closer to the river with a mean distance of 6.2 km compared with natural wetlands, with a mean distance of 11.1 km (Table 3.4), reflecting the need for storages to be built as close to the river as possible to reduce the distance that water has to be pumped.

Table 3.4: Mean values for each of the variables for both floodplain wetlands and storage sites. *Only 20 floodplain wetland sites were included for these variables

	Natural Wetlands				Storages			
	Max.	**Min.**	**Mean**	**S.E.**	**Max.**	**Min.**	**Mean**	**S.E.**
D (m)*	3.0	0.8	1.6	0.1	8	1.5	4.8	0.1
A (m^2)	1999802	10100	245398	37352.09	1780000	50000	469344	31266.82
P (m)	28710	750	6124	635.24	6270	1010	2733	95.06
L (m)	8959	185	1974	194.52	2030	320	893	33.74
W (m)	559	39	115	10.620	938	159	490	19.28
L:W	109.26	1.45	21.41	2.32	4.91	1.0	2.00	0.09
C (m^3)*	3999604	10100	705795	212904.0	10000000	150000	2028990	157223.47
Dis (km)	40.3	0.1	11.1	1.145	37.9	0	6.2	0.721
CI	0.416	0.012	0.119	0.01	1.187	0.4	0.738	0.02
HFF	525.58	17.69	119.61	10.22	1635.99	143.3	497.80	21.58
ER	1.01	0.11	0.33	0.02	2.06	0.5	0.86	0.02

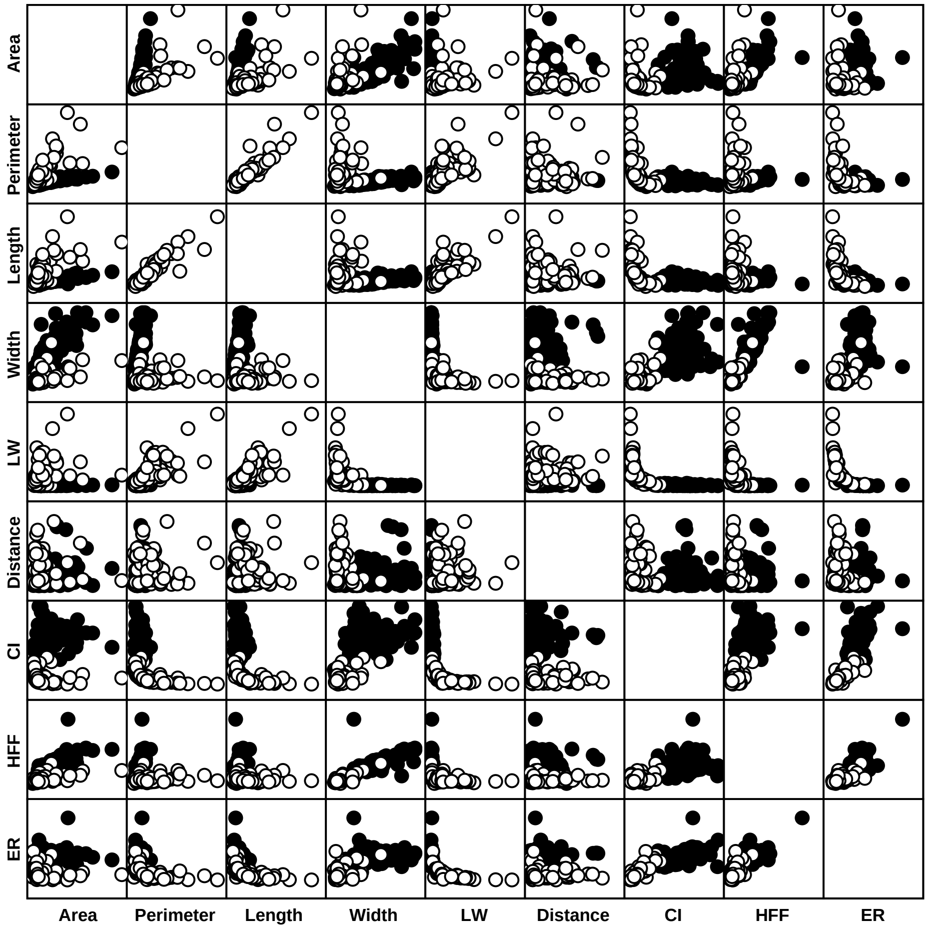


Fig. 3.1: Scatter matrix for 97 storages and 70 natural wetlands. Storages are represented by closed circles and natural wetlands are represented by open circles.

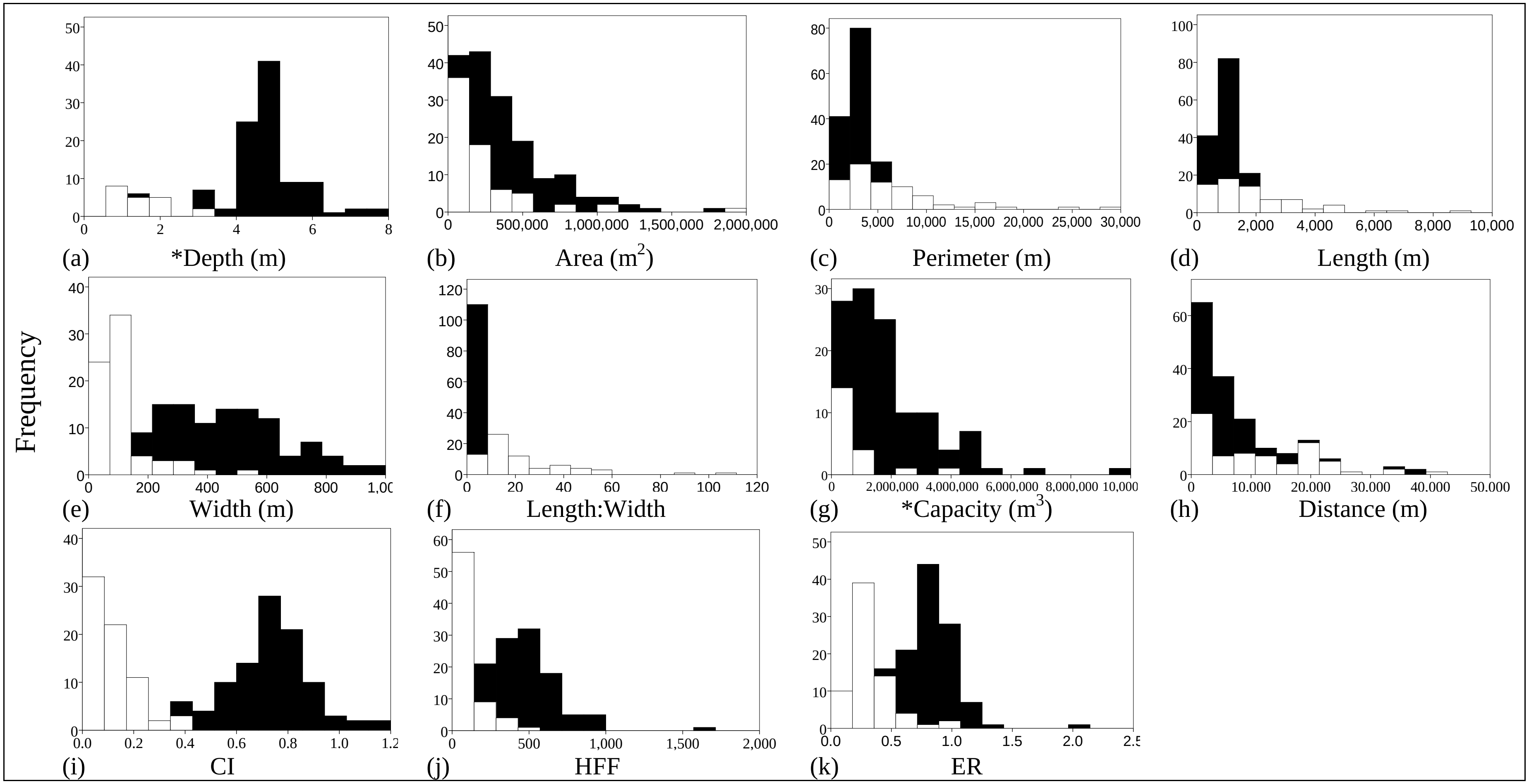
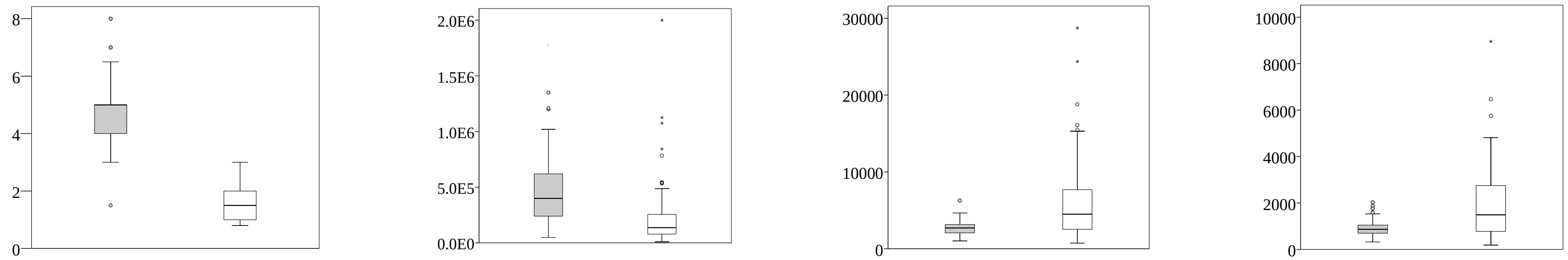
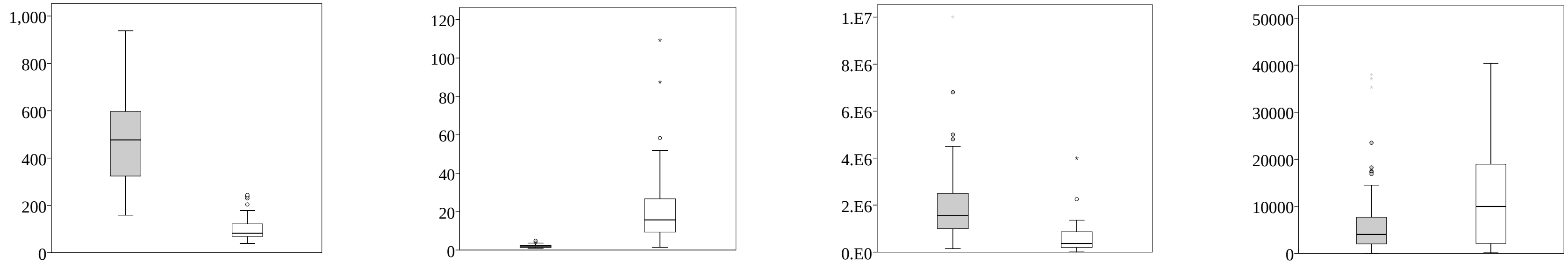


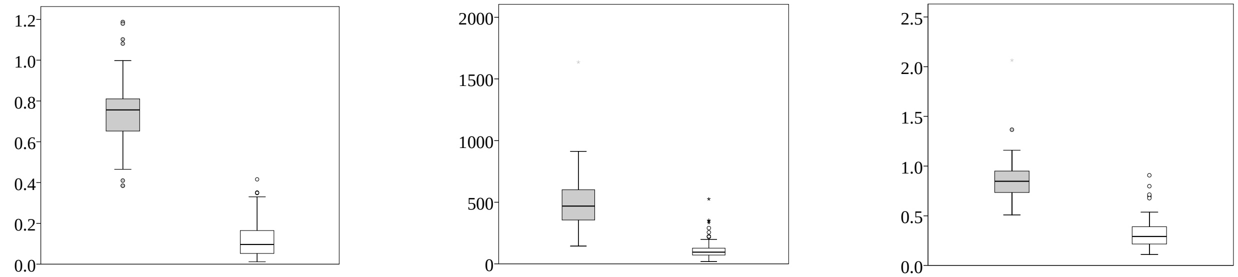
Fig. 3.2: Frequency of occurrence histograms of storages and natural wetlands. *Only includes 20 natural wetland sites. Black bars represent the storages and white bars represent the natural wetlands.



(a) *Depth (m) (b) Area (m^2) (c) Perimeter (m) (d) Length (m)



(e) Width (m) (f) Length:Width (g) *Capacity (m^3) (h) Distance



(i) CI (j) HFF (k) ER

Fig. 3.3: Box plots of morphology variables for storages and natural wetlands. *Only includes 20 natural wetland sites. Grey bars represent the storages and white bars represent the natural wetlands.

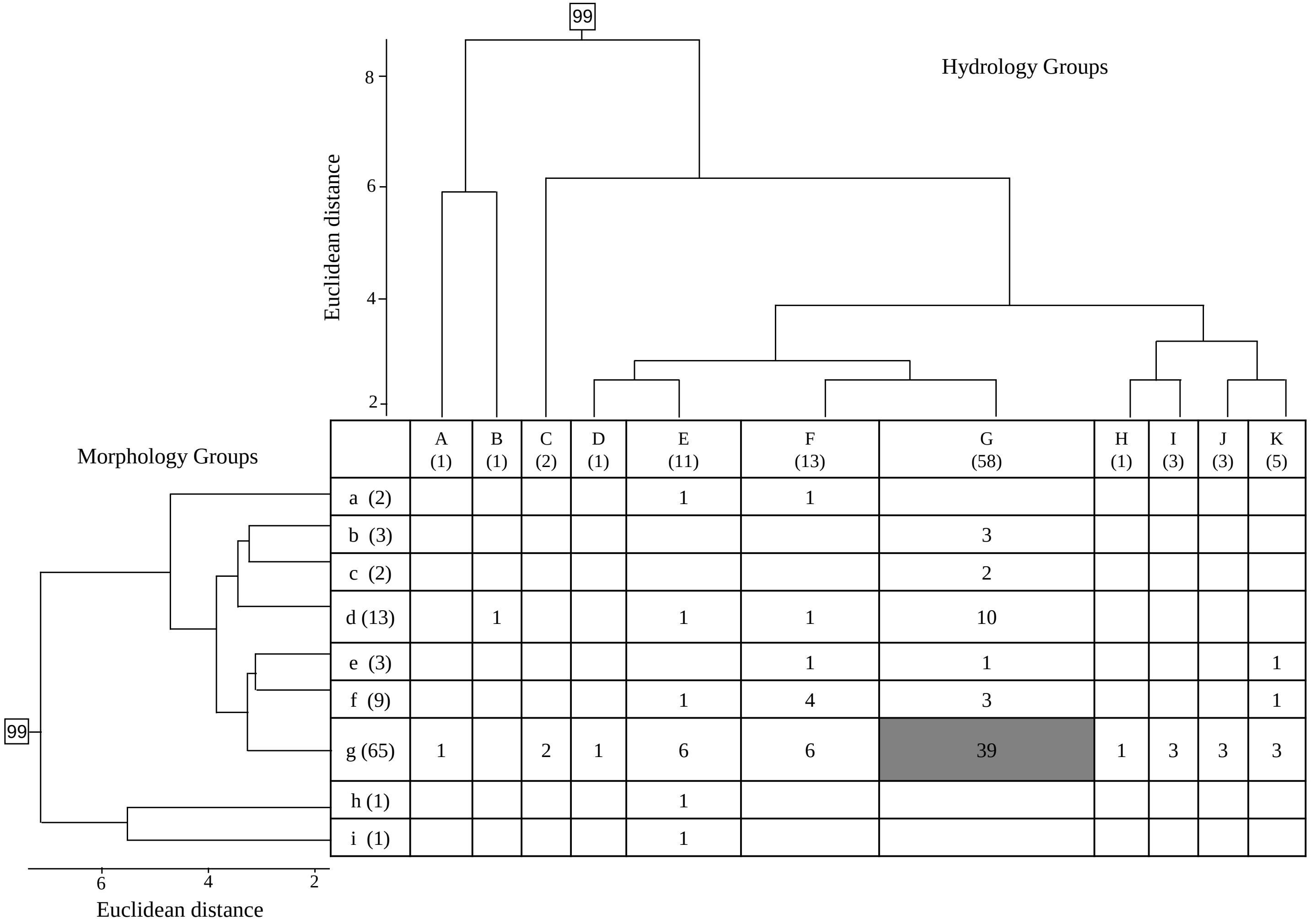
3.3.2 Classification of storages

Of the 99 storages analysed 12 did not contain tailwater and three had access to groundwater. Most were filled by a combination of allocated flows, unregulated flows and overland flows and contained recycled tailwater. All but two had access to river flows either in the form of allocated or unregulated flows or both (Appendix 4).

Morphologically, storages in the Border Rivers Catchment were quite diverse (Fig. 3.2). The oldest storage surveyed in this study was built in 1977 and the youngest in 2004. Older storages were generally smaller compared to younger storages; the oldest covered an area of only 110,000 m^2 and had a capacity of 300,000 m^3 while the youngest had a surface area of 810,000 m^2 and could hold 4,000,000 m^3 of water. Depth varied from 2 m to 8 m and capacity from 150,000 m^3 to a vast 10,000,000 m^3 (Appendix 5).

Hydrologically the storages clustered into 11 hydrological groups that reflected the source of water to the storage (Fig. 3.4 and Table 3.5). River flows, either in the form of unallocated or allocated flows, were present in all but two of the groups (A and B), overland flow was predominant in groups A, B, C, E, G, I & J, groundwater was evident only in Groups A and C and only four groups did not contain tailwater (H-K). The majority of sites, 58 in total, were found in one group (G). With the exception of groundwater, this group contained all sources of water including tailwater.

In relation to morphology, most storages were very similar, forming one large group in the cluster, six smaller groups and two extreme outliers; site 77 and site 33 (Fig. 3.4 and Table 3.6), however, four of the groups contained three or less sites. The majority of storages, 65 in total, were found in group ‘g’, which had a mean age of 10 years and was found close to the source river (mean of 3 km). Storages represented in group ‘g’ were relatively small when compared with the other groups (mean area of 378 809 m^2) (Fig. 3.4 and Table 3.6).



	A (1)	B (1)	C (2)	D (1)	E (11)	F (13)	G (58)	H (1)	I (3)	J (3)	K (5)
a (2)					1	1					
b (3)							3				
c (2)							2				
d (13)		1			1	1	10				
e (3)						1	1				1
f (9)					1	4	3				1
g (65)	1		2	1	6	6	**39**	1	3	3	3
h (1)					1						
i (1)					1						

Fig. 3.4: Two-way cluster output of the dendrogram based on hydrology data and dendrogram based on morphology data for 99 storages in the Border Rivers Catchment. Shaded box depicts 'typical' storages.

Table 3.5: Storage site groups created by cluster of hydrology data where AF = Allocated Flow, UF = Unallocated Flow, OF = Overland Flow, GW = Groundwater, TW = Tailwater.

Group	No. of Sites	Type of Water Present
A	1	OF, GW, TW
B	1	OF, TW
C	2	AF, UF, OF, GW, TW
D	1	UF, TW
E	11	UF, OF, TW
F	13	AF, UF, TW
G	58	AF, UF, OF, TW
H	1	UF
I	3	UF, OF
J	3	AF, UF, OF
K	5	AF, UF

Table 3.6: Mean values for characteristics of each cluster group based on morphology (± s.e.). Number in brackets is the number of sites in that group.

Groups	Mean Age (Yrs)	Mean Height (m)	Mean Area (,000 m^2)	Mean Perimeter (,000 m)	Mean Capacity (,000 m^3)	Mean Distance (km)
a (2)	23.5	2.75	225	5.50	450	1.55
	(± 1.5)	(± 0.75)	(± 75)	(± 0.50)	(± 150)	(± 1.450
b (3)	4.3	5.83	850	3.68	4267	8.33
	(± 0.3)	(± 0.60)	(± 150)	(± 0.39)	(± 233)	(± 0.33)
c (2)	6.5	7.50	560	2.96	3556	0.25
	(± 0.5)	(± 0.50)	(± 40)	(± 0.06)	(± 44)	(± 0.05)
d (13)	8.2	5.14	903	4.13	3900	1.00
	(± 1.5)	(± 0.24)	(± 57)	(± 0.12)	(± 233)	(± 0.27)
e (3)	17.0	2.50	150	1.49	350	0.30
	(± 3.6)	(± 0.50)	(± 29)	(± 0.14)	(± 104)	(± 0.25)
f (9)	22.3	4.78	228	1.90	872	7.56
	(± 1.0)	(± 0.22)	(± 47)	(± 0.23)	(± 182)	(± 0.83)
g (65)	10.1	4.73	379	2.50	1546	2.96
	(± 0.7)	(± 0.09)	(± 21)	(± 0.07)	(± 87)	(± 0.31)
h (1)	8.0	5.00	1780	6.27	6800	10.00
i (1)	4.0	8.00	1210	4.40	10000	0.20

The manual two-way cluster analysis combining output from the hydrological and morphological clusters separated the storages into 27 distinct groups based on a combination of hydrology and morphology variables (Fig. 3.4). Of these, 15 groups were comprised of only one storage. The large number of groups and the fact that many of them contained only one site, reflects the variability of storage types. However, the analysis revealed that most storages, 39 in total, could be placed in one group (Fig. 3.4). This group comprised storages containing all sources of water apart from groundwater and had very similar characteristics to morphology group ‘g’ (Table 3.6); with an average age of 10 years, mean embankment height of almost 5 m and area of just over 400,000 $m^{2.}$ They were located on average about 3 km from the source river, had a perimeter of 2.5 km and could hold almost 1,700,000 m^3 of water.

Storages found in this group were considered to be a “typical” on-farm storage in this region.

3.3.3 Descriptors of Storage Groups

When storage hydrology data were explored with PCA (Fig. 3.5) 57% of the total variation was captured in the first two principal components (Table 3.7). PC1 explained 32% of the variation with unregulated flows (-0.637) and groundwater (0.545) loading on this component. PC2 explained 25.4% of the variation with tailwater (0.725) and overland flow (0.534) loading on PC2. PCA placed hydrology group ‘G’ low on PC1 and high on PC2 (Fig. 3.5), suggesting sites in this group contained tailwater, overland flow and unregulated flows but not groundwater.

The PCA of storages based on morphology data (Fig. 3.6), again suggested differences between storage groups with 84% of the total variation explained by the first three principal components (Table 3.8). PC1 explained 49% of the variation and was related to storage size, with capacity (-0.556), area (-0.546) and perimeter (-0.478) loading on this component. PC2 explained 18% of the variation and was positively correlated with age (0.653); PC3 explained 17% of the variation with distance from river (0.683) and height (0.583) loading on this component. PCA placed morphology group ‘g’ high on PC1 (Fig. 3.6), suggesting storages in this group were smaller in size compared to storages in other groups. Group ‘g’ occurred low on PC2 which was correlated with age, suggesting storages in this group, while somewhat variable, were constructed relatively recently.

Table 3.7: Percent of variance of storage hydrology extracted by PCA for the first 5 axes and contribution of the measured variables to the first 5 eigenvectors.

	Axis	**Eigenvalue**	**% of Variance**	**Cum. % of Variance**	
	1	1.58	31.7	31.7	
	2	1.27	25.4	57.0	
	3	0.97	19.3	76.3	
	4	0.68	13.6	89.9	
	5	0.51	10.1	100.0	
Characteristic		**First 5 Eigenvectors**			
	1	2	3	4	5
Allocated Flow	-0.413	0.394	0.631	0.225	0.474
Unregulated Flow	-0.637	0.183	-0.164	0.336	-0.649
Overland Flow	0.315	0.534	-0.509	0.542	0.250
Groundwater	0.545	-0.016	0.556	0.431	-0.456
Tailwater	0.165	0.725	0.085	-0.597	-0.289

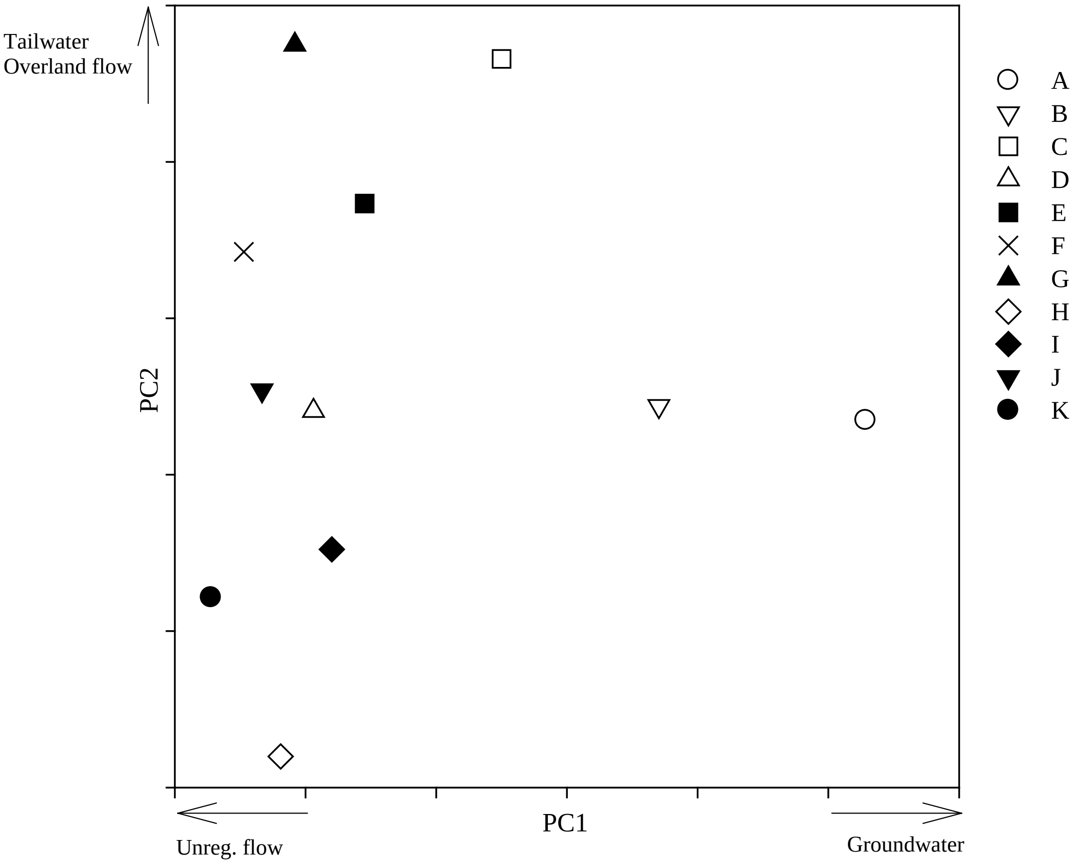


Fig. 3.5: Position of each storage site within the ordination space defined by the first two factors identified by PCA of hydrology data. Symbols show hydrology groups as identified by Cluster.

Table 3.8: Percent of variance of storage morphology extracted by PCA for the first 5 axes and contribution of the measured variables to the first 5 eigenvectors.

	Axis	**Eigenvalue**	**% of Variance**	**Cum. % of Variance**	
	1	2.93	48.8	48.8	
	2	1.07	17.8	66.6	
	3	1.05	17.4	84.0	
	4	0.67	11.2	95.2	
	5	0.21	3.5	98.7	
Characteristic	**First 5 Eigenvectors**				
	1	2	3	4	5
Age	0.251	0.653	-0.228	-0.672	0.076
Height	-0.316	-0.267	0.583	-0.615	-0.312
Area	-0.546	0.216	-0.115	0.051	0.319
Perimeter	-0.478	0.266	-0.356	0.127	-0.724
Capacity	-0.556	0.055	0.038	-0.083	0.516
Distance from River	0.044	0.618	0.683	0.380	-0.060

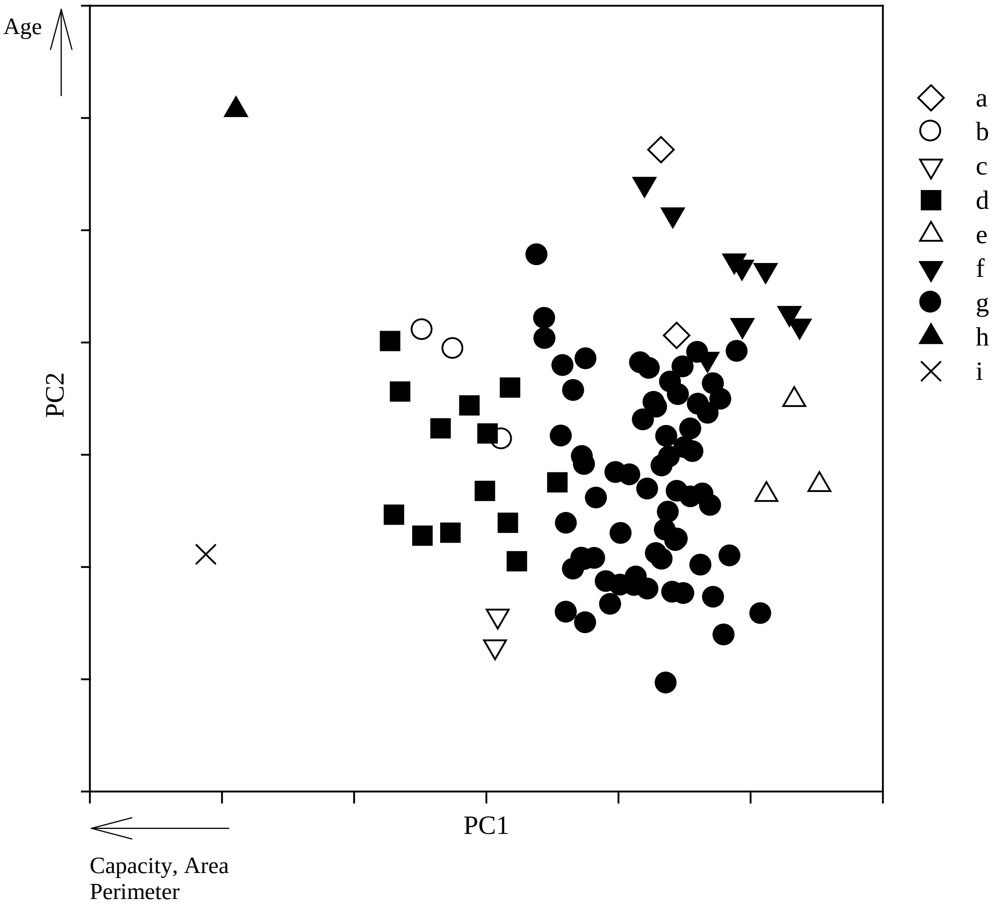


Fig. 3.6: Position of each storage site within the ordination space defined by the first two factors identified by PCA of morphology data. Symbols show morphology groups as identified by Cluster.

3.4 Discussion

3.4.1 Comparison of storages with floodplain wetlands

This study clearly showed that, within the Border Rivers Catchment, storages and natural wetlands differed markedly with respect to their morphology. In general, storages were large, deep structures with a more regular shape resembling squares or circles, while natural wetlands were shallower and had irregular shapes with high length to width ratios and large perimeters. These morphological differences between natural wetlands and the storages are significant as they are likely to influence aquatic species composition, diversity and abundance as well as important ecosystem processes such as littoral production (Hansson *et al.*, 2005).

The amount, type and complexity of habitat has been found to positively correlate with the abundance and diversity of fish and macroinvertebrate populations (O'Connor, 1991; Bunn and Arthington, 2002). As previously described, increased channel complexity is also likely to increase biodiversity (see Chapter 1, Section 1.4.1). Natural wetland sites in the Border Rivers Catchment have been described as generally having poor aquatic habitat (Medeiros, 2004). The uniform nature of storage sites and the observed lack of aquatic habitat (woody debris, snags) suggest that habitat diversity at storages will be even lower, limiting the aquatic biodiversity of storages and also the abundance of those organisms that can survive in storages..

The observed changes in storage design over the relatively short history of irrigated agriculture, from smaller, shallower structures to larger, deeper structures, suggest a progression towards ever larger storages over time. If this trend continues the number of artificial aquatic sites per cubic metre of water stored will reduce. This combined with the reduction in size and number of natural wetlands will continue to lessen the amount of aquatic habitat in the Border Rivers Catchment and have a detrimental effect on aquatic species across the region.

Seasonal changes in morphological characteristics of floodplain lagoons have been well documented in several Australian river systems (Hart and McGregor, 1982; Kennard, 1995). Over the course of a season natural wetlands will experience wetting and drying periods and in turn fluctuations in water levels. When these occur, the morphology of the natural wetland will change; for instance, as it is flooded a wetland will increase in overall size (area, volume and perimeter) and depth (Medeiros, 2004). In contrast, the steep sides and regular shape of storages means as they are filled the water does not spread over lower portions of the floodplain, thus changing the shape of the aquatic habitat, but merely changes the water level in the storage. Therefore, the temporal variation in habitat characteristics observed in natural wetlands in association with filling and drying (Medeiros, 2004) may not be evident at storage sites. Flooding also allows a two way exchange of material between floodplain wetlands and the river channel. In comparison, the major exchange pathway for biota and materials in storages is only one way, via the intake pump or channel.

The presence of tailwater in the majority of storages is likely to have a negative affect on the aquatic assemblages therein. A number of pesticides and herbicides have been detected in tailwater (Crossan, 2002; Rose, 2006) and might therefore accumulate in those storages that hold recycled tailwater (Crossan, 2002). The presence of chemicals in these storages is likely to limit the diversity of aquatic assemblages found within (Ward et al., 1995).

The age of wetlands has been found to influence the species richness of invertebrates and affect the abundance and richness of aquatic vegetation (Hansson *et al.*, 2005) with older wetlands likely to have more complex littoral zone habitats (Markwell and Fellows, 2008). This is particularly relevant as the storages were much younger structures than the established natural wetlands. The young age of many of the storages is also reflected in their reduced or absent riparian zone. Even if riparian vegetation was encouraged to grow on the banks of storages it would be very different to that found at natural wetlands because of the time required to establish trees and understorey vegetation. Natural wetlands have a multilayered band of riparian vegetation along their edge (Lovett *et al.*, 2003) which influences the abundance and richness of aquatic assemblages (Markwell and Fellows, 2008). The observed lack of riparian vegetation around many storages is therefore likely to have an adverse effect on the composition of aquatic communities.

3.4.2 Classification of storages

Despite the marked differences between storages and natural wetlands, there was a high degree of variability within the storages themselves. However, about 40% of the storages sampled could be placed in one group which shared similar morphological and hydrological characteristics and can be considered to be a ‘typical storage’ group. The ‘typical storage’ in the Border Rivers region contained water sourced from a combination of allocated flows, unallocated flows, overland flows and recycled tailwater, was not particularly large and was constructed relatively recently. Within and between site variability of macroinvertebrate populations as a result of habitat diversity has been well documented in Australian natural wetlands (Sheldon and Walker, 1998; Marshall *et al.*, 2006) and other parts of the world (Scarsbrook and Townsend, 1993; Sandin and Johnson, 2004). The fact that most storages are similar and morphologically simple suggests that this diversity will not be maintained if natural wetlands are replaced by farm storages.

3.4.3 Conclusions

Irrigated agriculture in semi-arid environments such as the Border Rivers region of Australia means storages have become an increasingly significant part of the floodplain landscape. As the abundance of natural wetlands decline, understanding how storages compare with natural wetlands is an important management and conservation issue.

Storages primarily function as water supplies and their associated management makes them mostly unsuitable as ‘replacement’ wetlands. However, given the large numbers of storages across the catchment, they might offset some of the loss of wetland habitat and help maintain regional biodiversity if they can be designed and managed with this additional objective. To maximise the biodiversity of storages it will be essential to reduce the morphological homogeneity of storages across the landscape and increase habitat diversity within storage sites. In the future, improved design of new storages and alterations to existing storages and their management could help overcome this problem of low diversity of habitat. New storages built with gently sloping sides, shallow areas and central islands would create habitat for wading birds, frogs and invertebrates (Broome and Jarman, 1983; Jarman and Montgomery, 2002). If existing storages were split into smaller cells this would not only decrease evaporation losses but increase the bottom surface area for pesticide breakdown and improve water quality (Kennedy and Jarman, 2006). Planting aquatic vegetation and adding coarse woody debris to the banks of storages will provide additional habitat, shelter and a food source for aquatic species. Tailwater could also be managed so that it is limited to only one storage on the property, improving the water quality in the remaining storages (Lutton, 2005; Kennedy and Jarman, 2006). A number of property holders are starting to implement some of these design changes to their storages which will not only improve aquatic biodiversity but also benefit the farmer (CCC CRC, 2008).

As a group, storages in the Border Rivers Catchment are still fundamentally different to natural waterbodies and most of the storages fall into a fairly uniform group. If we are to sustain the aquatic biodiversity in the Border Rivers Catchment and other similar irrigation regions we need to preserve the diversity of available aquatic habitat.

Chapter 4 Fish assemblages from three different waterbody types in the Border Rivers Catchment

4.1 Introduction

The wetlands of floodplain rivers are unique aquatic systems that straddle the divide between lentic habitats (no flow – specifically during periods of disconnection from the river network) and lotic habitats (during periods of connection by flooding). The structure of aquatic communities in natural wetlands is influenced by both physical and hydrological factors. Morphological features such as volume, surface area and depth are believed to shape the overall characteristics of the ecosystem (Jackson *et al.*, 2001; Olden *et al.*, 2001; Bunn and Arthington, 2002), while increased channel complexity is associated with variations in assemblage composition (Davis *et al.*, 2002; Thoms *et al.*, 2006). The amount of habitat available, and its complexity, is also known to be an important driver of aquatic community structure (House and Boehne, 1985; O'Connor, 1991), by providing protection (Everett and Ruiz, 1993) and in some cases suitable spawning sites (Rowland, 1983) or a food source (Marsalek, 2006).

In irrigation areas of Australia, water storages on the floodplain may provide a viable aquatic habitat for a range of flora and fauna. The on-farm wetlands and storages in the Lower Gwydir Valley, Murray-Darling Basin, are seen as useful aquatic habitats for waterbirds (Jarman and Montgomery, 2002) and macroinvertebrates (Brooks, 1995). However, the suitability of storages for waterbirds increases with increasing habitat complexity in the form of trees, aquatic vegetation, islands and shallows (Jarman and Montgomery, 2002). For most artificial storages, however, morphological differences between natural wetlands and the more uniform storages (Lutton *et al.*, In press) are likely to lead to differences in species composition, diversity and abundance. The lack of aquatic habitat and vegetation within storages is likely to reduce their aquatic diversity and abundance compared with natural wetlands.

There are also major hydrological differences between natural wetlands and storages. Over the course of an irrigation season the level of water in a storage may be raised and lowered a number of times depending on the availability of water in that year. During the peak irrigation period water levels can drop rapidly as a result of watering crops combined with natural water losses (e.g. evaporation, seepage). In comparison, the level of water in natural wetlands will only be affected by natural water losses. These differences, along with the morphological differences as a storage or wetland fills and dries (Lutton *et al.*, In press), are likely to result in different temporal patterns of aquatic communities in both types of water bodies.

For fish there is the added difficulty of connection. Dispersal between natural wetlands and in-channel waterholes within river networks occurs through connection resulting from channel and floodplain flow (Arthington *et al.*, 2005; Balcombe *et al.*, 2006; Balcombe *et al.*, In press). As artificial storages are rarely or never connected to the river network, the only way fish can access this habitat is through human transfer (stocking) or opportunistically as eggs, larvae or juveniles via irrigation pumps transferring water into storages from the river channel.

4.1.1 Aims

The aim of this chapter was to compare water quality and fish assemblage composition of artificial storages and natural wetlands. Water quality and fish assemblage data were collected from typical storages, ‘no tailwater’ storages and natural wetlands in the Border Rivers Catchment during the spring of 2005. The specific objectives of this chapter are to:

1) Compare water quality between three waterbody types, typical storages, ‘no tailwater’ storages and natural wetlands, in the Border Rivers Catchment to test the hypothesis that natural wetlands have less variable water quality.
2) Compare the diversity and abundance of fish between the three waterbody types, typical storages, ‘no tailwater’ storages and natural wetlands, in the Border Rivers Catchment to test the hypothesis that natural wetlands have greater diversity and abundance of fish compared with artificial storages.
3) Compare fish aquatic assemblage structure, size classes, between the three waterbody types in the Border Rivers Catchment to test the hypothesis that the fish composition of natural wetlands is represented by all sizes classes.

4.2 Study Area and Design

In order to evaluate whether there were differences in fish communities and biodiversity between different waterbody types, fish samples were taken from natural wetlands, typical storages and storages with the same morphological characteristics as the ‘typical’ category but without tailwater inputs (‘no tailwater’, see Chapter 3). The storages and natural wetlands were selected such that sampling sites were spread throughout the Border Rivers Catchment (Fig. 4.1). Sampling was carried out between August and October 2005, over an eight week period. Due to low water levels throughout the catchment it was only possible to collect fish samples from five typical storages, five natural wetlands and three ‘no tailwater’ storages (Table 4.1).

Table 4.1: Sites in the Border Rivers Catchment sampled for fish during the spatial study (Time 1) from August 2005 to October 2005, NW = natural wetland, TW = ‘no tailwater’ storage, TS = typical storage.

Site	Site Code
Crawler Lagoon	NW1
Newinga Weir Waterhole	NW4
Punbougal Lagoon	NW5
Tarrawatta Weir Waterhole	NW7
Yambocully Lagoon	NW8
Boolarwell S2	TW2
Newinga S2	TW3
Tarrawatta Weir Dam	TW4
Jericho S1	TS1
Royston S3	TS2
South Callandoon S2	TS3
Teriadi S2	TS4
Undabri S2	TS5

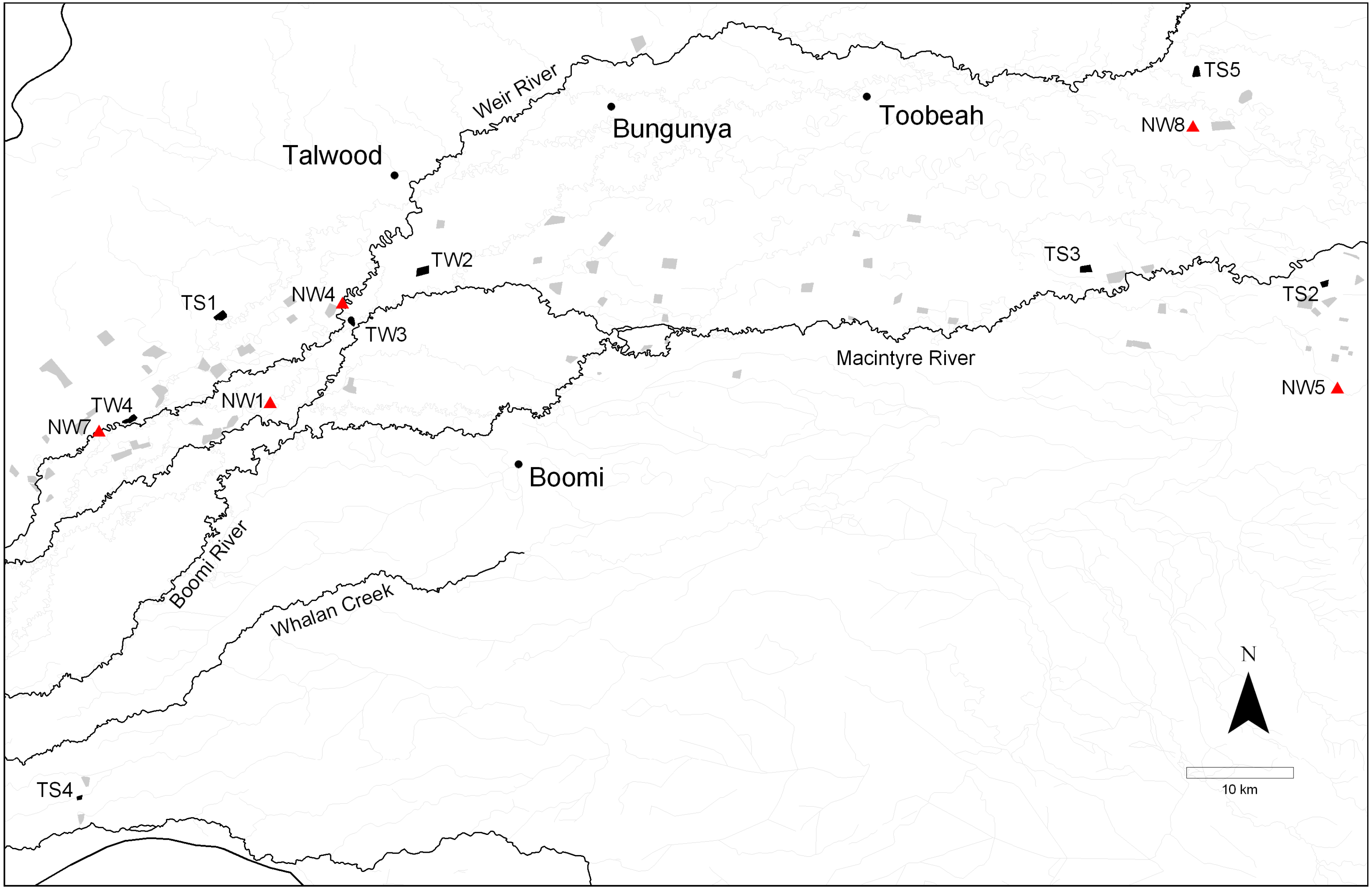


Fig. 4.1: Natural wetlands (triangles) and storage sites (black objects) in the Border Rivers Catchment sampled for fish during the spatial study from August 2005 to October 2005. Grey objects represent other storages in the area. NW = natural wetland, TW = 'no tailwater' storage, TS = typical storage.

4.3 Methods

4.3.1 Water Quality

Three replicate water samples were taken from each waterbody using either a 1 m plastic tube to obtain an integrated sample from the top 1 m of the water column or, at sites where the water depth was below 1 m, a bucket. Conductivity (mS cm^{-1}), pH, spot dissolved oxygen (DO) (mg L^{-1}), turbidity (NTU), spot water temperature (^{o}C) and salinity (%) were measured in the field using a Horiba multiprobe U-10.

Mean water quality variables were calculated from three replicate water samples taken from each site. All water quality variables fitted a normal distribution and a one-way analysis of variance (ANOVA) was used to test for significant differences between the three waterbody types sampled in August 2005.

4.3.2 Fish Assemblage Field Collection

Fish assemblage composition was assessed for each site using fyke nets under the ethical clearance of Griffith University Animal Ethics Committee (Protocol Number: AES/01/05/AEC). In each waterbody, three fyke nets were randomly located parallel to the bank in 1-2 m of water and spaced at least 50 m apart to avoid catch interference between neighbouring nets. The steep banks of the storages meant that the average wing width was only 6.5 m. To maintain consistency in sampling effort between natural wetlands and storages, an average wing width of 6.5 m was implemented in the natural wetlands as well. Fyke nets were set in the late afternoon and retrieved as early as possible the next morning with an average set duration of 20 hours. Both wing width and set duration were recorded for the subsequent calculation of catch per unit effort (CPUE).

All fish were identified to species, counted and, with the exception of exotic species, returned to the waterhole after a sample of each species was measured and weighed. Fish counts were standardised to provide data on the CPUE and allow comparison between water bodies. As fyke nets could take up to 1.5 hours per net to process, a greater level of fyke net replication was impractical. In other dryland river studies, two fyke nets and one beach-seine haul were sufficient to determine the total number of species in any waterhole (Arthington *et al.*, 2005). In addition a pilot study trialling sampling with seven fyke nets at a typical storage site, confirmed that the use of two fyke nets would be adequate (Appendix 6). Unfortunately, the use of seine nets to catch smaller fish/individuals was attempted but the steep sides of the banks in some storages made this impractical. Therefore, the assessment of fish assemblage composition was made using three fyke nets per site.

4.3.3 Fish Assemblage Data Analysis

Univariate and multivariate statistics were used to explore differences in fish assemblage composition between sites in natural wetlands and storages. Standardised CPUE data (sum of the three fyke net catches with a wing width of 6.5 m and sampling duration of 20 h) were used to obtain measures of total fish abundance, the abundance of individual species and the total number of species per wetland site. CPUE data were also used to determine patterns of variation in fish assemblage structure between different wetland types. Data were summarised as non-standardised $\log_{10}$(CPUE+1) (absolute abundance), standardised $\log_{10}$(CPUE+1) (relative abundance) and species presence/absence data.

Spatial and temporal patterns in fish assemblage structure were explored using standard multivariate techniques with Bray-Curtis similarity used as the measure of difference between samples. Differences in fish assemblage structure between the three waterbody types were explored using one-way analysis of similarities (ANOSIM) based on the calculated Bray-Curtis similarity matrix. Ordination of this matrix using non-metric multi-dimensional scaling (NMDS) and group average clustering (UPGMA) (PRIMER 5.2.9`; Clarke and Gorley, 2001) provided visual representations of the patterns in absolute abundance, relative abundance and species diversity between different waterbody types. Within PRIMER, default settings were used with 100 random starts for the NMDS (PRIMER 5.2.9`; Clarke and Gorley, 2001). Similarity percentages (SIMPER) were then used to highlight those taxa having a significant contribution to differences between waterbody types.

As assemblage differences are often related to differences in the physical or chemical characteristics of sites, the “biological – environmental” matching routine in PRIMER

(BIO-ENV) (Clarke and Warwick, 2001) was used to investigate relationships between habitat features and morphology variables (Chapter 3), water chemistry, macroinvertebrate assemblages (see Chapter 5) and fish assemblage structure. The BIO-ENV routine calculates a measure of agreement between a fixed biotic similarity matrix (the assemblage data) and each of the possible abiotic similarity matrices (the environmental data) (Clarke and Gorley, 2001). Morphology variables were available only for storage sites; as such, the fish assemblage data for natural wetlands was removed before BIO-ENV analyses with morphology variables were performed. Auto-correlated variables were also removed before the BIO-ENV analysis (Table 4.2). Environmental similarity matrices were based on normalised Euclidean distance, with no transformation of the data. A Spearman rank correlation was used with the maximum number of variables set at five unless otherwise indicated in the results section.

Spatial differences (between waterbody types) in the size of fish populations were explored using one-way ANOVA. Spatial differences in fish standard lengths were conducted using the four species where more than one individual was found in each waterbody type (*Nematolosa erebi*, *Leiopotherapon unicolor*, *Cyprinus carpio* and *Macquaria ambigua*). Standard length data were tested for normality before analysis. Standard length data for *L. unicolor* were normal and differences between waterbodies were tested using a one-way ANOVA. Standard length data for *N. erebi*, *C. carpio* and *M. ambigua* could not be rendered normal via transformation so differences between waterbody types for these species were explored using non-parametric Kruskal-Wallis tests.

Table 4.2: Morphology and water quality variables measured at the storage sites in the Border Rivers Catchment (see Chapter 3 for explanation). Shaded variables were removed prior to BIO-ENV analyses.

Morphology	Water Quality
Age (Y)	pH
Depth (D)	Conductivity
Area (A)	Turbidity
Perimeter (P)	Water temperature
Length (L)	Salinity
Width (W)	
Length:Width (LW)	
Capacity (C)	
Distance from Barwon River (Dis)	
Circularity Index (CI)	
Horton's Form Factor (HFF)	
Elongation Ratio (ER)	

4.4 Results

4.4.1 Water Quality

There were no significant spatial differences in any of the water quality variables (sampled at time 1) between the three waterbody types (p values for one-way ANOVA for each variable > 0.05).

4.4.2 Fish assemblage Spatial Patterns

Fish were collected from all 13 wetland sites during the spatial sampling programme. The fish fauna across all wetland sites comprised 9 different species of which two were exotic, *C. carpio* and *Carassius auratus* (Table 4.3). Eight of these species were present in typical storages and all the fish species that were found in typical storages were also collected from natural wetlands. The only species specific to natural wetlands was *Tandanus tandanus* (Table 4.3). Only five species were collected from ‘no tailwater’ storages. The most common species across all wetland sites was *N. erebi*, which was caught in 12 out of the 13 sites, closely followed by *L. unicolor* (10 out of 13). The reduced diversity in the ‘no tailwater’ storages could be due to the low replication effort.

Species richness per wetland site varied from 2 to 6 species. The most diverse assemblages were recorded in typical storages (4-6 species), while diversity in natural wetlands and ‘no tailwater’ storages was similar with 2-5 species and 3-5 species respectively.

Table 4.3: Fish fauna of natural wetlands and storages in the Border Rivers Catchment. Exotic species are marked with an asterisk. NW = natural wetland, TW = ‘no tailwater’ storage, TS = typical storage.

		Mean CPUE across wetland type					
	No. of sites with species present	NW		TW		TS	
Species	(max. 13)	mean	s.e.	mean	s.e.	mean	s.e.
Retropinna semoni	3	0.2	0.2			1.3	1.0
Nematolosa erebi	12	5.3	1.8	80.2	74.8	281.9	136.6
Hypseleotris spp.	3	6.3	5.2			0.2	0.2
Macquaria ambigua	9	3.2	1.3	10.3	6.8	16.0	8.8
Melanotaenia fluviatilis	2	0.2	0.2			0.4	0.4
Leiopotherapon unicolor	10	2.4	1.5	2.3	0.6	19.1	9.9
Tandanus tandanus	1	0.2	0.2			0.0	0.0
*Cyprinus carpio**	9	10.7	5.0	6.7	6.1	4.0	1.2
*Carassius auratus**	6	3.0	1.8	0.4	0.4	1.6	1.1

Total CPUE (fish per waterhole) varied from 11 at one natural wetland to 809 at one typical storage with typical storages having greater numbers of fish than natural wetlands. In typical storages, total CPUE varied from 83 to 809 individuals with an overall total of 1723 across all typical storage sites and average catch size of 345. For natural wetlands, catch per site varied from 11 to 45 with an overall total of only 158 and average catch size of 32. 'No tailwater' storages showed patterns in between typical storages and natural wetlands with catch sizes varying from 12 to 258 and an overall total of 299 and average catch size of 100.

Ordination (Fig. 4.2) based on relative abundance of each species suggested significant spatial variation in fish assemblage structure across the 13 wetland sites (ordination plots for presence/absence data and absolute abundance show a similar pattern; see Appendix 7). Both types of storages were more tightly grouped in ordination space than natural wetlands (Fig. 4.2); suggesting storages have a narrower range of variability than natural wetlands. Cluster analysis of waterbody sites based on absolute abundance data of fish across all sampling occasions further illustrates the similarity of both types of storages and the variation within natural wetlands (Fig. 4.3). UPGMA of CPUE data suggested 6 meaningful groups truncated at a similarity of 60 (Table 4.4 and Fig. 4.3).

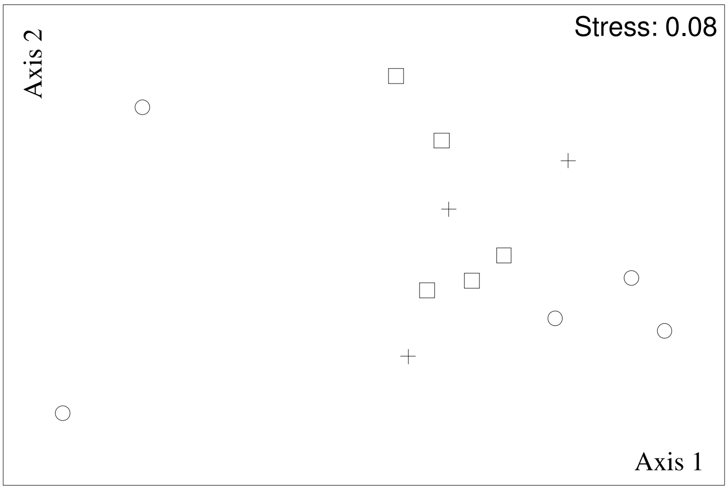


Fig. 4.2: Non-metric multi-dimensional scaling plot for fish assemblage structure based on $Log_{10}(CPUE+1)$ standardised for relative abundance. O Natural wetlands, + Storages without tailwater, □ Typical storages.

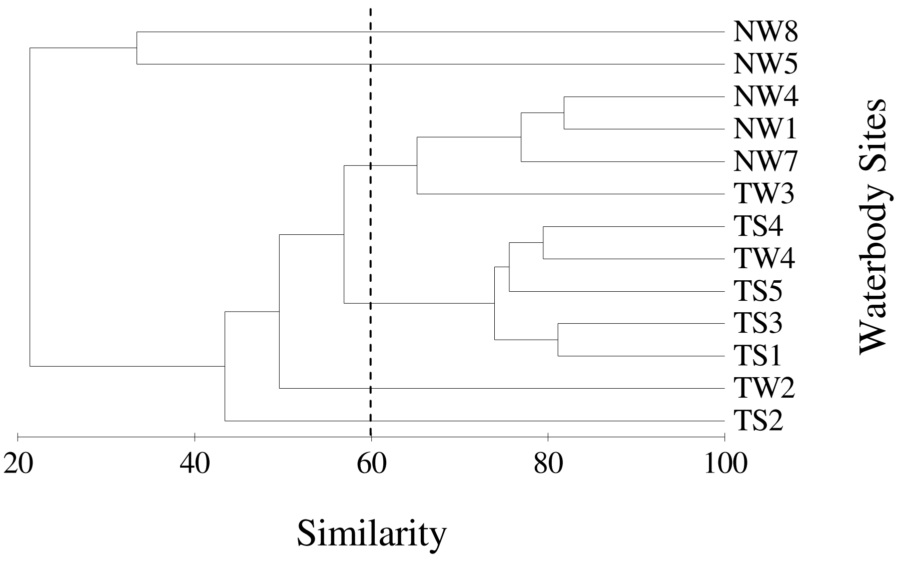


Fig. 4.3: Dendrogram of waterbody types based on absolute abundance (Log_{10}(CPUE+1)) data of fish across all sampling occasions. NW = natural wetland, TW = ‘no tailwater’ storage, TS = typical storage.

Table 4.4: Fish catches at natural wetlands and storage sites in the Border Rivers Catchment based on total CPUE. NW = natural wetland, TW = ‘no tailwater’ storage, TS = typical storage

Cluster Groups	**3**	**3**	**2**	**3**	**1**	**5**	**3**	**4**	**4**	**6**	**4**	**4**	**4**
Species	**NW1**	**NW4**	**NW5**	**NW7**	**NW8**	**TW2**	**TW3**	**TW4**	**TS1**	**TS2**	**TS3**	**TS4**	**TS5**
C. carpio	11	16	0	27	0	0	19	1	5	0	7	2	6
N. erebi	8	10	6	3	0	2	9	230	31	363	58	183	776
L. unicolor	6	0	0	0	6	2	1	3	1	5	12	56	21
M. ambigua	6	5	0	6	0	8	0	23	21	0	7	48	4
C. auratus	2	2	0	10	0	0	0	1	0	5	0	0	3
R. semoni	0	0	1	0	0	0	0	0	0	5	0	1	0
Hypseleotris spp.	0	0	27	0	5	0	0	0	0	1	0	0	0
M. fluviatilis	0	0	1	0	0	0	0	0	0	2	0	0	0
T. tandanus	0	1	0	0	0	0	0	0	0	0	0	0	0
Total Species	5	5	4	4	2	3	3	5	4	6	4	5	5
Total Catch Size	33	34	35	45	11	12	29	258	58	381	83	291	809

Table 4.5: Summary of ANOSIM results comparing fish assemblage structure among wetland groups.

	Transformation	**Global R**	***P***
Species Diversity	Presence/Absence	-0.032	0.535.
Absolute Abundance	Log_{10}(CPUE+1)	0.142	0.106
Relative Abundance	Log_{10}(CPUE+1)	0.105	0.163

Table 4.6: R value of pairwise tests from ANOSIM results comparing fish assemblage structure between natural wetlands (NW), ‘no tailwater’ storages (TW) and typical storages (TS).

Pairwise Test	**Presence/Absence**	**Absolute Abundance**	**Relative Abundance**
NW v TW	-0.179	-0.097	-0.087
NW v TS	0.006	0.248	0.248
TW v TS	-0.056	0.149	-0.138

No significant differences in absolute abundance, relative abundance or presence absence data were detected using a one-way ANOSIM (Table 4.5 and Appendix 8). However, wetland groups were more dissimilar in relation to abundance data compared with species diversity (Table 4.5 and Appendix 8). It was also apparent that natural wetlands and 'no tailwater' storages were barely separable with respect to abundance (R value very close to zero) while natural wetlands and typical storages were the least similar (R value = 0.248) (Table 4.6). In contrast the opposite was true for species diversity.

The two exotic species contributed over 43% of total CPUE across all natural wetlands (Fig. 4.4). This contrasts markedly with both types of storage where the contribution of exotic species was less than 2% in typical storages and only 7% in 'no tailwater' storages. The catch in both types of storages was dominated by *N. erebi*, (Fig. 4.4) contributing 87% of the total CPUE in typical storages and 80% in 'no tailwater' storages. Not only did *N. erebi* make up a greater percentage of the catch at storage sites but catches of *N. erebi* were also much higher at storages than natural wetlands (Table 4.3). The most common species in natural wetlands was *C. carpio* (Fig. 4.6), which contributed 34% of the total CPUE. In contrast to both types of storages, natural wetlands showed a more even distribution of species (Fig. 4.4).

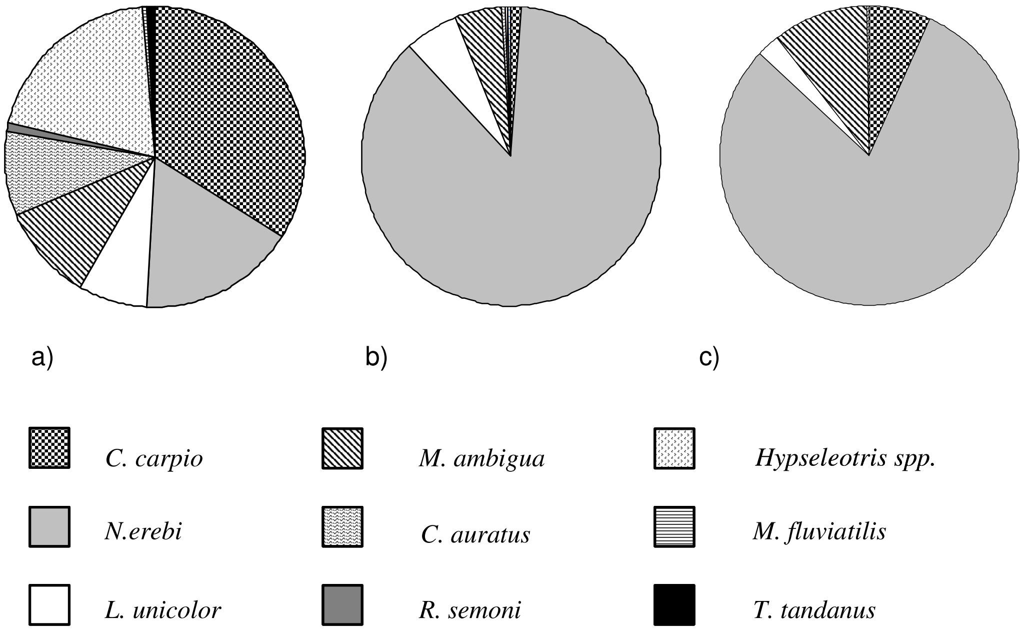


Fig. 4.4: Percentage contribution of each fish species to the total CPUE for each wetland group; a) natural wetlands, b) typical storages and c) storages without tailwater.

SIMPER suggested average dissimilarity was greatest between natural wetlands and typical storages (58%) (Table 4.7). As expected, *N. erebi* contributed most to the differences (31%) with this species more abundant in typical storages. The highest average similarity within sites was observed for typical storages (65%), with *N. erebi* contributing most to this similarity (51%). This lends support to previous evidence that there was more variability within natural wetlands than within typical storages.

Table 4.7: Summary of SIMPER results for within site similarity and between site dissimilarity based on $\log_{10}$(CPUE+1) data with a cut off for low contributions of 90%.

	Natural Wetland		**No Tailwater**		**Typical**	
Average Similarity	36.27		45.96		65.37	
	Species	%	Species	%	Species	%
	N. erebi	32.57	*N. erebi*	45.16	*N. erebi*	51.28
	C. carpio	23.27	*L. unicolor*	28.77	*L. unicolor*	19.45
	M. ambigua	16.12	*M. ambigua*	19.58	*M. ambigua*	15.11
	C. auratus	10.84			*C. carpio*	11.76
	Hypseleotris spp.	9.23				
Average Dissimilarity	**NW v TW**	56.87	**NW v TS**	57.58	**TW v TS**	43.58
	Species	%	Species	%	Species	%
	N. erebi	20.59	*N. erebi*	30.55	*N. erebi*	31.67
	C. carpio	19.06	*L. unicolor*	15.73	*M. ambigua*	17.95
	M. ambigua	17.19	*M. ambigua*	13.92	*L. unicolor*	16.65
	Hypseleotris spp.	14.52	*C. carpio*	12.78	*C. carpio*	16.02
	L. unicolor	12.48	*Hypseleotris spp.*	10	*C. auratus*	7.64
					R. semoni	5.83

There was only one case of association between the environmental data and observed fish assemblage structure (Table 4.8). Fish assemblage structure based on presence/absence data was related to the morphology variables, depth, width and length:width (Spearman rank correlations $r_s > 0.5$). The r_s values for all other variables were low (< 0.5). However, BIO-ENV does suggest that macroinvertebrate assemblages may be driving the fish assemblage structure within waterholes. *Branchinella* sp., *Paratya* sp. and tiny chironomids explained 71% of the variation in fish species presence/absence patterns. *Branchinella* sp., *Cryptochironomus* sp. and *Cladotanytarsus* sp. accounted for 62% and 69% of variation in fish assemblage structure based on absolute abundance and relative abundance data respectively (Table 4.8).

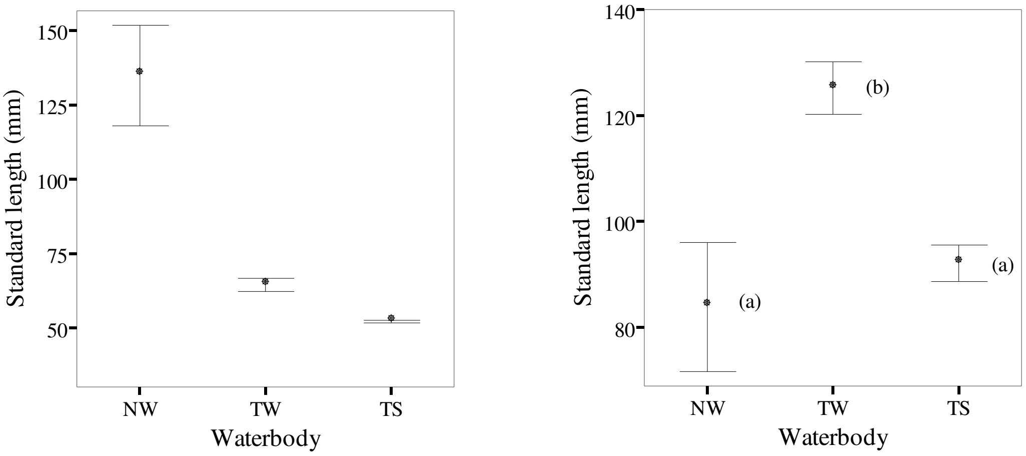
Table 4.8: Summary of BIO-ENV results based on Spearman rank correlations (r_s) between fish assemblage structure, habitat availability variables, water chemistry variables, morphology variables and macroinvertebrate data. Results presented for best possible solution only.

	Transformation	Best Variable Combination	Correlation
Habitat Availability Variables			
Species Diversity	Presence/absence	SN	0.200
Absolute Abundance	$\text{Log}_{10}(x+1)$	SN	0.273
Relative Abundance	$\text{Log}_{10}(x+1)$	SN	0.419
Water Chemistry Variables			
Species Diversity	Presence/absence	pH, conductivity, turbidity	0.338
Absolute Abundance	$\text{Log}_{10}(x+1)$	pH, conductivity	0.322
Relative Abundance	$\text{Log}_{10}(x+1)$	pH, conductivity	0.354
Morphology Variables*			
Species Diversity	Presence/absence	Depth, width, L:W	0.650
Absolute Abundance	$\text{Log}_{10}(x+1)$	Depth, width, L:W, HFF	0.430
Relative Abundance	$\text{Log}_{10}(x+1)$	Depth, L:W., HFF	0.374
Macroinvertebrate Assemblages**			
Species Diversity	Presence/absence	1, 4, 54	0.841
Absolute Abundance	$\text{Log}_{10}(x+1)$	1, 28, 37	0.790
Relative Abundance	$\text{Log}_{10}(x+1)$	1, 28, 37	0.831

SN = snags, 1 = *Branchinella* sp., 4 = *Paratya* sp., 54 = Tiny Chironomids, 28 = *Cryptochironomus* sp., 37 = *Cladotanytarsus* sp. *Only includes storage sites. **Maximum number of variables was set at 3.

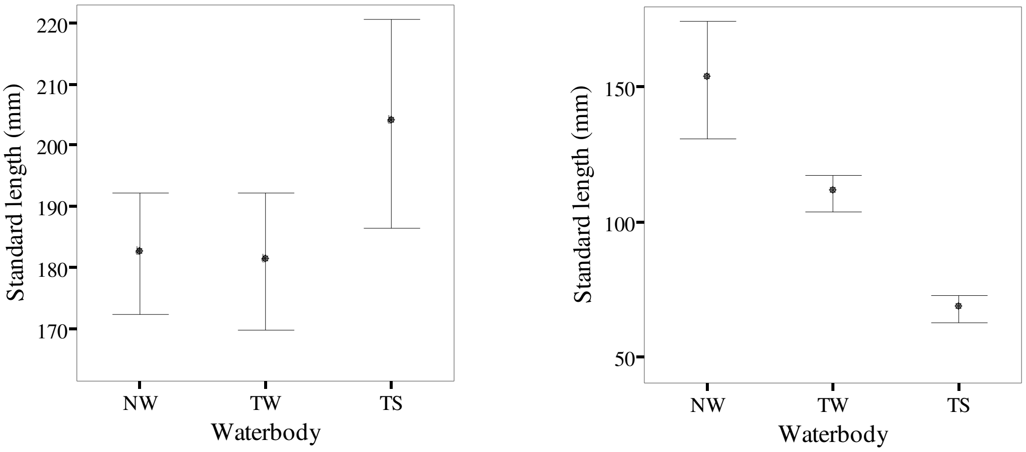
4.4.3 Fish standard lengths

Fish mean standard lengths varied between sites for the different taxa (Appendix 9). There was no significant difference between different waterbody types ($p > 0.05$) for *C. carpio;* however, there was for *N. erebi* ($p < 0.001$), *M. ambigua* ($p < 0.001$) and *L. unicolor* ($F_{2,93} = 3.692$, $p < 0.05$). Post hoc tests suggested mean standard length of *L. unicolor* was higher at 'no tailwater' sites compared to the two other waterbody types (Fig. 4.5). Mean standard lengths for *N. erebi* and *M. ambigua* were highest in natural wetlands followed by 'no tailwater' storages and typical storages in that order (Fig. 4.5).



a) *N. erebi*

b) *L. unicolor*



c) *C. carpio*

d) *M. ambigua*

Fig. 4.5: Mean (±s.e.) standard length (mm) of four fish species at three different waterbody types collected from the Border Rivers Catchment between August 2005 and October 2005. Groups detected by post-hoc tests shown in brackets. NW = natural wetland, TW = 'no tailwater' storage, TS = typical storage.

The small number of individuals caught in particular waterbodies makes it difficult to interpret size distributions. However, there were some peculiarities worth noting. The size distribution of all taxa in natural wetlands followed a normal distribution, apart from *C. carpio*, which was positively skewed (Fig. 4.6). In contrast to this, *C. carpio* displayed a normal distribution at typical storages while the other three species were right skewed. The size distribution of *N. erebi* was positively skewed at both storage types with large numbers of small fish between 25 and 60 mm in length (Fig. 4.6). The size distribution of *C. carpio* was also positively skewed at 'no tailwater' storages while, *M. ambigua* and *L. unicolor* exhibited a normal distribution. If the high number of small *N. erebi* found in storages was due to only small fish being pumped from the river, you would expect the same size distributions for all species in the storages.

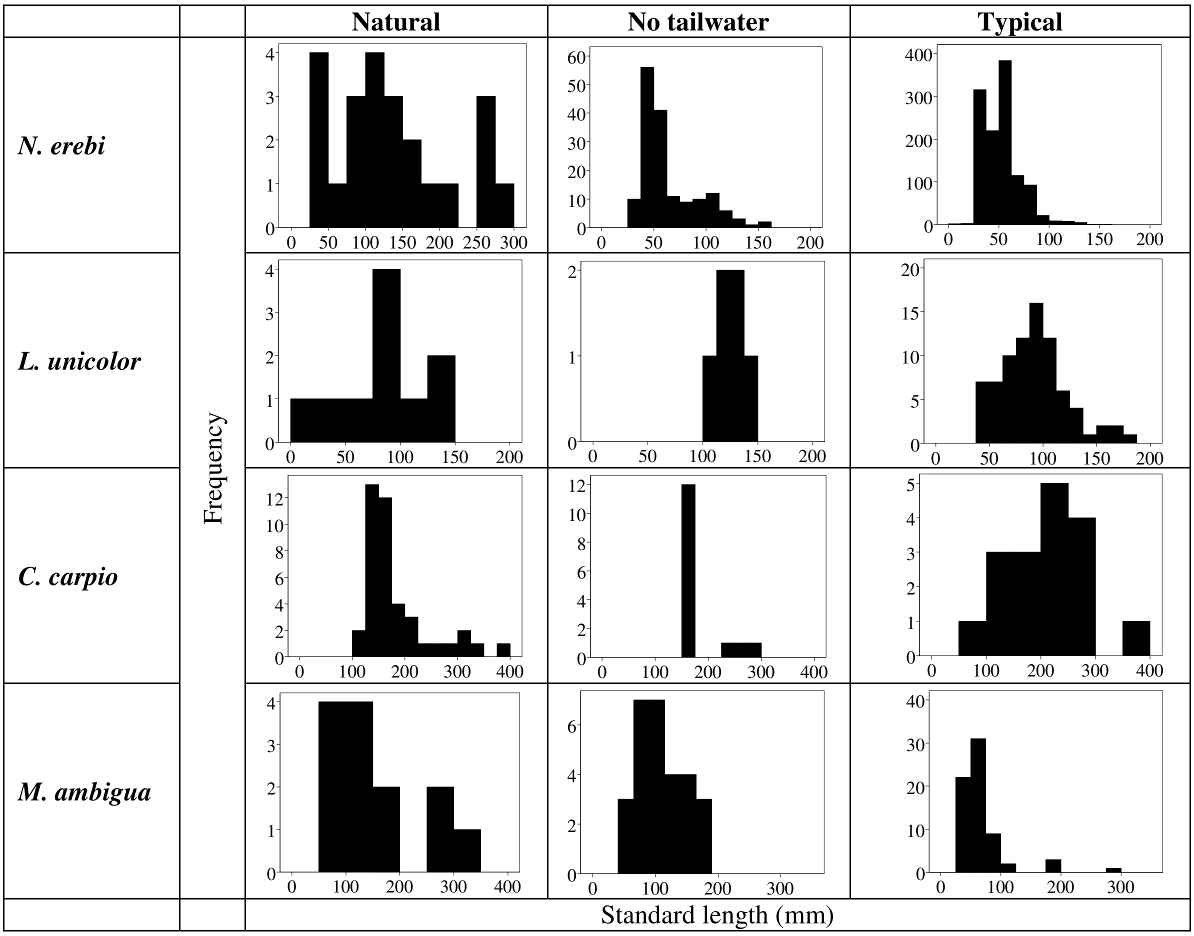


Fig. 4.6: Frequency of occurrence histograms in relation to the standard length (mm) of four different species of fish common to the three waterbody types.

4.5 Discussion

A number of factors may potentially influence the fish community of storages (Fig. 4.7), namely; the fish community of the river from which water is extracted (Source Community), the ability of species to colonise storages determined by their behaviour and survival rate during extraction (Connectivity) and the environment within the storage which will affect fish survival, recruitment and growth (Habitat).

Fourteen species of native fish and three species of exotic fish are found in the Border Rivers Catchment (Moffatt and Voller, 2002). Only seven native species and two exotic species were collected during this study. Moffatt and Voller (2002) divided the Queensland Murray Darling Basin into four zones; the upland zone in the east, the upper foothill zone, the lower foothill zone and the lowland zone to the far west of the basin. The Border Rivers Catchment covers three of these zones, the upland zone east of Stanthorpe, the upper foothill zone between Stanthorpe and Texas and the lower foothill zone west of Texas. As the sampling for this study took place in the lower foothill zone only, it is to be expected that species more suited to other zones would not be recorded; for example the river blackfish, *Gadopsis marmoratus* and mountain galaxias, *Galaxias olidus* are only found in streams of the upland zone (Moffatt and Voller, 2002). The reduced diversity may also have been due to the low number of sites sampled, and because each site was only sampled once.

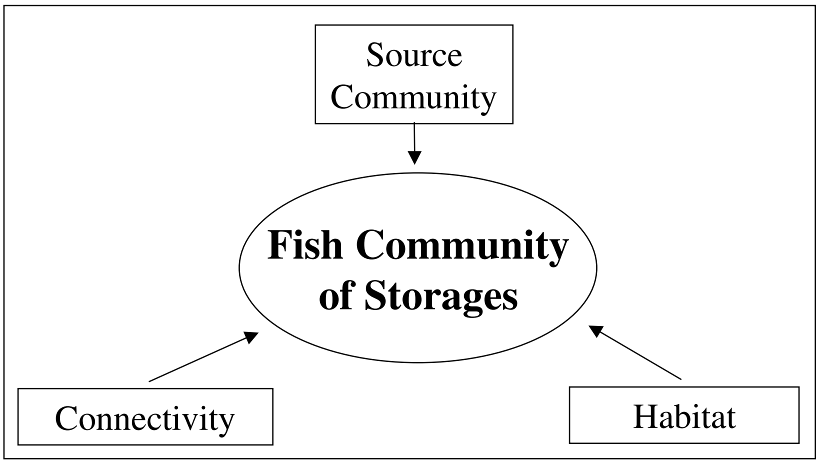


Fig. 4.7: Conceptual model depicting potential factors influencing the fish community of storages.

Only one individual of *T. tandanus* from a natural wetland site was recorded during the study. This was expected given that numbers of *T. tandanus* have undergone a dramatic decline in riverine habitats in the Murray Darling Basin (Moffatt and Voller, 2002). The Fisheries Scientific Committee (2008) recently proposed that *T. tandanus* in the Murray-Darling Basin should be listed as an endangered population. The low numbers of *R. semoni*, *Hypseleotris* spp., *M. fluviatilis* and *C. auratus* found in storages reflect the low numbers recorded in natural wetlands. *N. erebi* was the most common species found in 12 out of the 13 sampling sites. This was not surprising as *N. erebi* is widespread in central, eastern and northern Australia (Allen *et al.*, 2002; Pusey *et al.*, 2004) and is one of the few fish species that may have become more abundant since European settlement of Australia (Moffatt and Voller, 2002).

In contrast to our hypothesis, fish diversity was similar between waterbody types, suggesting artificial storages provide adequate habitat for a range of fish taxa. However, there was one marked difference, both types of storages displayed reduced variation in fish assemblage structure between sites in comparison to natural wetlands. Assemblage variation between sites is a common feature of dryland rivers (see Arthington *et al.*, 2005; Balcombe *et al.*, 2006) and may offer some level of assemblage resilience at larger spatial scales. The relationship identified between fish community composition and morphology variables suggests that increased diversity in the structure of storages could improve the variation in assemblage composition between storage sites. Spatial patterns in fish abundances were the opposite of what was predicted, with catch sizes much larger in storages (58 to 809) compared to natural wetlands (11 to 45). Catch sizes in natural wetlands were also very low compared to those from natural waterholes in the Cooper Creek system, Lake Eyre Basin (84 to 46,591) (Arthington *et al.*, 2005), but only slightly lower than those recorded from other Murray-Darling Basin systems; Warrego River (Balcombe *et al.*, 2006) (~10 to 300) and the Border Rivers (Balcombe *et al.*, In press)(6 to 455). However, it is difficult to compare abundances between studies due to variation in sampling times and methods.

There were also obvious differences in the assemblage composition of storages compared with natural wetlands, notably, the low numbers of exotic species in storages and the more even distribution of species in natural wetlands. Interestingly, bony bream (*Nematalosa erebi)* dominated the abundance in storages (~80%) while accounting for less than 30% of the abundance in natural wetlands. *N. erebi* has been found to dominate the assemblage composition of natural wetlands elsewhere, accounting for 40% of the total catch from 15 waterbodies sampled in the Warrego system in 2001-2002 (Balcombe *et al.*, 2005) and 68% of the catch from 15 sites sampled within the Border Rivers, Murray-Darling Basin, in 2003 (Balcombe *et al.*, In press). Although not as abundant as *N. erebi*, there were higher numbers of *L. unicolor* and *M. ambigua* in storages compared with natural wetlands. The perceived preference of certain species for storages from my study may actually be because the 'rare' species are less susceptible to the river pumps. Every stage of the life cycle of fish can potentially be extracted through irrigation pumps, including the eggs, larvae and adult fish. Variation in behaviour of all these stages may increase or decrease the likelihood of extraction of different species.

T. tandanus deposits demersal, non-adhesive eggs in a nest made of coarse sand or gravel, which is then guarded by the male (Pusey *et al.*, 2004). Once they leave the parental 'nest', larvae of *T. tandanus* are estimated to spend between 5-7 days drifting in the river currents (Humphries, 2005; Rowland, 2005). The enclosure of eggs within a nest along with the short drifting period may help reduce the number of eggs entering the irrigation system. In comparison female, *M. ambigua* lay semi-buoyant, non adhesive, pelagic eggs which drift with the current for their short incubation period (Pusey *et al.*, 2004). After hatching and for up to 25 days the larvae are unable to swim freely and continue to drift in the current (Gehrke, 1990) making them more susceptible to pumping than *T. tandanus*. However, adult *M. ambigua* remain relatively stationary, preferring habitats with deep, slow flowing water (Battaglene and Callanan, 1991) and have a preference for woody debris, macrophytes and overhead canopy cover (Crook *et al.*, 2001), but do become more active during times of flow (Moffatt and Voller, 2002; Pusey *et al.*, 2004). This suggests that eggs and larvae of *M. ambigua* may be more at risk of extraction than adults, contributing to the large numbers of small *M. ambigua* in storages.

L. unicolor is commonly found nearby some form of cover, predominantly woody debris, root masses and aquatic macrophytes (Kennard, 1995), in the bottom two thirds of the water column (Pusey *et al.*, 2004). Eggs are non-adhesive but they are demersal (Pusey *et al.*, 2004) which may reduce the number transported by the pumping process. In contrast, *N. erebi* has not been associated with microhabitat cover such as woody debris and aquatic vegetation and although they are most frequently recorded in the lower one-third, they are observed throughout the water column (Pusey *et al.*, 2004). Females spawn in shallow still-water habitats producing eggs that are demersal at time of spawning but later become buoyant (Puckridge and Walker, 1990). These behavioural traits may contribute to the abundance of *N. erebi* in storages.

Three categories of larval drift have been identified for Murray-Darling Basin fish; obligate drifters for which downstream transport is necessary for survival, facultative drifters for which drifting serves only as a means of dispersal and non-drifters which do not drift unless they are caught in strong currents (Humphries and King, 2003). It is therefore likely that species which are obligate or facultative drifters (*M. peelii*, *M. ambigua*, *N. erebi* and *C. carpio*) will be more common in irrigation systems (Gilligan and Schiller, 2003).

Another factor affecting numbers and species of fish extracted may be the time of extraction. The timing of spawning, larval drift and development times are likely to influence the susceptibility of different species to entrainment into irrigation channels. In southeast Queensland many native species of fish spawn during spring and summer (*M. peelii*, *M. ambigua*, *N. erebi*, *L. unicolor* and *T. tandanus*) believed to be triggered by rising water temperatures and increasing photo-period, (Pusey *et al.*, 2004; Humphries, 2005; Rowland, 2005). It has also been well documented that species of native fish in the Murray-Darling Basin exhibit migrations at specific times of the year (e.g. Mallen-Cooper, 1996; Baumgartner, 2004). Humphries and King (2003) reported that the majority of larval drift occurred between August and March. Unfortunately both these activities (spawning and larval drift) coincide with the peak irrigation period in the catchment making it unlikely that extraction could be avoided at these times.

An interesting discovery was the lower numbers of exotic species in storages in comparison to natural wetlands. The adults of *C. carpio* favour areas adjacent to aquatic vegetation but also snags and other cover. Adhesive eggs are scattered by spawning adults and become attached to aquatic vegetation (Moffatt and Voller, 2002). In comparison to the larvae of many native species, which are small and easily transported by the current, larvae of *C. carpio* are large when they emerge and may not be carried by the current as easily (S. Rowland pers. comm.). *C. carpio* are also relatively slow swimmers (S. Rowland pers. comm.) so may be more susceptible to predation as the water flows along open channels to the storage dam. The combined influence of these characteristics may be the reason for the reduced abundance of *C. carpio* in storages.

A second theory is that all species are equally likely to be extracted from the river channel but that some have a reduced survival rate during or immediately after the pumping process. Blackley (2003) suggested that native fish, particularly juveniles have a tendency to follow the natural flow of the waterway. It is fair to assume that this will hold true even if the flow is into an irrigation offtake. It is probable that a number of fish do not survive the extraction process as a direct result of being killed by the actual pump or due to the stress of the process (see Chapter 6).

A final theory is that if all species are equally likely to be extracted and survive the pumping process, the greater abundance of certain species of fish is because some species thrive in storages while others find the conditions unsuitable. As discussed there may be a number of reasons for the high numbers of *N. erebi* found in storages. Their wide distribution also suggests *N. erebi* are highly tolerant of a wide range of environmental conditions (Pusey *et al.*, 2004). In this study, the water quality of storages was similar to that of natural wetlands, suggesting water quality would not be limiting their distribution in either habitat. A distinct difference between *N. erebi* in natural wetlands and storages in this study was the dominance of small fish within the storages, compared to a more even distribution of size classes in the wetlands. If the dominance of small *N. erebi* in storages were due to a size selection survival through the pumps, you would expect the same size distribution for all species of fish in storages – i.e only small individuals. As this was not the case, with large individuals of both *L. unicolor* and *C. carpio* collected, it is possible that a small number of mature adults of *N. erebi* have been pumped into storages and begun to breed. *N. erebi* do not require a flood to initiate spawning (Puckridge and Walker, 1990) and they mature early, usually in their first or second year (Pusey *et al.*, 2004). *N. erebi* populations are also capable of rapid recovery after unfavourable conditions due in part to their high fecundity, up to 880,000 eggs per individual (Puckridge and Walker, 1990), suggesting low numbers of sexually mature adults could be responsible for the high numbers of *N. erebi* in storages. Puckridge and Walker (1990) found sexually mature females as small as 150 mm total length (TL) and males of 130 mm TL. At the storage sites fish of up to 161 mm standard length (SL) were recorded making it very likely that sexually mature adults are present in storages. The higher abundance of predatory taxa may help control the growth of such large populations of *N. erebi* in natural wetlands.

L. unicolor also reach sexual maturity within their first year but have lower fecundity, up to 113,200 eggs (Puckridge and Walker, 1990). Similar to *N. erebi* there were high numbers of small *M. ambigua* in the storages. However, it is unlikely that *M. ambigua* are breeding in the storages; although the males of *M. ambigua* are mature from 2-3 years, the females are not mature until 4 years (Pusey *et al.*, 2004). Anecdotal evidence suggests that many storages in the catchment will be dry one in every three years, and more frequently during a drought, reducing the likelihood that fish with delayed development times will breed in these systems.

The fish communities of storages may also be driven by food availability. In Australian freshwaters, aquatic insects are the most important food resource for fish species (Kennard *et al.*, 2001). It is not surprising then that macroinvertebrate assemblages (see Chapter 5) explained approximately 70% of the variation in fish diversity and abundance within storages.

4.5.1 Conclusions

In summary, my data suggest that artificial storages support the same richness and higher biomass of fish compared with natural wetlands, but the abundance is dominated by one species *N. erebi* which has life history strategies and tolerances that allow it to proliferate in these artificial environments. The lower evenness of the assemblage in the artificial storages suggests these environments are unable to support the same assemblage composition as natural wetlands and do not replace natural wetlands as viable aquatic habitat for these floodplain rivers. However, alterations to increase the structural diversity of storages across the landscape may improve the variation in fish assemblage composition between sites.

Chapter 5 Spatial and temporal variation of macroinvertebrate and zooplankton assemblages of three different waterbody types in the Border Rivers Catchment

5.1 Introduction

In dryland river systems, and their associated wetlands, hydrological variability is seen as a major contributor to the variation in aquatic diversity in space and time (Walker *et al.*, 1997; Sheldon *et al.*, 2002; Leigh and Sheldon, 2009). The floodplain habitats of Australian dryland rivers vary between times of high productivity and reproduction associated with immense flooding (Kingsford, 2000a; Puckridge *et al.*, 2000; Shiel *et al.*, 2006) and periods of low productivity associated with droughts; the 'boom' and the 'bust' (Walker *et al.*, 1997). The associated temporal variation in aquatic assemblage composition and structure is well documented for arid regions of Australia (Arthington *et al.*, 2005; Marshall *et al.*, 2006; Shiel *et al.*, 2006); with the change in assemblage structure during the dry (no flow) phase linked to a reduction in habitat diversity as the floodplain dries and structural habitat elements, such as bars and boulders, are exposed (Arthington *et al.*, 2005; Leigh and Sheldon, 2009). In comparison, interactions between species, such as competition, parasitism and predation, are more likely to be driving the temporal variations of aquatic assemblages in lakes (Jackson *et al.*, 2001; Tonn *et al.*, 2004).

While the value of natural wetlands for maintaining aquatic diversity is well recognised (Semlitsch and Bodie, 1998; Kingsford, 2000b; Dudgeon *et al.*, 2006), artificial wetlands may also provide a valuable environment for a range of aquatic species (Hazell *et al.*, 2004) including waterbirds (Hazell *et al.*, 2004) and frogs (Hazell *et al.*, 2001; Knutson *et al.*, 2004). In England man-made ditches support uncommon species of macroinvertebrates and macrophytes (Williams *et al.*, 2003), while a wide range of native aquatic flora and fauna have been described from artificial farm ponds (Markwell and Fellows, 2008), with many of these environments recognised for their high species richness of aquatic vegetation (Linton and Goulder, 2000). In this way artificial wetlands may provide alternative habitat for aquatic assemblages, and be particularly important where natural aquatic habitats may have been destroyed. In southeast Queensland, the invertebrate communities recorded in constructed farm ponds were comparable to surrounding natural streams (Markwell and Fellows, 2008).

The physical isolation of many artificial storages may also contribute to their differing aquatic faunal community. Many aquatic invertebrates have an adult life stage which can readily disperse between suitable habitats; however, many of these restrict their dispersal to the riparian corridor where humidity is relatively high (Lynch *et al.*, 2002). In dryland rivers there appear to be three major types of dispersal mode for aquatic fauna: highly mobile organisms who disperse readily between isolated waterbodies without the need for aquatic connection (eg. beetles and hemipterans), mobile organisms which disperse readily between habitats when connected by flow, and organisms with very limited dispersal ability even during periods of aquatic connection (Sheldon *et al.*, In press). To explore the role of physical isolation of storages in structuring the faunal assemblage, aquatic macroinvertebrates can be assigned to different functional mobility groups (FMG's) depending on their ability to move between wetland habitats. Mobility allows animals such as *Anisops* sp. to colonise ephemeral areas as they flood and return to permanent waterbodies as flood waters recede (Jeffries, 1996). It is therefore probable, that due to the lack of habitat and riparian vegetation at storage sites, highly mobile taxa that are pumped into storages will relocate to nearby natural wetlands leaving only low mobility taxa within storages.

5.1.1 Aims

The aim of this chapter was to explore spatial and temporal patterns in water quality and aquatic assemblage composition of artificial storages and natural wetlands. Water quality and macroinvertebrate assemblage composition data were collected from typical storages, 'no tailwater' storages and natural wetlands in the Border Rivers Catchment between the 2005 and 2007. The specific objectives of this chapter are:

1) To explore the spatial variation in structure (taxonomic abundances of macroinvertebrates and zooplankton) and function (functional mobility group proportions of macroinvertebrates) of aquatic assemblages of wetlands and storages in the Border Rivers Catchment.

2) To explore the temporal variation in structure and function of aquatic assemblages of wetlands and storages in the Border Rivers Catchment.
3) To assess the health of the sampled waterbodies based on commonly used macroinvertebrate indicators of ecosystem health.

5.2 Methods

5.2.1 Study Area and Design

To investigate spatial patterns in the data, macroinvertebrate and water quality data were collected from three wetlands ‘types’; natural wetlands, typical storages and storages with the same morphological characteristics as the ‘typical’ category but without tailwater inputs (‘no tailwater’, see Chapter 3). The storages and natural wetlands were selected such that sampling sites were spread throughout the Border Rivers Catchment (Fig. 5.1). Sampling was carried out between August and October 2005, over an eight week period (T1). In total, five typical storages, eight natural wetlands and four ‘no tailwater’ storages were selected for sampling (Table 5.1).

To investigate temporal patterns in wetland invertebrate community structure, macroinvertebrate and zooplankton samples were taken during the 2005/06 and 2006/07 irrigation seasons in relation to changes in water levels. Macroinvertebrates were sampled 6 times (T2-T7) while zooplankton was only collected on five occasions, February 2006 to February 2007 (T3-T7) (Table 5.2). Five typical storages and five natural wetlands were selected within the Talwood area of the Border Rivers Catchment for this temporal study (Fig. 5.2). Over the course of the sampling some of these waterbodies dried up, therefore not all sites were sampled six times (Table 5.2). Only three sites were sampled in November 2005 due to rainfall in the area. Gaining access to Neilo Double Bay Dam was often problematic; hence it was only sampled three times.

The data collected allowed me to explore spatial differences in macroinvertebrate assemblages at time 1 (spatial sampling) between three waterbody types: natural wetlands, typical storages and ‘no tailwater’ storages. Using the data from time 2 to time 7 (temporal sampling) spatial patterns in macroinvertebrate and zooplankton assemblages were also explored between natural wetlands and typical storages. This same data set was used to investigate temporal patterns in macroinvertebrate and zooplankton assemblages across the sampling period. Spatial and temporal patterns in the data were explored using both univariate and multivariate statistical techniques in the SPSS 15.0 (SPSS, 2006) and PRIMER v.5 (Clarke and Gorley, 2001) software packages. Finally the macroinvertebrate data was used to examine how waterbody health compared between the three waterbody types

Table 5.1: Sites sampled in the Border Rivers Catchment during the spatial study (Time 1) from August 2005 to October 2005, NW = natural wetland, TW = 'no tailwater' storage, TS = typical storage.

Site	**Site Code**	**Macroinvertebrates**
Natural Wetlands		
Crawler Lagoon	NW1	✓
Gil Gil Creek	NW2	✓
Barra Lagoon	NW3	✓
Newinga Weir Waterhole	NW4	✓
Punbougal Lagoon	NW5	✓
South Callandoon Lagoon	NW6	✓
Tarrawatta Weir Waterhole	NW7	✓
Yambocully Lagoon*	NW8	✓
Tailwater Storages		
Barra S2	TW1	✓
Boolarwell S2	TW2	✓
Newinga S2	TW3	✓
Tarrawatta Weir Dam*	TW4	✓
Typical Storages		
Jericho S1	TS1	✓
Royston S3	TS2	✓
South Callandoon S2	TS3	✓
Teriadi S2	TS4	✓
Undabri S2	TS5	✓

Table 5.2: Sites and dates sampled in the Border Rivers Catchment during the temporal study from November 2005 to February 2007.

		2005-2006 Irrigation Season			**2006-2007 Irrigation Season**		
		22/11/05	**03/02/06**	**08/03/06**	**07/12/06**	**08/01/07**	**06/02/07**
Natural Wetlands	**Time**	**2**	**3**	**4**	**5**	**6**	**7**
Crawler Lagoon	NW1		✓	✓	✓	✓	✓
Newinga Weir Waterhole	NW4		✓	✓	✓	✓	✓
Tarrawatta Weir Waterhole	NW7	✓	✓	✓	Dry	Dry	Dry
Kerry's Creek	NW9		✓	✓	Dry	Dry	Dry
Warril Creek Talwood	NW10		✓	✓	Dry	Dry	Dry
Typical Storages							
Jericho S1	TS1		✓	✓	✓	✓	Dry
Yattlewondi S4	TS6		✓	✓	✓	✓	✓
Tarrawatta N End Dam	TS7	✓	✓	✓	✓	✓	Dry
Boolarwell S1	TS8	✓	✓	✓	Dry	Dry	Dry
Neilo Double Bay Dam	TS9			✓		✓	✓

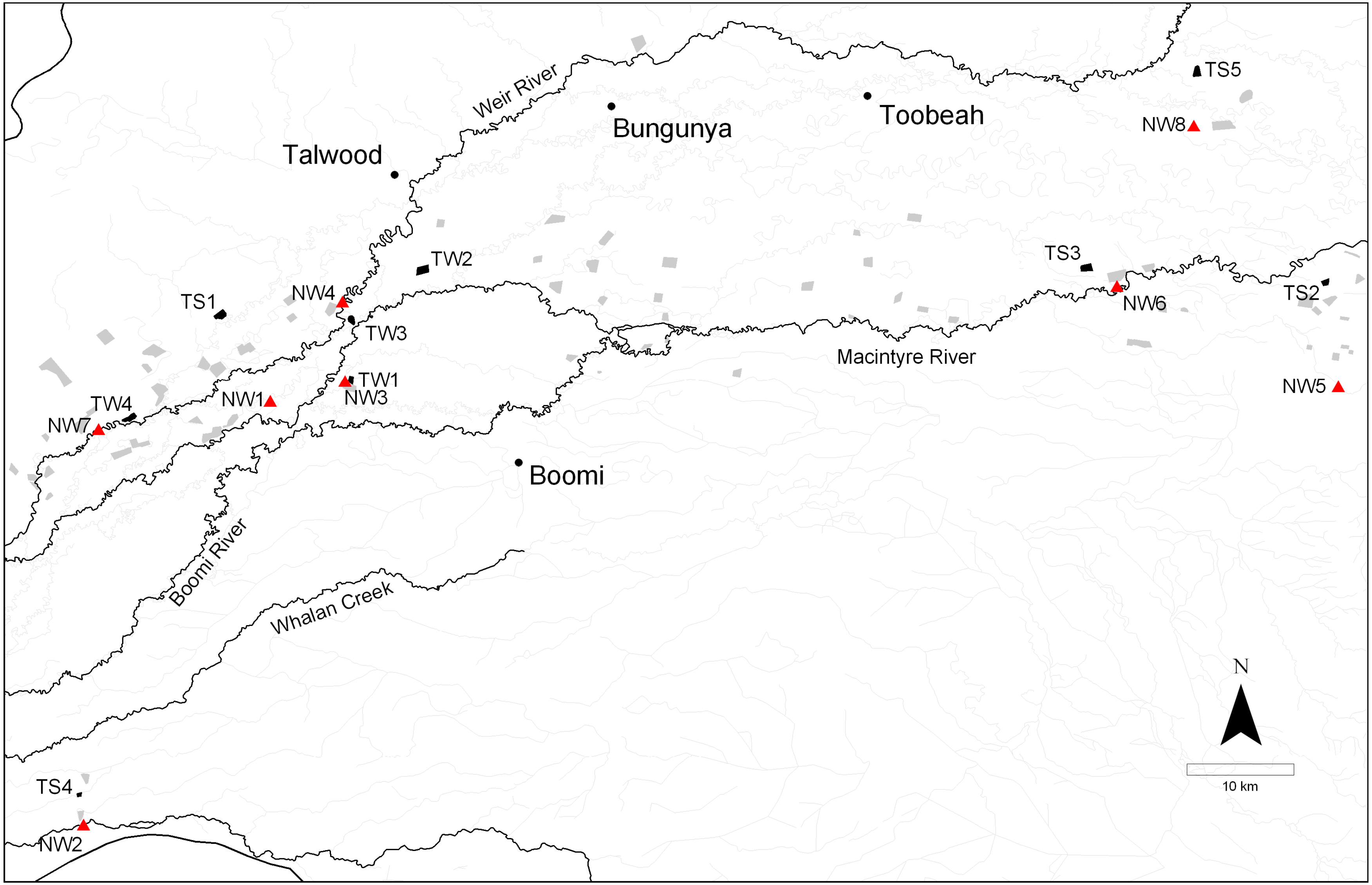


Fig. 5.1: Natural wetlands (triangles) and storage sites (black objects) sampled in the Border Rivers Catchment during the spatial study from August 2005 to October 2005. Grey objects represent other storages in the area.

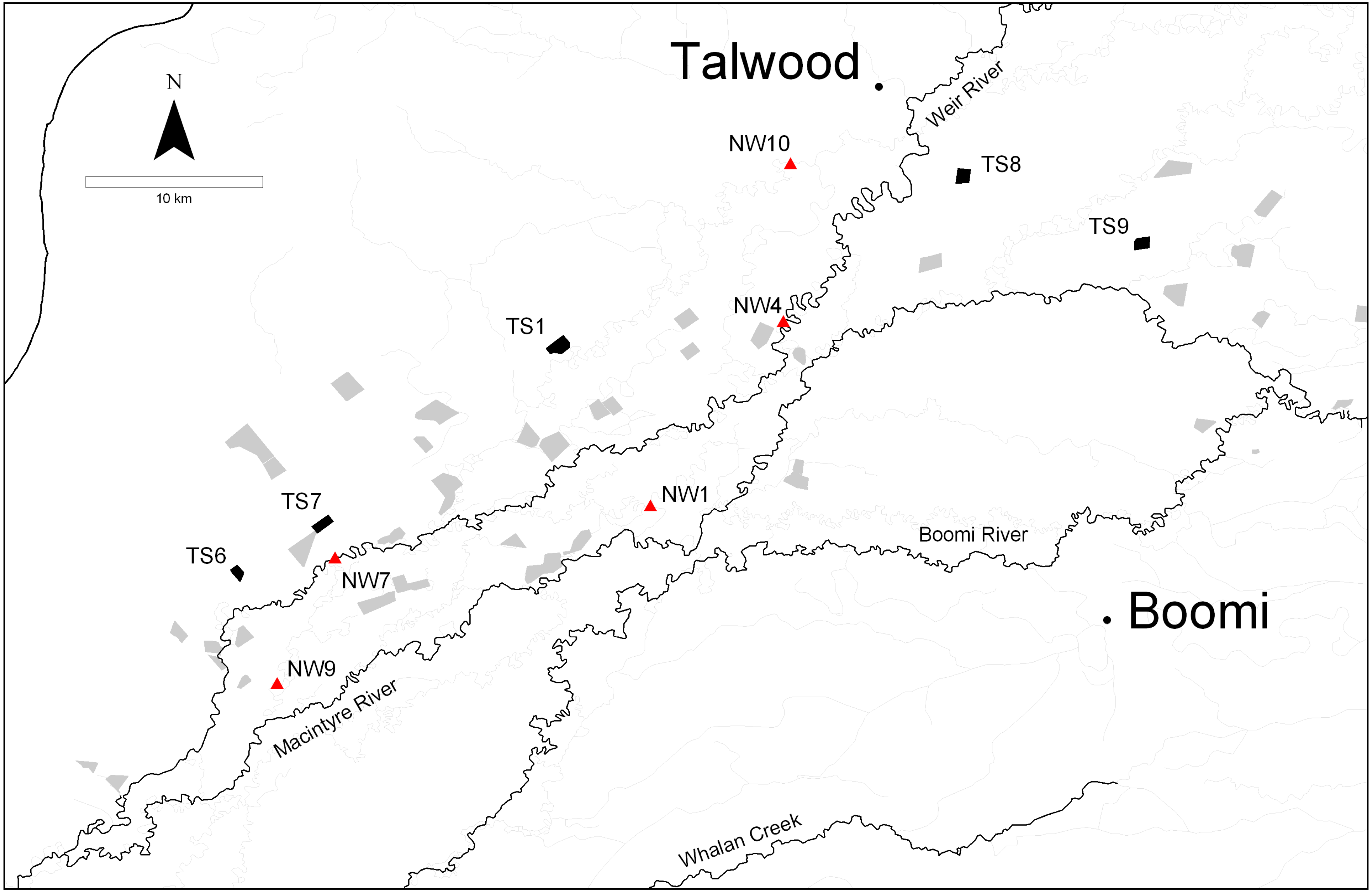


Fig. 5.2: Typical storage (black) and natural wetland (red triangles) sites sampled during the temporal study between November 2005 and February 2007. Grey objects represent other storages in the area.

5.2.2 Water Quality

Three replicate water samples were taken from each waterbody using either a 1 metre plastic tube to obtain an integrated sample from the top 1 m of the water column or, at sites where the water depth was below 1 m, a bucket. Conductivity (mS cm^{-1}), pH, spot dissolved oxygen (DO) (mg L^{-1}), turbidity (NTU), spot water temperature (^{o}C) and salinity (%) were measured in the field using a Horiba multiprobe U-10.

Mean water quality variables were calculated from three replicate water samples taken from each site on each sampling occasion, apart from T5 owing to equipment error. Values were range standardised and tested for normality before analysis. All water quality variables from the spatial study (T1) fitted a normal distribution and a one-way analysis of variance (ANOVA) was used to test for significant differences between the three waterbody types sampled in August 2005.

Of the water quality variables from the temporal study, an independent samples two-tailed *t*-test was used to test for significant spatial differences (between natural wetlands and typical storages) across sampling times T2 to T7 for those variables which were normal or rendered normal by transformation (Table 5.3). For those measures which could not be rendered normal via transformation (Table 5.3), differences between sampling occasions were explored using a non-parametric Kruskal-Wallis test.

Table 5.3: Transformations performed on water quality measures from the temporal study (Nov 05 to Feb 07) prior to analysis. C.N. = couldn't be rendered normal, NW = natural wetlands, TS = typical storages.

	All Sites	**NW**	**TS**
pH	Normal	Normal	Square root
Conductivity	C.N.	Normal	C.N.
Turbidity	Fourth root	Normal	Normal
Water temperature	Normal	Normal	Normal
Salinity	Square Root	Square root	Square root

To investigate temporal variations (T2 to T7) in water quality natural wetlands and typical storages were analysed separately. Results of normality tests and resultant transformations are summarised in Table 5.3. For normalised parameters one-way ANOVA was used to test for significant temporal differences while non-parametric

Kruskal-Wallis tests were used to test for temporal differences in those water quality parameters unable to be rendered normal via transformation. Analysis was undertaken using SPSS 15.0 (SPSS, 2006).

To characterise the overall water quality of sites across time, all water quality data were summarised using Principal Component Analysis (PCA) function in the multivariate statistics package PRIMER v. 5 (Clarke and Gorley, 2001). PCA settings of normalised data and a maximum of 5 components were used.

5.2.3 Macroinvertebrates

5.2.3.1 Field Collection

Macroinvertebrates were collected following a composite sampling regime (Marshall *et al.*, 2006). The distribution of habitats, including bare bank, aquatic vegetation and woody debris (snags), in a 100 m reach was surveyed from the shore and recorded. A 250 µm mesh sweep net was used to representatively sample habitats over 20 m in the proportions that they were present. Thus, the length of sample collected from each habitat type was proportional to their contribution to the reach.

Three replicate samples were taken from each site. Samples were preserved in 70% ethanol in the field and later washed through nested sieves (4000, 2000, 1000, 250 µm) and the organisms sorted, counted and identified as far as practical using various keys listed in Hawking (2000). Unidentified specimens were recorded as separate taxa (e.g. tiny Chironomid).

5.2.3.2 Data Analysis

Two measures of assemblage composition were used for exploring macroinvertebrate patterns: assemblage structure (based on taxonomic abundances) and assemblage function (based on FMG proportions calculated from abundance data). Taxa were split into four FMG's: zero mobility (e.g. *Macrobrachium* sp.), low mobility (e.g. Odonata), high mobility (e.g. Corixidae) and very high mobility (e.g. Coleoptera). FMG proportions were based on the abundances of taxa within each group.

A number of summary measures of community structure were calculated for each sample from the raw abundance data, these included: number of taxa (S), species richness (d), abundance (N), Shannon diversity (H'), Simpson diversity (1-Lambda') and Pielou's evenness (J'). Results of normality tests and resultant transformations are summarised in Table 5.4. Using the data from the spatial study (T1), where variables were normal or rendered normal by transformation, a one-way ANOVA was used to identify significant differences in these summary variables between waterbody types. Rarefaction analysis was used to test whether higher species richness was merely a consequence of a higher number of individuals sampled by creating species accumulation curves in EstimateS (Colwell, 2006).

When the data from the temporal study for both waterbody types was tested for normality the summary variables were transformed as required (Table 5.4). An independent samples two-tailed *t*-test was used to explore spatial differences (between waterbodies) in those summary measures which were normal or could be rendered normal via transformation. Any spatial differences in those measures which were not normal were identified using non-parametric Kruskal-Wallis tests.

The summary measures from the temporal study were transformed as needed (Table 5.4) and a one-way ANOVA was used to identify significant temporal differences in these variables for the two waterbody types separately.

Table 5.4: Transformations performed on summary variables of macroinvertebrate data prior to analysis. C.N. = couldn't be rendered normal, Ref. = reflected, NW = natural wetlands, TS = typical storages.

Variable	All waterbody types	NW	TS
Spatial Study -			
S: number of taxa	Square root		
d: species richness	Square root		
N: abundance	Log		
H': Shannon diversity	Normal		
1-Lambda': Simpson diversity	Normal		
J': Pielou's evenness	Normal		
Temporal Study -			
S: number of taxa	Square root	Normal	Normal
d: species richness	Square root	Normal	Normal
N: abundance	Log	Log	Log
H': Shannon diversity	Normal	Normal	Normal
1-Lambda': Simpson diversity	C.N.	Ref. square root	Normal
J': Pielou's evenness	C.N.	Ref. square root	Normal

Patterns of variation in macroinvertebrate assemblage structure across wetland types and sampling times were analysed using the non-standardised $\log_{10}(x+1)$ (absolute abundance), standardised $\log_{10}(x+1)$ (relative abundance) and species presence/absence data. Samples not containing macroinvertebrate taxa were removed before analysis; this constituted five samples, all from typical storages with Bray-Curtis similarity used as the measure of similarity between samples. One-way ANOSIM, based on Bray-Curtis similarity was used to explore differences in macroinvertebrate assemblage structure between the waterbody types sampled during the spatial study. Two-way crossed ANOSIMs, using natural wetland and typical storage sampling data, were used to identify differences in macroinvertebrate assemblage structure between both sampling times and waterbody types. Temporal and spatial assemblage patterns were visualised using MDS ordinations and UPGMA with default (PRIMER) settings and 100 random starts (PRIMER 5.2.9; Clarke and Gorley, 2001). SIMPER was used to reveal those taxa having a significant contribution to differences between waterbody types.

BIO-ENV (Clarke and Warwick, 2001) was used to investigate relationships between habitat features, water chemistry, fish and macroinvertebrate assemblage structure. Auto-correlated variables were removed before BIO-ENV analysis (Table 5.5). Environmental similarity matrices were based on normalised Euclidean distance with no transformation of the data. A spearman rank correlation was used with the maximum number of variables set at five, unless otherwise indicated.

Table 5.5: Morphology and water quality variables measured at the storage sites in the Border Rivers Catchment (see Chapter 3 for explanation). Shaded variables were removed prior to BIO-ENV analyses.

Morphology	**Water Quality**
Age (Y)	pH
Depth (D)	Conductivity
Area (A)	Turbidity
Perimeter (P)	Water temperature
Length (L)	Salinity
Width (W)	
Length:Width (LW)	
Capacity (C)	
Distance from Barwon River (Dis)	
Circularity Index (CI)	
Horton's Form Factor (HFF)	
Elongation Ratio (ER)	

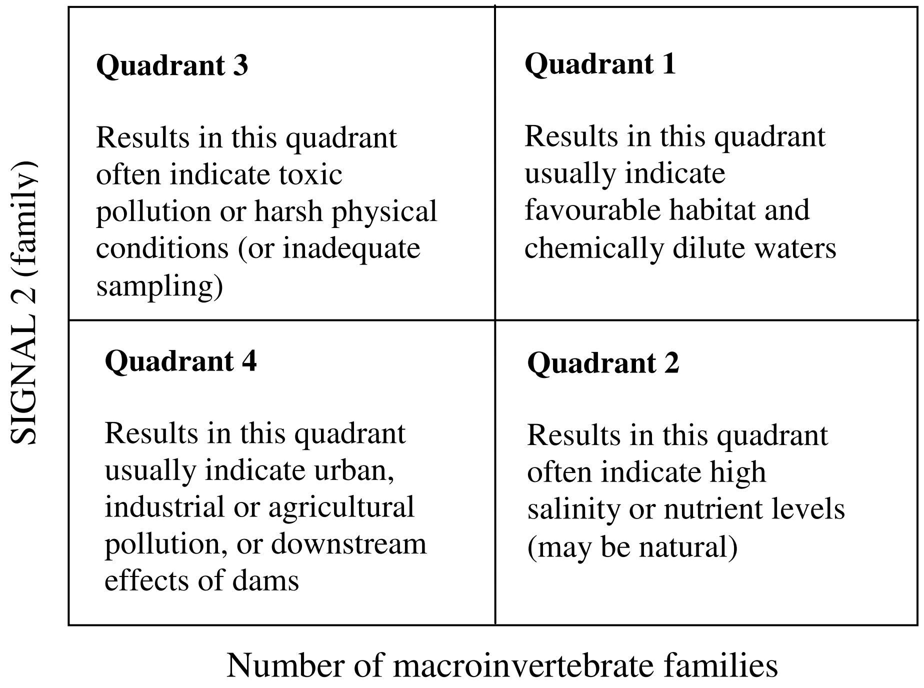
5.2.3.3 EPT index values and SIGNAL scores

The EPT index, the sum of the number of distinct taxa within the orders Ephemeroptera, Plecoptera and Trichoptera (EPT), is based on the premise that high-quality streams and rivers usually have the greatest species richness (WSI, 2001). As these three orders of macroinvertebrates are highly sensitive to pollution, they are often used as water quality indicators. In this study the EPT index was calculated for each replicate sample and the mean of the total replicates from each waterbody type was calculated to give an overall EPT index for each waterbody type. In the case of natural wetlands during the spatial sampling this was the mean of 24 index values (3 reps x 8 sites).

Stream Invertebrate Grade Number – Average Level (SIGNAL) scores are another method used to indicate water quality in the river from which the macroinvertebrate sample was collected (Chessman, 2003). This study used the grades developed by Chessman (2003) in SIGNAL 2 for family level. As SIGNAL was developed for streams, some of the orders that have the highest SIGNAL 2 sensitivity grades will not be abundant in wetlands, for example stoneflies. Therefore wetlands are likely to have naturally lower SIGNAL scores than streams in the same region.

SIGNAL scores were calculated for each replicate sample at each site. SIGNAL scores were assigned to each taxa using the family grades given in Chessman (2003). In cases were specimens could not be identified to family level (e.g. ‘unidentified damselfly’, Acarina and Oligochaeta), these were given a score from the order-class-phylum grades (Chessman, 2003). Taxa that had no grade number assigned were removed from the data. To interpret SIGNAL 2 scores, the SIGNAL 2 score is plotted against the number of taxa in each replicate. The resulting graph is then split into four quadrants to give an indication of the water quality at each site (Fig. 5.3). Borders between quadrants vary with geographic area, sampling method and habitat type (Chessman, 2003).

To explore the differences in EPT index values and SIGNAL 2 scores between the six different sampling times and two waterbody types, two-way crossed ANOSIMs based on natural wetland and typical storage data were used.



Quadrant 3 Results in this quadrant often indicate toxic pollution or harsh physical conditions (or inadequate sampling)	**Quadrant 1** Results in this quadrant usually indicate favourable habitat and chemically dilute waters
Quadrant 4 Results in this quadrant usually indicate urban, industrial or agricultural pollution, or downstream effects of dams	**Quadrant 2** Results in this quadrant often indicate high salinity or nutrient levels (may be natural)

Fig. 5.3: The quadrant diagram for the family version of SIGNAL 2 (Chessman, 2003).

5.2.4 Zooplankton

5.2.4.1 Field Collection

Zooplankton samples were obtained by collecting 10 litres of water from the edge of the waterbody in a bucket and pouring it through a 53 µm net. Three replicate samples were taken at each site and preserved in 70% ethanol in the field. In the laboratory, samples were coloured with Rose Bengal Dye and made up to a known volume. A Sedgewick Rafter counting chamber was used to count and identify the organisms to genus using the key of Shiel (1995).

5.2.4.2 Data Analysis

The same summary variables as calculated for macroinvertebrate assemblages were calculated for zooplankton assemblage composition. The resultant data was tested for normality and each variable transformed accordingly prior to analysis (Table 5.6). An independent samples two-tailed *t*-test was used to identify any significant spatial differences (between waterbodies) across all sampling times in those variables which were normal or could be rendered normal via transformation. Spatial differences in variables which could not be made normal were explored using a non-parametric Kruskal-Wallis test.

Table 5.6: Transformations performed on summary variables of zooplankton data from the temporal study (Nov 05 to Feb 07) prior to analysis. C.N. = couldn't be rendered normal, Ref. = reflected, NW = natural wetlands, TS = typical storages.

Variable	All waterbody types	NW	TS
S: number of taxa	Normal	Normal	Normal
d: species richness	Square root	Square root	Normal
N: abundance	Log	Log	Log
H': Shannon diversity	Normal	Normal	Normal
1-Lambda': Simpson diversity	C.N.	Ref. Log	C.N.
J': Pielou's evenness	Ref. Log	Normal	Ref. Square root

Differences between all sampling times (T2-T7), for each waterbody type separately, were explored using one-way ANOVA; where data could not be rendered normal by transformation non-parametric Kruskal-Wallis tests were used to test for significant temporal variations (T2 to T7) for each waterbody type.

Patterns of variation in zooplankton assemblage structure across waterbody types and sampling times were analysed using non-standardised $\log_{10}(x+1)$ (absolute abundance), standardised $\log_{10}(x+1)$ (relative abundance) and species presence/absence data with all taxa included. Bray-Curtis similarity was used as the measure of difference between samples. Assemblage patterns were visualised using non-metric multi dimensional scaling (MDS) ordinations with default (PRIMER) settings and 100 random starts (PRIMER 5.2.9; Clarke and Gorley, 2001). Solutions were calculated in two and three dimensions and the two dimensional solution is presented if its stress was less than or equal to 0.2.

Two-way crossed ANOSIMs based on the same Bray-Curtis similarity matrices were used to test for differences in zooplankton assemblage structure between the five different sampling times and two waterbody types. Default (PRIMER) settings with 10,000 permutations were used (PRIMER 5.2.9; Clarke and Gorley, 2001). SIMPER was used to identify the contributions of different taxa to the similarity within waterbody types and the dissimilarities between different waterbody types.

BIO-ENV was used to investigate relationships between water quality data, habitat availability and zooplankton assemblages. Environmental similarity matrices were based on normalised Euclidean distance, with no transformation of the data. Within BIO-ENV a Spearman rank correlation was used with the maximum number of variables set at five.

5.3 Results

5.3.1 Water Quality

Continuing drought meant both rainfall and river flow were low for the duration of the study, with water levels in both storages and natural wetlands decreasing throughout the study (Fig. 5.4). A flow pulse in the Weir River saw increases in water depth until March 2006 in two natural wetlands, Crawler Lagoon (NW1) and Tarrawatta Weir Waterhole (NW7), but both subsequently had decreasing water levels for the remainder of the study. Water levels in the typical storage, Tarrawatta N End Dam (TS7), also increased in March 2006. Depth measurements were recorded only at the time of sampling and there may have been minor fluctuations in water levels between sampling times. This is especially likely at storages where, under certain licence conditions, water can be ordered from dams upstream for irrigation of crops. In this case, water may be in the storage for only a short period of time.

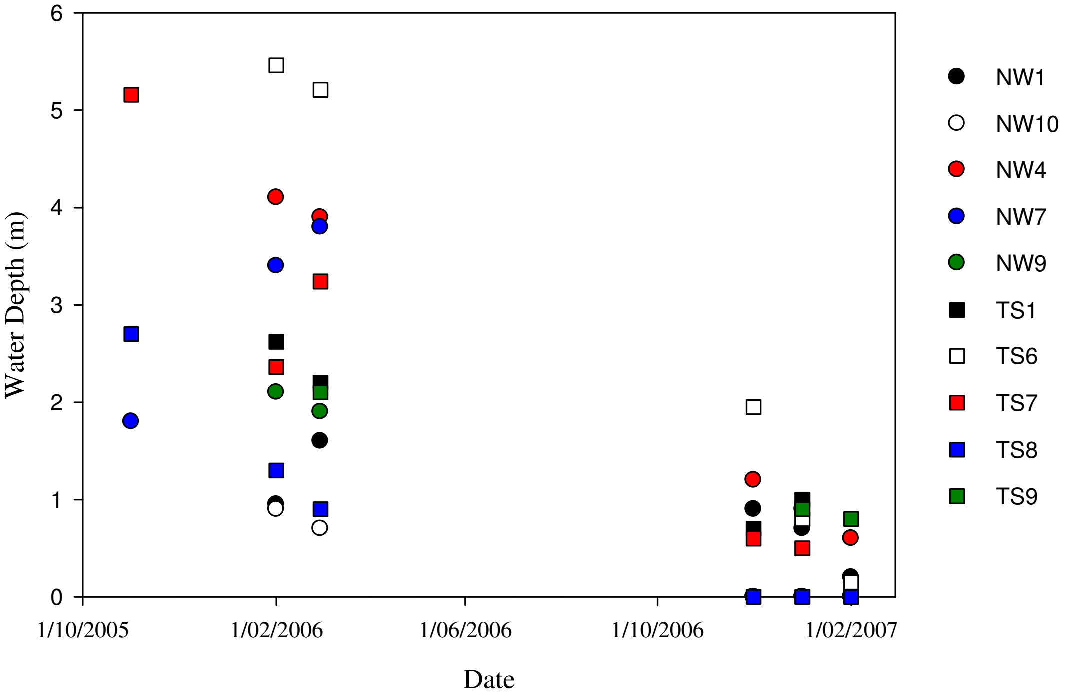


Fig. 5.4: Changes in water levels over the course of the temporal study period at natural wetlands (NW) and typical storages (TS).

Mean spot DO measurements ranged from 5.90 mg L^{-1} to 13.15 mg L^{-1} across all sampling occasions (Appendix 10), however, as spot measurement of DO varies with temperature and the time of sampling, spatial and temporal variation in measured DO will not be discussed further.

Mean conductivity in natural wetlands ranged from 0.102 mS cm^{-1} in Warril Creek (NW10) to 0.528 mS cm^{-1} in Gil Gil Creek (NW2), with mean salinity ranging from 0.00% to 0.02% at all sites (Appendix 10). Mean water temperature varied from 13.8 ^{o}C to 34.2 ^{o}C, with winter temperatures ranging from 13.2 ^{o}C to 26.3 ^{o}C and summer temperatures from 23.6 ^{o}C to 34.2 ^{o}C. Mean turbidity levels varied greatly in the natural wetlands, from a low of 16 NTU at Punbougal Lagoon (NW5) to a high of 1590 NTU at Crawler Lagoon (NW1). All sites were slightly alkaline with mean pH ranging from 7.26 to 8.84.

In the 'no tailwater' storages, conductivity ranged from 0.155 mS cm^{-1} at Tarrawatta Weir Dam (TW4) to 0.547 mS cm^{-1} at Barra Storage (TW1), with salinity ranging from 0.02% at Barra Storage while all other sites recorded values of either 0.00% or 0.01%. Minimum water temperature (20.8 ^{o}C) was slightly higher than recorded in natural wetlands while maximum water temperature only reached 24.1 ^{o}C. This is to be expected as 'no tailwater' storages were sampled only in the spring of 2005 and not during the summer months. Again there was substantial variation in turbidity readings from a low of 108 NTU at Barra Storage (TW1) to a high of 713 NTU at Boolarwell Storage (TW2). pH levels were higher than recorded in natural wetlands with values ranging from 7.91 at Newinga Storage (TW3) to 9.15 at Barra Storage (TW1).

There was considerable variation in conductivity and salinity values in typical storages, ranging from 0.194 mS cm^{-1} to 1.453 mS cm^{-1} and from 0.00% to 0.06%, respectively. The highest values for both conductivity and salinity were recorded in February 2007 when water levels were very low. Maximum mean water temperature (30.7 ^{o}C) was lower than recorded in natural wetlands while mean minimum water temperature was similar (14.8 ^{o}C). Minimum turbidity was higher than in other waterbody types with a low of 184 NTU at Tarrawatta N End Dam (TS7) while maximum turbidity (857 NTU) was similar to that recorded in 'no tailwater' storages but lower than in natural wetlands,. The pH values were similar to 'no tailwater' storages with values ranging from 7.42 at Royston Storage (TS2) to 9.88 at Tarrawatta N End Dam (TS7).

There were no significant differences in any of the water quality variables during spatial sampling (T1) between the three waterbody types (p values for one-way ANOVA for each variable > 0.05). During temporal sampling there were significant spatial differences in pH (t-test, p < 0.001), square root of salinity (t-test, $p < 0.01$) and conductivity (Kruskal-Wallis, $p < 0.05$). All three parameters were higher in typical storages compared with natural wetlands.

There were also some noteworthy temporal patterns (Appendix 10). Lowest water temperatures were recorded on the first sampling occasion, reflecting spring sampling, and water temperatures increased from August 2005 to February 2006 and again from March 2006 to February 2007. Conductivity and salinity were highest, and became more variable in January and February 2007 as waterbodies dried, with the highest values exhibited in typical storages. Turbidity levels were high throughout sampling with highest readings in early 2007 in natural wetlands (Appendix 10). In natural wetlands there were significant temporal differences in conductivity ($F_{4,10} = 9.152$, $p < 0.01$), water temperature ($F_{4,10} = 39.652$, $p < 0.001$) and the square root of salinity ($F_{4,10} = 4.663$, $p < 0.05$). In typical storages, there were significant temporal differences in the square root of pH ($F_{4,12} = 6.711$, $p < 0.01$) with higher pH in January 2007 compared to November 2005 and March 2006. There was also a significant difference in conductivity (Kruskal-Wallis, $p < 0.05$) between sampling times, with higher conductivity readings in January and February 2007 compared with November 2005 and February and March 2006.

Spatial and temporal variation in mean water quality parameters were further summarised by PCA (Fig. 5.5). The first two axes explained 79% of the total variation in the data, with 54% attributed to the first axis alone (Table 5.7). The first axis was negatively correlated with pH, conductivity and salinity, in that, samples with higher pH, conductivity and salinity readings were found to the left of this axis.

The second axis was positively correlated with water temperature and turbidity. Therefore, the highly turbid samples with higher temperatures were aligned to the top of this axis (Fig. 5.5). The PCA suggested that, collectively, storage sites exhibited water quality variables that spanned a wider range than observed in natural wetlands in relation to Axis 1 but were more tightly grouped on Axis 2 (Fig. 5.5). Variation between sites within sampling times appeared to increase from August 2005 to February 2007 (Fig. 5.5). As predicted, the highest recordings for turbidity, conductivity and salinity were from later sampling occasions as the sites were drying (Fig. 5.5).

Table 5.7: Percent variance of site water quality extracted by PCA for the first five axes and contribution of the measured variables to the first five eigenvectors based on range standardised data.

	Axis	Eigenvalue	% of Variance	Cum. % of Variance	
	1	2.69	53.9	53.9	
	2	1.28	25.6	79.4	
	3	0.62	12.4	91.8	
	4	0.38	7.6	99.5	
	5	0.03	0.5	100	
Characteristic			**First 5 Eigenvectors**		
	1	2	3	4	5
pH	-0.503	-0.171	-0.18	-0.828	0.015
Conductivity	-0.579	-0.089	0.222	0.334	0.704
Turbidity	-0.045	0.771	0.58	-0.258	0
Temperature	-0.273	0.598	-0.726	0.201	-0.015
Salinity	-0.579	-0.104	0.232	0.31	-0.71

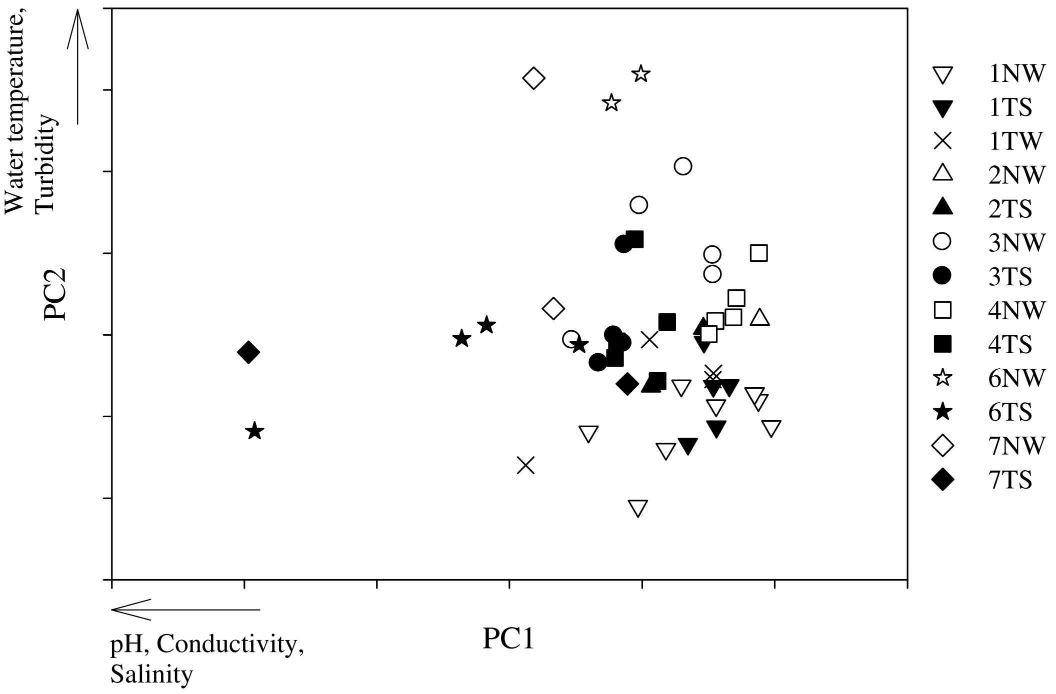


Fig. 5.5: Position of each study site within the ordination space defined by the first two factors identified by PCA of range standardised water quality data. a) depicts the different waterbody types and b) depicts the different waterbody types and sampling times. NW = natural wetland, TW = 'no tailwater' storage, TS = typical storage, number = sampling time (1=August-October 2005, 2 = November 2005, 3 = February 2006, 4 = March 2006, 5 = December 2006, 6 = January 2007, 7 = February 2007).

5.3.2 Macroinvertebrate assemblages

5.3.2.1 Spatial community composition

In natural wetlands, a total of 14,629 individuals from 84 taxa were recorded from 24 samples (Appendix 11). In 15 samples from typical storages, 2,142 individuals from 34 taxa were recorded. Only 12 samples from 'no tailwater' storages were collected and these totalled 7,611 individuals from 34 taxa (not the same 34 taxa as collected in typical storages). Pooled data from all three waterbody groups comprised of 95 taxa with 17 taxa common to all waterbody types and 45 taxa specific to natural wetlands. Typical storages had lower abundances than natural wetlands and 'no tailwater' storages (mean number of individuals per sample, 142, 610 and 634 respectively).

Insects comprised 83% of taxa and 62% of individuals in natural wetlands (Table 5.8). The richest of all invertebrate orders were Diptera (27) and Coleoptera (15) while the most abundant were Diptera and Nematoda (Table 5.8), Coleoptera accounted for less than 1% of the abundance. After nematodes, other abundant taxa included *Cladotanytarsus* sp. and *Cricotopus* sp. (Chironomidae) and unidentified tiny

Table 5.8: Taxa and their contribution to the macroinvertebrate assemblage as a percentage of the total individuals and a percentage of the total taxa at each waterbody type from samples collected between August 2005 and October 2005 (Time 1).

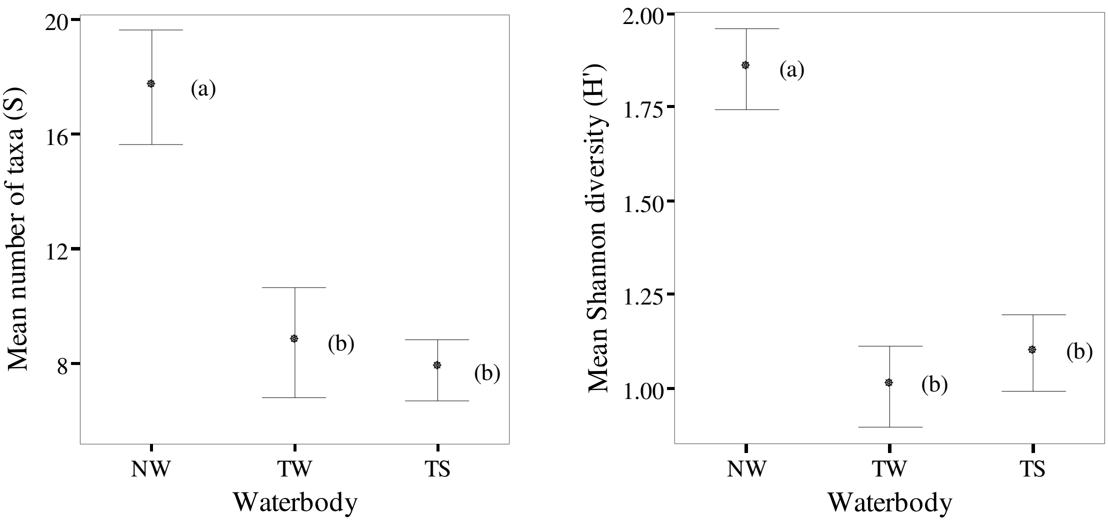
	Natural Wetlands		**'No tailwater' storages**		**Typical Storages**	
	% Individuals	% Taxa	% Individuals	% Taxa	% Individuals	% Taxa
Taxa						
Crustacea	1.47	5.95	0.01	2.94	1.68	8.82
Gastropoda	1.00	4.76	0.00	0.00	0.09	5.88
Oligochaeta	0.00	0.00	0.00	0.00	0.23	2.94
Nemertea	0.34	1.19	0.00	0.00	4.81	2.94
Arachnida	0.40	2.38	0.00	0.00	0.05	2.94
Nematoda	34.31	1.19	0.22	2.94	0.00	0.00
Bryozoa	0.14	1.19	0.00	0.00	0.00	0.00
Insecta	62.34	83.33	99.76	94.12	93.14	76.47
Lepidoptera	0.04	1.19	0.00	0.00	0.00	0.00
Coleoptera	0.30	17.86	0.04	5.88	0.84	20.59
Diptera	37.63	32.14	16.37	50.00	87.96	35.29
Ephemeroptera	6.88	3.57	0.59	11.76	1.03	5.88
Hemiptera	8.21	9.52	79.62	8.82	2.99	8.82
Odonata	7.74	11.90	0.00	0.00	0.09	2.94
Trichoptera	1.54	7.14	3.14	17.65	0.23	2.94
Mobility Groups						
Zero	37.66	16.67	0.24	5.88	6.86	23.53
Low	53.79	54.76	20.10	79.41	89.31	47.06
High	8.21	9.52	79.62	8.82	2.99	8.82
Very High	0.34	19.05	0.04	5.88	0.84	20.59

chironomids, ceratopogonids (Ceratopogoninae) and unidentified tiny ceratopogonids, *Micronecta* sp. (Corixidae) and *Cloeon* sp. (Baetidae). Eighteen taxa occurred only once. In the natural wetlands the low mobility taxa were the most abundant group and had the highest richness (Table 5.8).

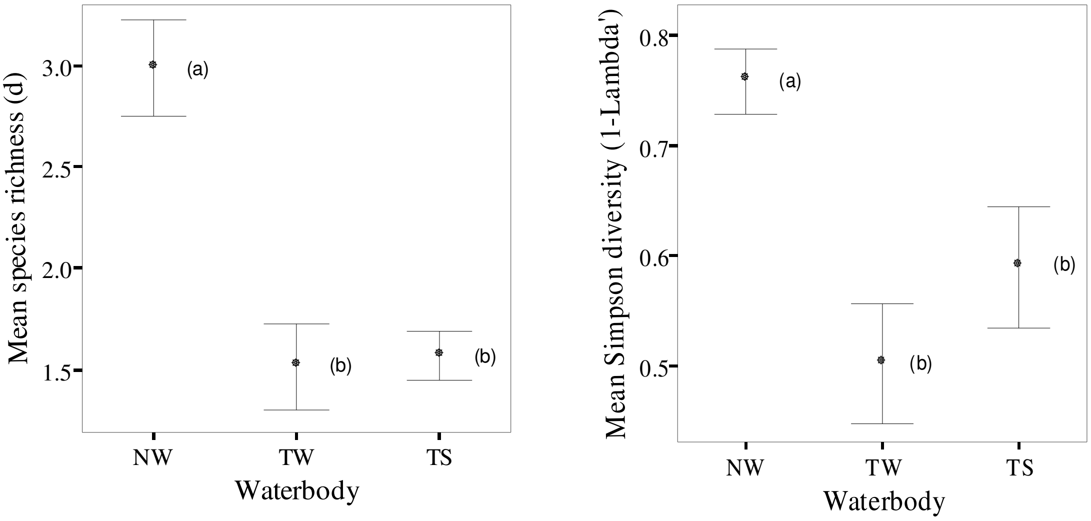
In 'no tailwater' storages, insects again dominated the assemblage (94% of taxa and greater than 99% of individuals). The richest order was the Diptera (17) but the most abundant order was Hemiptera (80%). *Micronecta* spp. (Corixidae), were the single most abundant taxon, accounting for 73% of individuals. Other abundant taxa included *Ablabesmyia* sp. (Chironomidae) and other tiny chironomids, *Anisops* sp. (Notonectidae), *Agraptocorixa* sp. (Corixidae) and *Hellyethira malleoforma* (Hydroptilidae). Six taxa occurred only once. Highly mobile taxa were the most abundant functional group while the low mobility group comprised the most taxa (Table 5.8).

In typical storages, insects made up 76% of taxa and 93% of individuals (Table 5.8). Diptera were again the richest (12) and most abundant (88%) order. The Chironomidae family accounted for 85% of individuals with *Dicrotendipes* sp. as the single most abundant taxon. Other abundant taxa included *Cricotopus* sp. (Chironomidae), and other tiny chironomids. Nine taxa occurred only once. Functional groups at typical storages followed a similar pattern to that found at natural wetlands (Table 5.8). Low mobility functional group dominated the abundance and contained the greatest number of taxa (Table 5.8).

There was a significant difference in the square root of number of taxa (S) ($F_{2,48}$ = 10.996, $p < 0.001$), Shannon diversity (H') ($F_{2,48}$ = 19.448, $p < 0.001$), square root of species richness (Margalef d) ($F_{2,48}$ = 16.757, $p < 0.001$) and Simpson diversity (1-Lambda') ($F_{2,48}$ = 9.518, $p < 0.001$) between the three waterbody types. Neither the log of abundance (N) ($F_{2,48}$ = 2.674, $p > 0.05$) or Pielou's evenness (J') ($F_{2,48}$ = 1.818, $p > 0.05$) differed significantly between waterbody types. Post hoc tests suggested higher richness and diversity in natural wetlands compared to both 'no tailwater' storages and typical storages (Fig. 5.6). Rarefaction analysis confirmed that for equivalent abundances species richness in natural wetlands still exceeds that found in both storage types (Fig 5.7).



a) mean number of taxa (S) b) mean Shannon diversity (H')



c) mean species richness (Margalef d) d) mean Simpson diversity (1-lambda)

Fig. 5.6: Mean (± s.e.) number of taxa, Shannon diversity, species richness and Simpson diversity for the three waterbody types. Groups suggested by post hoc tests are shown, (a) and (b). NW = natural wetland, TW = 'no tailwater' storage, TS = typical storage.

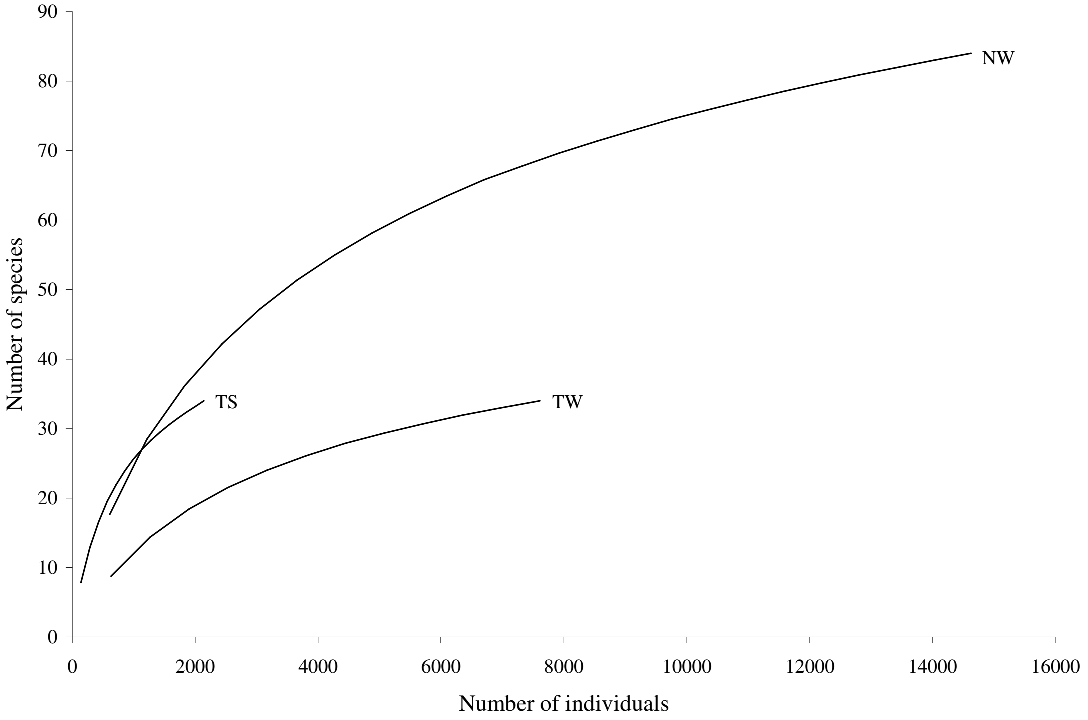


Fig. 5.7: Rarefaction curves for macroinvertebrate samples collected during the spatial study (T1) at natural wetlands (NW), typical storages (TS) and 'no tailwater' storages (TW).

The spatial variation in macroinvertebrate assemblage composition across the 17 waterbody sites can be seen in both the ordination (Fig.5.8) and the dendrogram based on UPGMA cluster (Fig. 5.9) (see Appendix 12 for ordination plots based on presence/absence data and absolute abundance data). Similarities between groups and sites were low, most splits occurred below 60% similarity; generally, natural wetland sites formed a distinct group, while storage sites formed another. One-way ANOSIM suggested significant spatial variation between wetlands and storage types in absolute abundance, relative abundance and presence/absence data (Table 5.9 and Appendix 13). In all cases pairwise tests suggested natural wetlands were significantly different to the storages (Table 5.9).

Table 5.9: Summary of one-way ANOSIM results comparing macroinvertebrate assemblage structure among wetland groups based on presence/absence data, absolute abundance $\mathrm{Log}_{10}(x+1)$ and relative abundance $\mathrm{Log}_{10}(x+1)$.

	Transformation	Global R	*P*	Pairwise Test
Species Diversity	Presence/Absence	0.272	< 0.001	NW-TW(0.001), NW-TS (< 0.001), TW-TS(0.002)
Absolute Abundance	$\mathrm{Log}_{10}(x+1)$	0.276	< 0.001	NW-TW(0.001), NW-TS (< 0.001), TW-TS(0.003)
Relative Abundance	$\mathrm{Log}_{10}(x+1)$	0.371	< 0.001	All three < 0.001

The average dissimilarity in macroinvertebrate assemblage composition between natural wetlands and the storages was 79%, while the average similarity within waterbody types was very low for all waterbodies (25 – 30%). This suggested that, although there was high variability between sites, the three waterbody types were very different to each other in relation to macroinvertebrate assemblage structure. Tiny chironomids and the hemipteran *Micronecta* sp. contributed to the uniqueness of the natural wetland assemblage, while in the 'no tailwater' storages it was the highly mobile hemipterans, *Anisops* sp. and *Micronecta* sp., and in the typical storages it was the chironomids *Dicrotendipes* sp. and *Cricotopus* sp. (Table 5.10).

Water quality, available habitat, morphology or fish assemblages of the waterbody site did not explain a significant amount of the observed spatial variation in macroinvertebrate assemblage structure (Table 5.11). Of the variability explained, morphology variables accounted for more than water quality or habitat availability but all r_s were low (< 0.5).

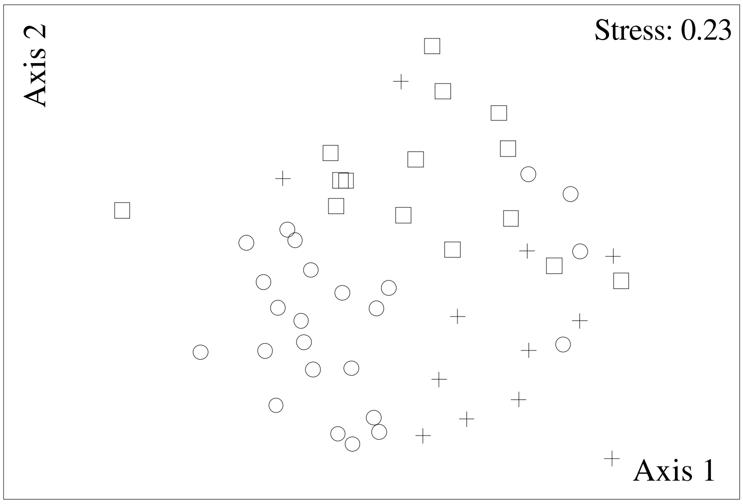


Fig. 5.8: Non-metric multi-dimensional scaling plot for macroinvertebrate assemblage structure based on $Log_{10}(x+1)$ standardised for relative abundance data. O Natural wetlands, + Storages without tailwater, □ Typical storages.

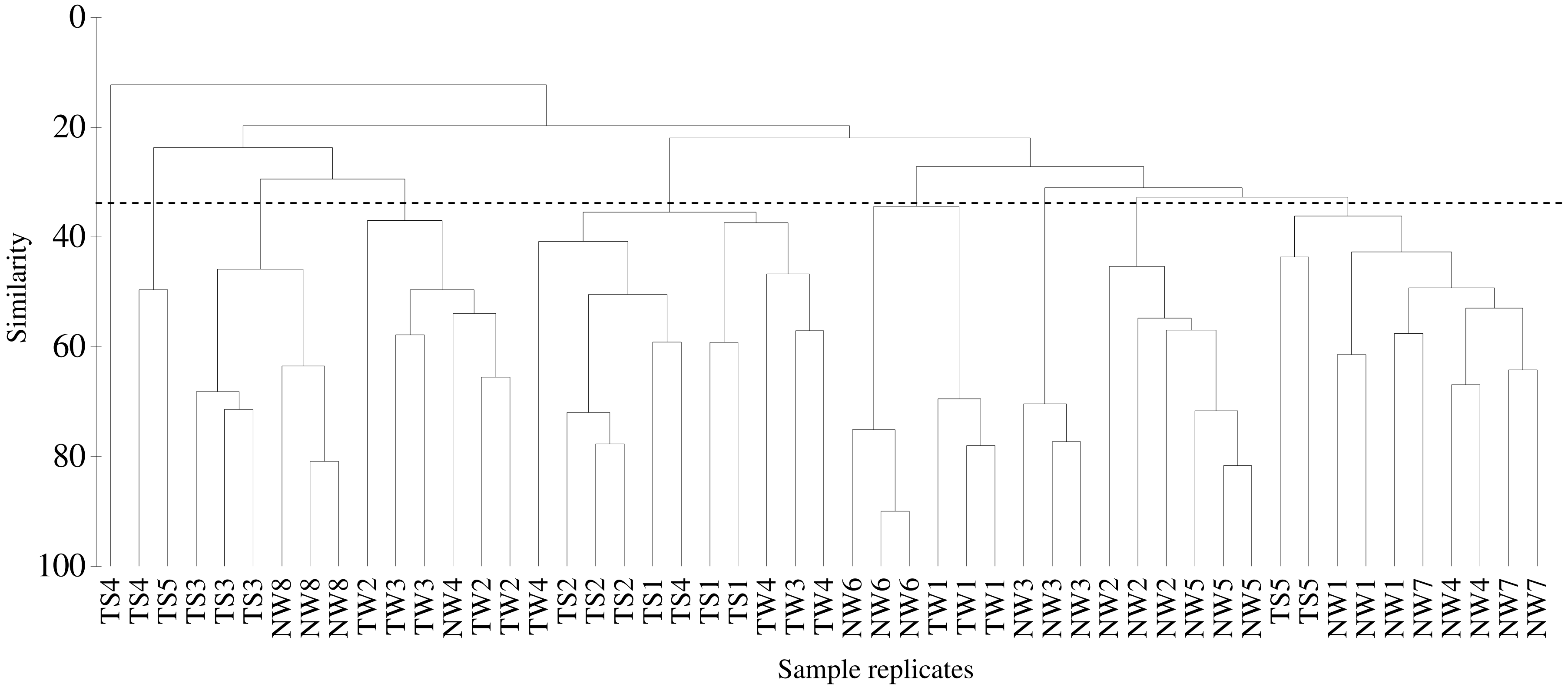


Fig. 5.9: Dendrogram of waterbody types based on relative abundance $Log_{10}(x+1)$ data of macroinvertebrates across all sampling occasions. NW = natural wetland, TW = 'no tailwater' storage, TS = typical storage.

Table 5.10: Summary of SIMPER results for within waterbody type similarity and between waterbody type dissimilarity based on $\log_{10}(x+1)$ data with a cut off for low contributions of 90%. Taxa contributing >5% to the similarity or dissimilarity of groups are noted.

	Natural Wetland		**No Tailwater**		**Typical**	
Average Similarity	30.02		26.60		25.49	
	Species	%	Species	%	Species	%
	Tiny Chironomids	18.44	Anisops sp.	30.76	Dicrotendipes sp.	37.98
	Micronecta sp.	11.81	Micronecta sp.	23.57	Cricotopus sp.	22.2
	Nematoda	7.82	Cricotopus sp.	16.03	Tiny Chironomids	8.99
	Parachironomus sp.	7.39	Dicrotendipes sp.	6.48	Parachironomus sp.	6.93
	Cladotanytarsus sp.	6.64	Tiny Chironomids	5.65		
	Cloeon sp.	5.21	Coelopynia sp.	5.37		
	Cricotopus sp.	5.14				
	Dicrotendipes sp.	5.08				
	Ceratopogoninae	5.03				
Average Dissimilarity	**NW v TW**	79.84	**NW v TS**	78.84	**TW v TS**	80.98
	Species	%	Species	%	Species	%
	Tiny Chironomids	7.26	Tiny Chironomids	7.18	Anisops sp.	9.79
	Micronecta sp.	6.52	Dicrotendipes sp.	6.29	Dicrotendipes sp.	9.74
	Nematoda	5.3	Micronecta sp.	5.71	Cricotopus sp.	9.59
	Anisops sp.	5.15	Nematoda	5.55	Micronecta sp.	9.33
			Cricotopus sp.	5.48	Tiny Chironomids	7.59

Table 5.11: Summary of BIO-ENV results based on Spearman rank correlations (r_s) between macroinvertebrate assemblage structure, habitat availability variables, water chemistry variables and macroinvertebrate taxa. Results presented for best possible solution only.

	Transformation	**Best Variable Combination**	**Correlation**
Habitat Availability Variables			
Species Diversity	Presence/absence	SN, BB, AQV, AQV+SN	0.105
Absolute Abundance	$\text{Log}_{10}(x+1)$	SN, BB, AQV, AQV+SN	0.089
Relative Abundance	$\text{Log}_{10}(x+1)$	SN, BB, AQV, AQV+SN	0.085
Water Chemistry Variables			
Species Diversity	Presence/absence	Turbidity, Water temperature	0.252
Absolute Abundance	$\text{Log}_{10}(x+1)$	Turbidity, Water temperature	0.277
Relative Abundance	$\text{Log}_{10}(x+1)$	Turbidity, Water temperature	0.249
Morphology Variables*			
Species Diversity	Presence/absence	Area, Width, Distance, CI	0.324
Absolute Abundance	$\text{Log}_{10}(x+1)$	Area, Width, Distance, CI	0.398
Relative Abundance	$\text{Log}_{10}(x+1)$	Age, Distance	0.310
Fish Assemblages			
Species Diversity	Presence/absence	1,2,3,4,7	0.210
Absolute Abundance	$\text{Log}_{10}(x+1)$	3, 4	0.269
Relative Abundance	$\text{Log}_{10}(x+1)$	4	0.148

SN = snags, BB = bare bank, AQV = aquatic vegetation, AQV+SN = aquatic vegetation + snags, *Natural wetland sites removed

5.3.2.2 Spatial variation in macroinvertebrate mobility

Low mobility taxa dominated the assemblage from typical storages and natural wetlands (Fig. 5.10) with site means greater than 66% at all natural wetlands and typical storages, with the exception of South Callandoon Lagoon (NW6) and Yambocully Lagoon (NW8). In contrast, the majority of macroinvertebrates collected from ‘no tailwater’ storages were high mobility taxa.

There were also statistically significant differences between the waterbody types with regard to the functional composition of assemblages (ANOSIM; R = 0.462, p = 0.001) (Table 5.12 and Appendix 14). Ordination also suggested spatial variation in the assemblage functional composition of sampling sites (Fig. 5.11). SIMPER suggested greater proportions of zero mobile taxa at the natural wetland sites compared with ‘no tailwater’ and typical storages. In comparison, highly mobile taxa were found in greater proportions at ‘no tailwater’ storages.

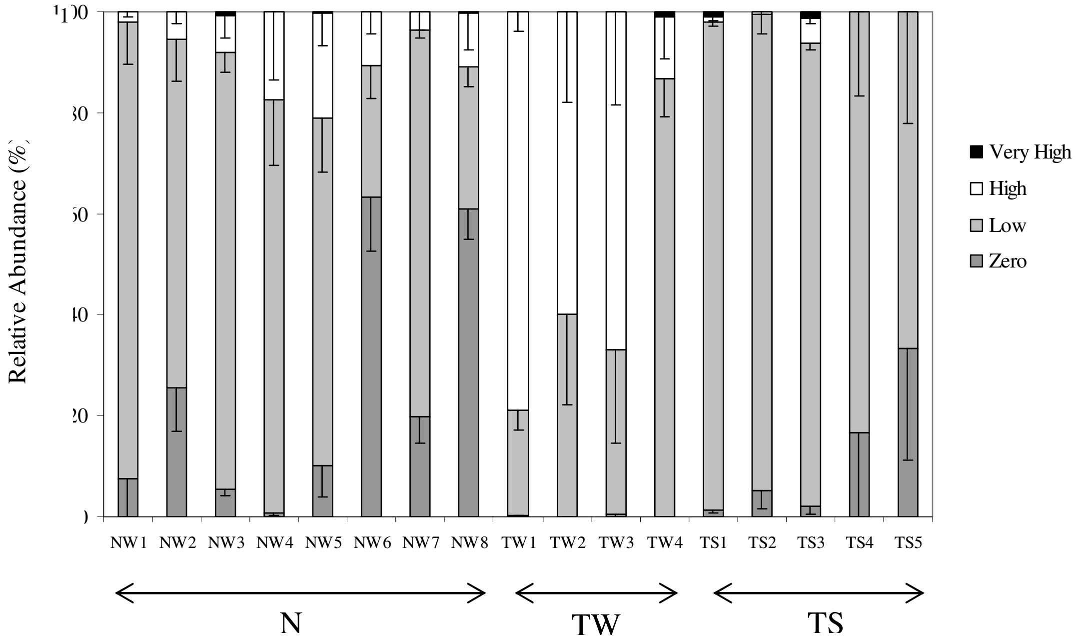


Fig. 5.10: Mean relative abundances of taxa within functional mobility groups for natural wetlands (NW), ‘no tailwater’ storages (TW) and typical storages (TS) sampled in spring 2005, presented with -1 s.e. bars.

Table 5.12: Summary of one-way ANOSIM results comparing macroinvertebrate assemblage function among waterbody groups (NW = natural wetland, TW = ‘no tailwater’ storage, TS = typical storage) based on relative abundance $\text{Log}_{10}(x+1)$ data. Results are presented with FMG’s identified by SIMPER as contributing to more than 40% of the significant difference between waterbodies.

Factor	Global R	*P*	Pairwise Test	ANOSIM R value (*p* value)	Significant FMG’s (SIMPER)
Waterbody	0.462	0.001	NW v TW	0.561 (0.001)	Zero (greater proportions in NW)
			NW v TS	0.258 (0.001)	Zero (greater proportions in NW)
			TW v TS	0.689 (0.001)	High (greater proportions in TW)

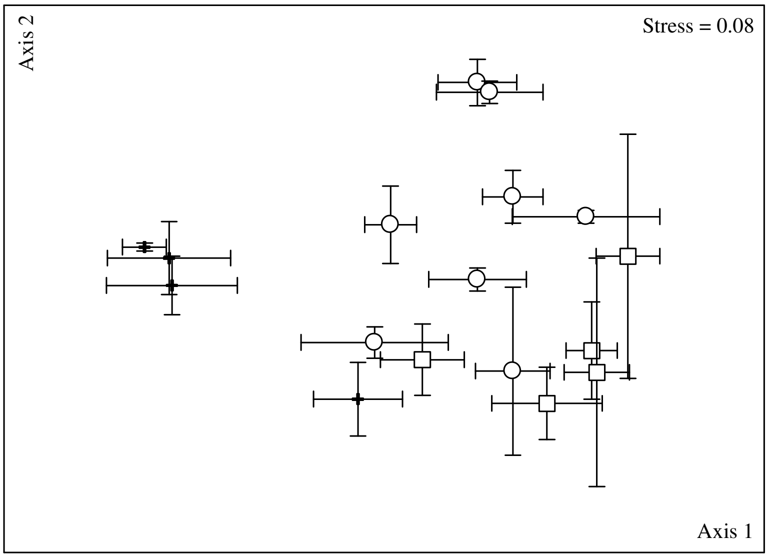


Fig. 5.11: Non-metric multi-dimensional scaling plots for macroinvertebrate assemblage function based on $Log_{10}(x+1)$ data standardised for relative abundance, for natural wetlands (o), 'no tailwater' storages (+) and typical storages (□). Samples collected from each waterbody on each sampling occasion are represented as centroids (mean x-y coordinates for n=3 samples) ± 1 s.e. bars.

5.3.2.3 Temporal patterns in macroinvertebrate assemblage composition

The pooled data for both typical storages and natural wetlands over all six temporal sampling occasions (November 2005 (T2) to February 2007 (T6)) comprised 17,006 individuals from 78 taxa. Four of these taxa were specific to typical storages and 24 taxa were specific to natural wetlands, while 50 taxa were common to both waterbody types. In contrast to the spatial study, typical storages had higher abundances than natural wetlands, with mean number of individuals per sample of 164 and 141, respectively. However, they were still less diverse; 54 taxa compared with 74 taxa in natural wetlands. Five samples, all from typical storages, did not contain any macroinvertebrate taxa (February 2006 – TS8 Rep 3 and TS7 Rep 2 & 3; March 2006 – TS8 Rep 1 & 2).

From 51 samples collected in natural wetlands, insects contributed 86% of the taxa and 78% of the individuals (Table 5.13). The richest and most abundant of all the invertebrate orders were Diptera (38% of taxa and 58% of individuals) and Hemiptera (12% of taxa and 17% of individuals). After Hemiptera, Coleoptera and Odonata were the next richest orders (both contributing 11%) but each accounted for less than 1% of the total individuals. Other abundant groups included Bryozoa, Culicidae, *Procladius* sp. (Chironomidae) and other unidentified tiny chironomids, and *Micronecta* sp. (Corixidae). Sixteen taxa occurred only once.

All four mobility groups were represented in natural wetlands, with abundances and species richness dominated by low mobility taxa (60% and 62%, respectively) (Table 5.13). Zero mobility taxa were the next most abundant and richest group contributing 22% to the total individuals and 14% to the total number of taxa. The least abundant was the very high mobility group making up less than 1% of the total individuals. However, the same number of taxa was found in the high mobility group as the very high mobility group even though it accounted for considerably more of the individuals collected (17%).

Table 5.13: Taxa and their contribution to the macroinvertebrate assemblage as a percentage of the total individuals and a percentage of the total taxa in natural wetlands and typical storages sampled across the six temporal sampling times (November 2005 – February 2007).

	Natural Wetlands		**Typical Storages**	
	% Individuals	% Taxa	% Individuals	% Taxa
Taxa				
Crustacea	2.38	2.70	0.33	3.70
Gastropoda	0.33	4.05	0.06	5.56
Oligochaeta	1.19	1.35	0.11	1.85
Nemertea	0.46	1.35	0.00	0.00
Nematoda	0.35	1.35	4.37	1.85
Bryozoa	11.43	1.35	0.15	1.85
Cnidaria	6.28	1.35	0.36	1.85
Insecta	77.58	86.49	94.62	83.33
Lepidoptera	0.03	1.35	0.03	1.85
Coleoptera	0.70	10.81	0.61	12.96
Diptera	57.77	37.84	38.85	37.04
Ephemeroptera	1.19	5.41	0.41	3.70
Hemiptera	17.15	12.16	54.50	11.11
Odonata	0.46	10.81	0.09	9.26
Plecoptera	0.04	1.35	0.04	1.85
Trichoptera	0.24	6.76	0.09	5.56
Mobility Groups				
Zero	22.42	13.51	5.38	16.67
Low	59.70	62.16	39.48	57.41
High	17.15	12.16	54.50	11.11
Very high	0.73	12.16	0.64	14.81

In typical storages, insects accounted for 83% of the taxa and 95% of the individuals collected from 60 samples. Similar to natural wetlands, Diptera and Hemiptera were the most abundant orders (39% and 55% respectively) and two of the richest orders (37% and 11% respectively). Coleoptera was the second richest order (13%) but accounted for less than 1% of the individuals. Unlike the natural wetland sites, Bryozoa made up less than 1% of the individuals. The single most abundant taxon was *Micronecta* sp. (Corixidae), totalling 5,284 individuals. Other abundant taxa included *Cladopelma* sp. and *Cricotopus* sp. (Chironomidae) and other unidentified tiny chironomids, and the Nematoda. Fifteen taxa occurred only once.

All four mobility groups were also found at typical storages (Table 5.13). However, in contrast to natural wetlands, abundances were dominated by high mobility taxa (55%), followed by low (39%), zero (5%) and very high mobility taxa (< 1%). The richness of the mobility groups was similar to that found at natural wetlands with low mobility taxa representing 57% of the taxa, followed by the zero (17%), very high (15%) and high mobility taxa (11%).

On each sampling occasion, typical storages were less diverse than natural wetlands but this was most obvious in November 2005 and February and March 2006 (Fig. 5.12a). This is reflected in Shannon diversity (H') and species richness (d) which were both also higher in natural wetlands on each sampling occasion (Fig. 5.12d and e). Across all sampling times there was a significant difference between the two waterbody types in all summary variables (*t*-test, $p < 0.001$; Kruskal-Wallis, $p < 0.001$) with the exception of Pielou's evenness (J') (Kruskal-Wallis, $p > 0.05$).

When considering the natural wetland sites separately, there was a significant difference between the six sampling times in mean number of taxa (S) ($F_{5,45} = 3.273$, $p < 0.05$), mean species richness (d) ($F_{5,45} = 5.599$, $p < 0.001$) and mean Shannon diversity (H') ($F_{5,45} = 3.542$, $p < 0.01$). Significantly higher numbers of taxa (S) were evident in November 2005 compared with all other sampling times apart from March 2006 and significantly higher species richness (d) in November 2005 compared with all other sampling occasions. There was significantly higher diversity in November 2005 compared with December 2006 and February 2007 (Fig. 5.12). No significant temporal variation was detected in the other three summary measures.

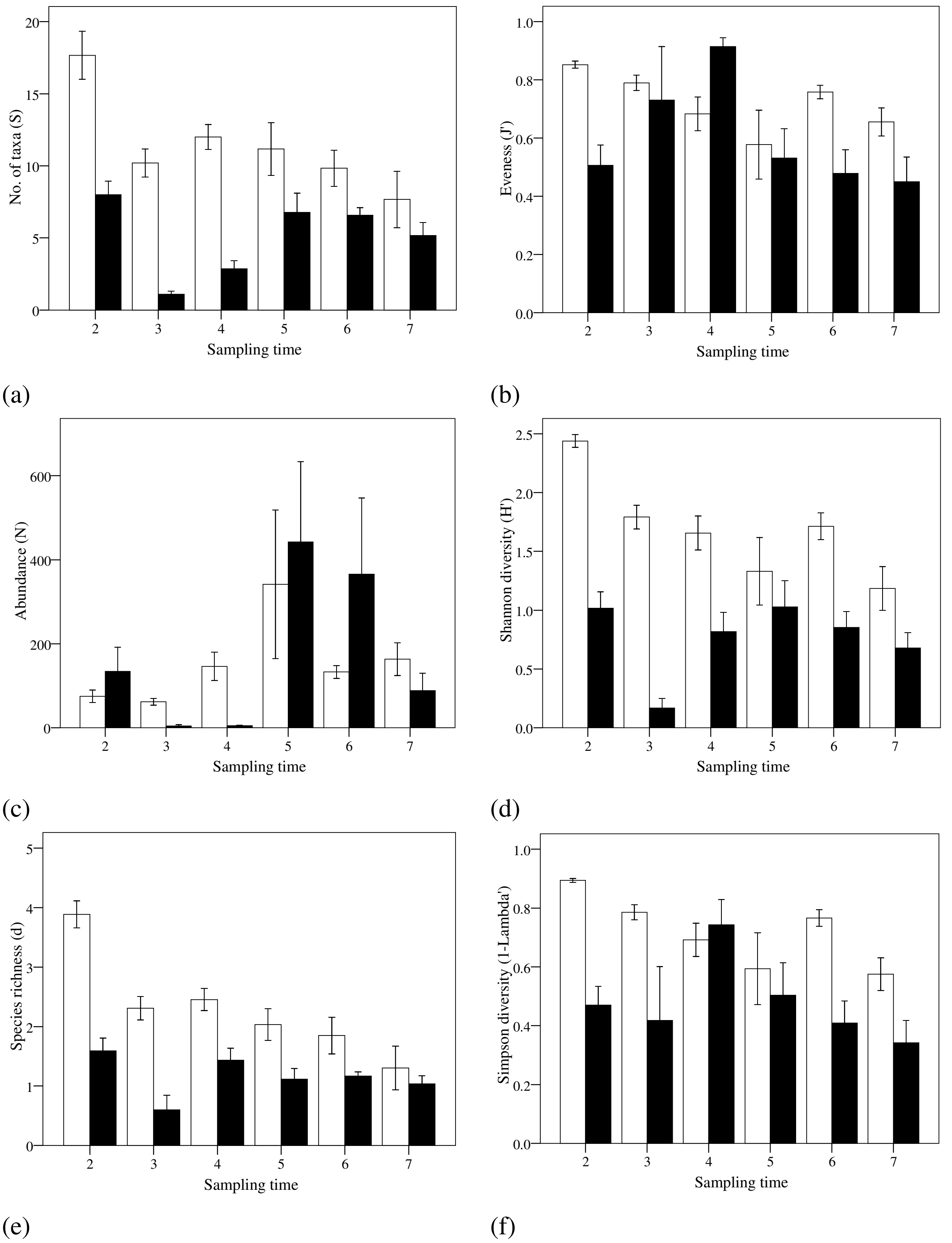


Fig. 5.12: Mean number of taxa, abundance, richness, evenness, Shannon diversity and Simpson diversity (± s.e.) of macroinvertebrates across all sampling times for natural wetlands (white bars) and typical storages (black bars). 2 = November 2005, 3 = February 2006, 4 = March 2006, 5 = December 2006, 6 = January 2007, 7 = February 2007.

There was a significant difference in all diversity measures in typical storages between the six temporal sampling times (one-way ANOVA for all measures, $p < 0.05$; see Table 5.14). March 2006 stands out with lower abundance (N) and number of taxa (S) compared with other sampling times but with increased evenness (J') (Fig. 5.12). There was lower mean abundance (N) in March and December 2006 compared with other sampling times (Fig. 5.12).

Table 5.14: F statistics from one-way ANOVA tests on all diversity measures in typical storages across the six temporal sampling times (November 2005 - February 2007).

Diversity measure	***Degrees of freedom***	***F***	***P***
Mean number of taxa (S)	5,54	13.584	0.000
Mean species richness (d)	5, 44	2.721	0.032
Mean Pielou's Eveness (J')	5, 42	5.281	0.001
Mean Shannon diversity (H')	5, 54	4.284	0.002
Mean Simpson diversity (1-Lambda')	5, 44	2.460	0.048
Mean log of abundance (N)	5, 49	18.623	0.000

Highly significant spatial and temporal differences in absolute abundance, relative abundance and presence/absence patterns were detected across all times using ANOSIM and two-way crossed ANOSIM (Appendix 15-19). Significant temporal and spatial variation in assemblage composition between sampling sites was evident (Fig. 5.13, see Appendix 20 for ordination plots based on presence/absence data and absolute abundance data). In general, there appeared to be less variation within natural wetlands across sampling occasions compared with typical storages (see error bars in Fig. 5.13). Samples from both waterbody types taken in November 2005 appeared to be quite distinct from samples collected on other dates, especially in relation to natural wetland samples. Both waterbody types followed a similar trajectory through ordination space over time, moving from a position high on Axis 1 and low on Axis 2 in November 2005 in a clockwise direction to a position high on Axis 2 and central on Axis 1 in February 2007.

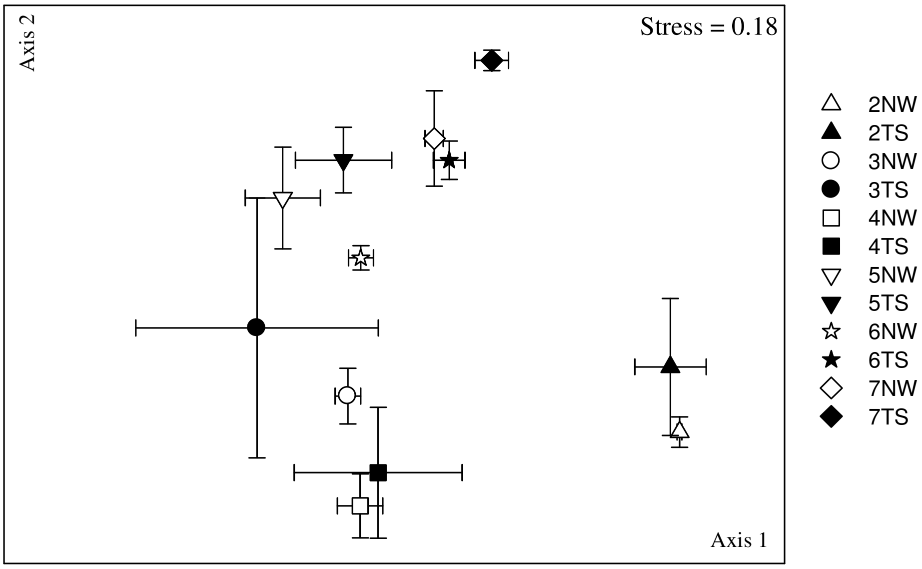


Fig. 5.13: Non-metric multi-dimensional scaling plots for macroinvertebrate assemblage structure based on $\text{Log}_{10}(x+1)$ standardised for relative abundance data, across all temporal sampling times for natural wetlands (open) and typical storages (closed). Samples collected from each waterbody type on each sampling occasion are represented as centroids (mean x-y coordinates) ± 1 s.e. bars. 2 = November 2005, 3 = February 2006, 4 = March 2006, 5 = December 2006, 6 = January 2007, 7 = February 2007. NW = natural wetlands and TS = typical storage.

At all times apart from December 2006 and February 2007, similarity within natural wetlands was higher than within typical storages (Table 5.15). As suggested by ordination, similarity was lowest within each waterbody type in February and March 2006. Similarity between natural wetlands and typical storages increased over time (Table 5.16) as water levels decreased. Taxa contributing to differences between the two waterbody types in November 2005 were very different to those taxa contributing to pairwise differences at other sampling times (Table 5.16). Four taxa, *Anisops* sp. (Notonectidae), *Procladius* sp. (Chironomidae) and other unidentified tiny chironomids and *M. australiense* (Palaemonidae) were identified as major contributors to differences between waterbody types on other sampling occasions (Table 5.16).

Table 5.15: Summary of SIMPER results for similarity within waterbody types (NW = natural wetland, TS = typical storage) on each different sampling occasion (2 = November 2005, 3 = February 2006, 4 = March 2006, 5 = December 2006, 6 = January 2007, 7 = February 2007). Based on $\text{Log}_{10}(x+1)$ data standardised for relative abundance listing taxa contributing > 10% to the similarity of the waterbodies.

NW			**TS**		
Time	% Similarity	Contributing taxa	Time	% Similarity	Contributing taxa
2	68	*Cloeon* sp. (16%), *Wundacaenis* sp. (12%), *Harnischia* sp. (10%)	2	24	*Cricoptopus* sp. (27%), *Nanocladius* sp. (23%), *Micronecta* sp. (20%), *Agraptocorixa* sp. (10%)
3	44	*Tiny Chironomids* (30%), *Procladius* sp. (24%),	3	5	*Anisops* sp. (58%), *M. australiense* (26%), Chaoboridae (15%)
4	35	*Parachironomus* sp. (19%), tiny chironomids (14%), Bryozoa (14%)	4	12	*Dicroptendipes* sp. (40%), *Cricotopus* sp. (15%), Chaoboridae (11%)
5	35	*Anisops* sp. (30%), tiny chironomids (25%), Culicidae (12%)	5	46	*Micronecta* sp. (31%), *Coelopynia* sp. (20%), tiny chironomids (20%), *Cladopelma* sp. (15%)
6	69	Tiny chironomids (21%), *Procladius* sp. (20%), *Anispos* sp. (17%), *Micronecta* sp. (16%), *Cladopelma* sp. (16%)	6	53	*Micronecta* sp. (64%), *Procladius* sp. (11%)
7	54	*Micronecta* sp. (51%), *Anisops* sp. (19%), Culicidae (11%)	7	63	*Micronecta* sp. (67%), *Anisops* sp. (22%)

Table 5.16: Summary of SIMPER results of dissimilarity between waterbody types (NW = natural wetland) across sampling occasions (2 = November 2005, 3 = February 2006, 4 = March 2006, 5 = December 2006, 6 = January 2007, 7 = February 2007). Based on $\text{Log}_{10}(x+1)$ data standardised for relative abundance listing taxa contributing > 5% to the dissimilarity between waterbodies.

Time	**Comparison**	**% dissimilarity**	**Contributing taxa**
All times	NW v. TS	81	*Micronecta* sp. (9.0%), Tiny Chironomids (8.2%), *Procladius* sp. (7.2%), *Anisops* sp. (6.3%), *Cladopelma* sp. (5.3%)
2	NW v. TS	76	*Cloeon* sp. (7.5%), *Cricotopus* sp. (6.9%), *Nanocladius* sp. (6.8%), *Harnischia* sp. (6.5%), *Wundacaenis* sp. (5.8%), Ochthebiinae subtribe (5.4%), Bryozoa (5.3%)
3	NW v. TS	93	Tiny Chironomids (12.7%), *Procladius* sp. (11.4%), *M. australiense* (7.2%), *Anisops* sp. (7.0%), Bryozoa (6.4%), *Parachironomus* sp. (5.9%), *Cladopelma* sp. (5.6%)
4	NW v. TS	85	Bryozoa (8.0%), *Parachironomus* sp. (7.4%), Tiny Chironomids (6.9%), *Dicrotendipes* sp. (6.8%), Chaoboridae (6.5%), *Procladius* sp. (6.0%), *M. australiense* (5.5%)
5	NW v. TS	74	*Micronecta* sp. (12.1%), *Anisops* sp. (11.6%), Culicidae (10.0%), Tiny Chironomids (9.2%), *Cladopelma* sp. (7.6%), *Coelopynia* sp. (5.5%), Nematoda (5.2%), *Cladotanytarsus* sp. (5.1%)
6	NW v. TS	56	Tiny Chironomids (13.4%), *Anisops* sp. (12.3%), *Micronecta* sp. (9.6%), *Procladius* sp. (9.0%), *Cladopelma* sp. (8.7%), Chaoboridae (5.8%)
7	NW v. TS	50	Culicidae (13.4%), *Procladius* sp. (11.9%), *E. australis* (10.7%), *Anisops* sp. (10.3%), *Cladopelma* sp. (7.8%), *M. australiense* (7.2%), Oligochaeta (5.4%), Tiny Chironomids (5.1%)

Table 5.17: Summary of BIO-ENV results based on Spearman rank correlations (r_s) between macroinvertebrate assemblage structure, habitat availability variables and water chemistry variables. Results presented for best possible solution only.

		Transformation	**Best Variable Combination**	**Correlation**
Site	**Habitat Availability Variables**			
NW	Species Diversity	Presence/absence	SN, AQV	0.296
	Absolute Abundance	$\text{Log}_{10}(x+1)$	BB, AQV	0.296
	Relative Abundance	$\text{Log}_{10}(x+1)$	SN, BB, AQV	0.314
TS	Species Diversity	Presence/absence	BB	0.356
	Absolute Abundance	$\text{Log}_{10}(x+1)$	BB	0.342
	Relative Abundance	$\text{Log}_{10}(x+1)$	BB	0.362
NW&TS	Species Diversity	Presence/absence	BB, AQV	0.340
	Absolute Abundance	$\text{Log}_{10}(x+1)$	BB, AQV	0.344
	Relative Abundance	$\text{Log}_{10}(x+1)$	BB, AQV	0.371
	Water Chemistry Variables *			
NW	Species Diversity	Presence/absence	Turbidity, Water temperature	0.168
	Absolute Abundance	$\text{Log}_{10}(x+1)$	Conductivity, Water temperature	0.239
	Relative Abundance	$\text{Log}_{10}(x+1)$	Water temperature	0.244
TS	Species Diversity	Presence/absence	Turbidity	0.020
	Absolute Abundance	$\text{Log}_{10}(x+1)$	Turbidity	0.042
	Relative Abundance	$\text{Log}_{10}(x+1)$	Turbidity	0.038
NW&TS	Species Diversity	Presence/absence	Water temperature	0.030
	Absolute Abundance	$\text{Log}_{10}(x+1)$	Water temperature	0.024
	Relative Abundance	$\text{Log}_{10}(x+1)$	Water temperature	0.032

SN = snags, BB = bare bank, AQV = aquatic vegetation, *Results for December 2006 removed.

There were no strong associations between the environmental data and observed macroinvertebrate assemblage structure (Table 5.17). Available habitat accounted for more of the variability than water chemistry but all r_s values were low (< 0.4). Water chemistry explained more of the variability at natural wetlands than at typical storages but again the r_s values were very low (< 0.25).

5.3.2.4 Temporal variation in macroinvertebrate mobility

Over the course of the 2005/06 and 2006/07 irrigation seasons there were notable changes in the presence of different macroinvertebrate mobility groups. Initially, assemblages in both natural wetlands and typical storages were once again dominated by low and zero mobility taxa (Fig. 5.14). Low and zero mobile taxa combined had site means of greater than 75% of the abundance in natural wetlands and, with the exception of Boolarwell S1 (TS8), greater than 72% of the abundance at typical storages. However, from December 2006 (T5), taxa from the high mobility group became more common in both waterbody types. By February 2007 highly mobile taxa comprised more than half the assemblage in natural wetlands (site means > 50%) and more than three quarters of the assemblage in typical storages (site means > 88%).

There were significant differences between waterbody types (R = 0.314, $p = 0.001$) and sampling times (R = 0.352, $p = 0.001$) (Table 5.18 and Appendix 21 and 22). This is also apparent in the ordination based on FMG proportions (Fig. 5.15). The greater variation in typical storages between sampling times compared with natural wetlands is evident in the ordination (Fig. 5.15), however, there appeared to be greater variation between typical storages at particular sampling times than in natural wetlands (see error bars in Fig. 5.15).

At all sampling times similarity within the natural wetlands was higher than within the typical storages (Table 5.19) and there was no obvious pattern in the similarity between natural wetlands and typical storages over time (Table 5.20). The similarity between the two waterbody types was highest in November 2005 (T2) and lowest in February 2006 (T3). Across all sampling times there was relatively low similarity between the natural wetlands and the typical storages. High and zero mobility taxa contributed most to the differences.

Table 5.18: Summary of ANOSIM results comparing macroinvertebrate assemblage function (based on $\log_{10}(x+1)$ standardised for relative abundance) among waterbody types (NW = natural wetlands, TS = typical storages) and sampling times occasions (2 = November 2005, 3 = February 2006, 4 = March 2006, 5 = December 2006, 6 = January 2007, 7 = February 2007). * $p = 0.001$, ** $p < 0.01$, ***$p < 0.05$.

ANOSIM	Waterbody	Transformation	Factor	Global R	*p*	Pairwise tests
One-way	NW	$\text{Log}_{10}(x+1)$Std	Time	0.351	0.001	2-4, 4-6, 4-7* 2-3, 3-5, 3-7, 4-5** 2-6, 2-7, 3-6***
	TS	$\text{Log}_{10}(x+1)$Std	Time	0.353	0.001	2-6, 3-6, 4-6, 4-7* 2-7, 6-7** 3-7, 5-6, 5-7***
Two-way crossed	NW & TS	$\text{Log}_{10}(x+1)$Std	Time	0.352	0.001	2-6, 2-7, 3-6, 3-7, 4-5, 4-6, 4-7, 6-7* 2-3, 2-4, 3-5, 5-6** 3-4, 5-7***
	NW & TS	$\text{Log}_{10}(x+1)$Std	Waterbody	0.314	0.001	

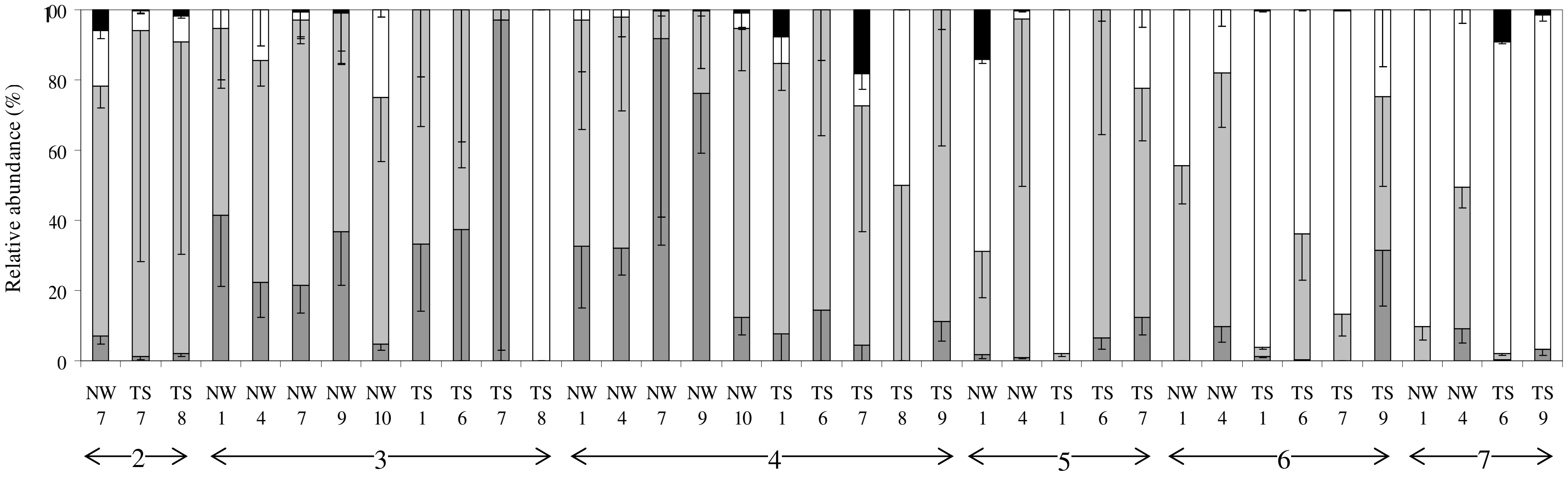


Fig. 5.14: Mean relative abundances of taxa within FMG's (black = very high, white = high, light grey = low, dark grey = zero) for natural wetlands (NW) and typical storages (TS) sampled during the 2005/06 and 2006/07 irrigation seasons, presented with -1 s.e. bars. Sampling times: 2 = November 2005, 3 = February 2006, 4 = March 2006, 5 = December 2006, 6 = January 2007, 7 = February 2007.

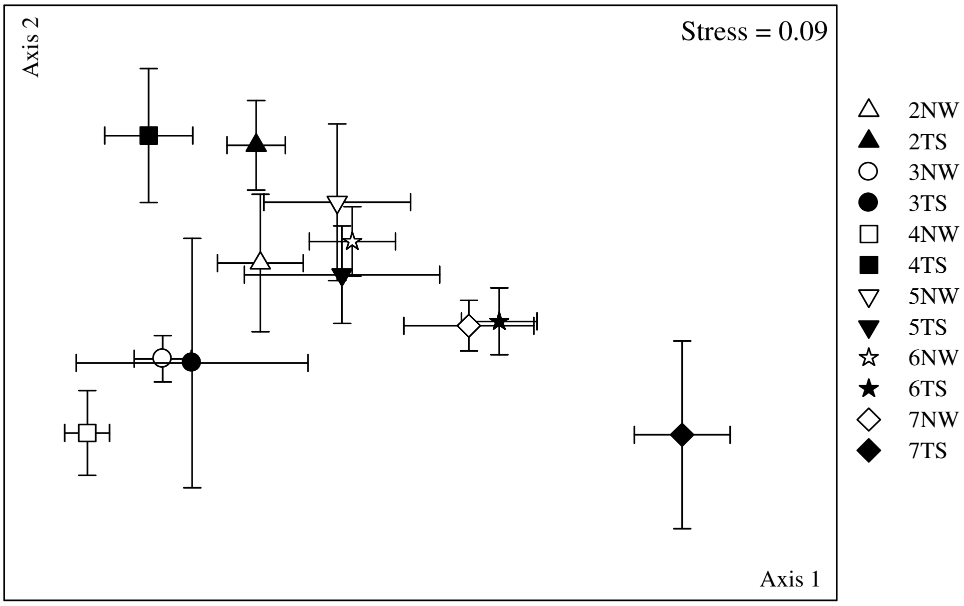


Fig. 5.15: Non-metric multi-dimensional scaling plots for macroinvertebrate assemblage function based on $Log_{10}(x+1)$ data standardised for relative abundance, for natural wetlands (NW) and typical storages (TS). Samples collected from each waterbody on each sampling occasion are represented as centroids (mean x-y coordinates for $n \geq 3$ samples) ± 1 s.e. bars. 2 = November 2005, 3 = February 2006, 4 = March 2006, 5 = December 2006, 6 = January 2007, 7 = February 2007.

Table 5.19: Summary of SIMPER results for similarity in functional composition of macroinvertebrate assemblages within waterbody types (NW = natural wetland, TS = typical storage) on each different sampling occasion (2 = November 2005, 3 = February 2006, 4 = March 2006, 5 = December 2006, 6 = January 2007, 7 = February 2007). Based on $Log_{10}(x+1)$ data standardised for relative abundance listing FMG's contributing > 25% to the similarity of the waterbodies.

NW			TS		
Time	**% Similarity**	**Contributing taxa**	**Time**	**% Similarity**	**Contributing taxa**
2	91	Low (43%)	2	83	Low (62%), high (26%)
3	83	Low (54%), zero (33%)	3	37	Low (65%), zero (27%)
4	80	Low (50%), zero (45%)	4	69	Low (89%)
5	67	Low (57%), high (33%)	5	58	Low (56%), high (33%)
6	85	Low (56%), high (41%)	6	81	High (63%), low (35%)
7	80	High (59%), low (36%)	7	75	High (79%)

Table 5.20: Summary of SIMPER results of dissimilarity in functional composition of macroinvertebrate assemblages between waterbody types (NW = natural wetland, TS = typical storage) across sampling occasions (2 = November 2005, 3 = February 2006, 4 = March 2006, 5 = December 2006, 6 = January 2007, 7 = February 2007). Based on $Log_{10}(x+1)$ data standardised for relative abundance listing FMG's contributing >25% to the dissimilarity between waterbodies.

Time	Comparison	% dissimilarity	Contributing taxa
All times	NW v. TS	39	High (34%), zero (33%)
2	NW v. TS	16	Zero (37%), very high (34%)
3	NW v. TS	45	Zero (36%), high (31%), low (31%)
4	NW v. TS	35	Zero (50%)
5	NW v. TS	35	High (38%), low (29%)
6	NW v. TS	23	Low (38%), zero (34%), high (27%)
7	NW v. TS	35	Low (45%), very high (27%)

5.3.2.5 EPT index values

During the spatial study, the EPT index (or in this case Ephemeroptera and Trichoptera, since Plecoptera were not found) did not differ significantly between waterbody types ($F_{2,48} = 3.166$, $p > 0.05$). ‘No tailwater’ storages and natural wetlands supported similar assemblages of EPT taxa while typical storages supported lower numbers of EPT taxa (Table 5.21). The variation between samples from ‘no tailwater’ storages was relatively high compared to the other two waterbody types; most of this variation came from Barra Storage (TW1) which had a comparatively high EPT index. When this site was removed the mean EPT index value for ‘no tailwater’ storages dropped to 0.56 (± 0.242 s.e.), which was comparable to typical storages.

Table 5.21: EPT index values calculated from spatial sampling for each waterbody type.

	Natural	No TW	Typical
No of Samples	24	12	15
Max	5	8	2
Min	0	0	0
Mean	1.80	1.92	0.53
s.e.	0.307	0.763	0.192

From 2005 through 2007 there was considerable variation in the EPT index both spatially and temporally. EPT taxa were collected from natural wetlands on five out of the 6 sampling occasions while they were only found in typical storages in November 2005 and January 2007 (Table 5.22). In February 2007, EPT taxa were not collected from any of the sites sampled. Over all sampling occasions, EPT taxa were collected from all natural wetlands apart from Kerry’s Creek (NW9) while they were only found at 3 typical storages.

The EPT index differed significantly between November 2005 (T2) and all other times in both waterbody types (Table 5.23 and Appendix 23). This was, most likely, due to the fact that only three sites were sampled in November 2005 and 8 of the 9 replicates contained EPT taxa. Across all sampling occasions there was a significant spatial difference in EPT index values (Table 5.21 and Appendix 24).

Table 5.22: EPT index values calculated from temporal sampling for each waterbody type. Times: 2 = November 2005, 3 = February 2006, 4 = March 2006, 5 = December 2006, 6 = January 2007, 7 = February 2007.

	Time 2	**Time 3**	**Time 4**	**Time 5**	**Time 6**	**Time 7**
Natural Wetland						
Number of sites	1	5	5	2	2	2
Number of samples	3	15	15	6	6	6
Maximum EPT value	4	1	2	2	3	0
Minimum EPT value	2	0	0	0	0	0
Mean EPT value	3.00	0.20	0.53	0.50	0.50	0
s.e.	0.58	0.11	0.19	0.34	0.50	0
Number of sites with EPT taxa	1	2	3	2	1	0
Number of samples with EPT taxa	3	3	6	2	1	0
Typical Storages						
Number of sites	2	4	5	3	4	2
Number of samples	6	12	15	9	12	6
Maximum EPT value	2	0	0	0	1	0
Minimum EPT value	0	0	0	0	0	0
Mean EPT value	1.33	0	0	0	0.08	0
s.e.	0.33	0	0	0	0.08	0
Number of sites with EPT taxa	2	0	0	0	1	0
Number of samples with EPT taxa	5	0	0	0	1	0

Table 5.23: Summary of ANOSIM results comparing EPT scores among waterbody types (NW = natural wetlands, TS = typical storages) and sampling times. 2 = November 2005, 3 = February 2006, 4 = March 2006, 5 = December 2006, 6 = January 2007, 7 = February 2007.

Waterbody	**Factor**	**Global R**	***p***	**Significant pairwise tests**
NW	Time	0.106	0.034	2-3 (0.001), 2-4 (0.004), 2-5 (0.024), 2-6 (0.036), 2-7 (0.012)
TS	Time	0.145	0.001	2-3 (0.001), 2-4 (0.001), 2-5 (0.002), 2-6 (0.007), 2-7 (0.015)
NW & TS	Waterbody	0.071	0.006	
NW & TS	Time	0.127	0.001	2-3 (0.001), 2-4 (0.001), 2-5 (0.001), 2-6 (0.001), 2-7 (0.002)

5.3.2.6 SIGNAL scores

To set the borders on the SIGNAL score quadrant I used actual data from the spatial study and the boundaries were set such that the majority of the natural wetlands fell within quadrant one (Fig. 5.16). This left the majority of the storages spread between Quadrant 3 and Quadrant 4. Only 3 samples were positioned in Quadrant 2. There did not appear to be any difference in the position on the plot between typical storages and 'no tailwater' storages. There was no significant difference in SIGNAL 2 scores between the three waterbody types ($F_{2,48} = 0.235$, $p > 0.5$).

During the temporal study, five samples, all from typical storages, contained no macroinvertebrates at all. These replicates were given a SIGNAL 2 score of zero. As with the spatial sampling results the quadrant boundaries were set so that most natural wetlands fell within quadrant 1 (Fig. 5.17). This left the majority of typical storages in Quadrant 3 but also 3 natural wetland samples from February 2007 in quadrant 4. Both spatial and temporal differences in SIGNAL 2 scores were significant (Table 5.24 and Appendix 25 and 26).

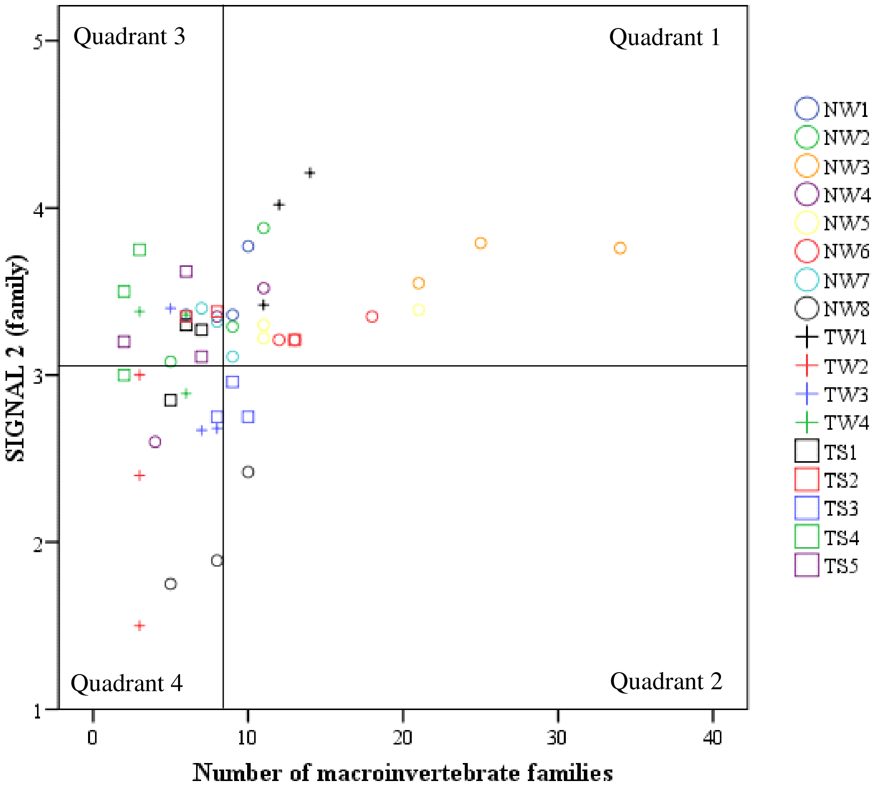


Fig. 5.16: A bi-plot of the SIGNAL 2 scores and number of macroinvertebrate taxa for the spatial sampling programme. O = natural wetlands, + = 'no tailwater' storages, □ = typical storages.

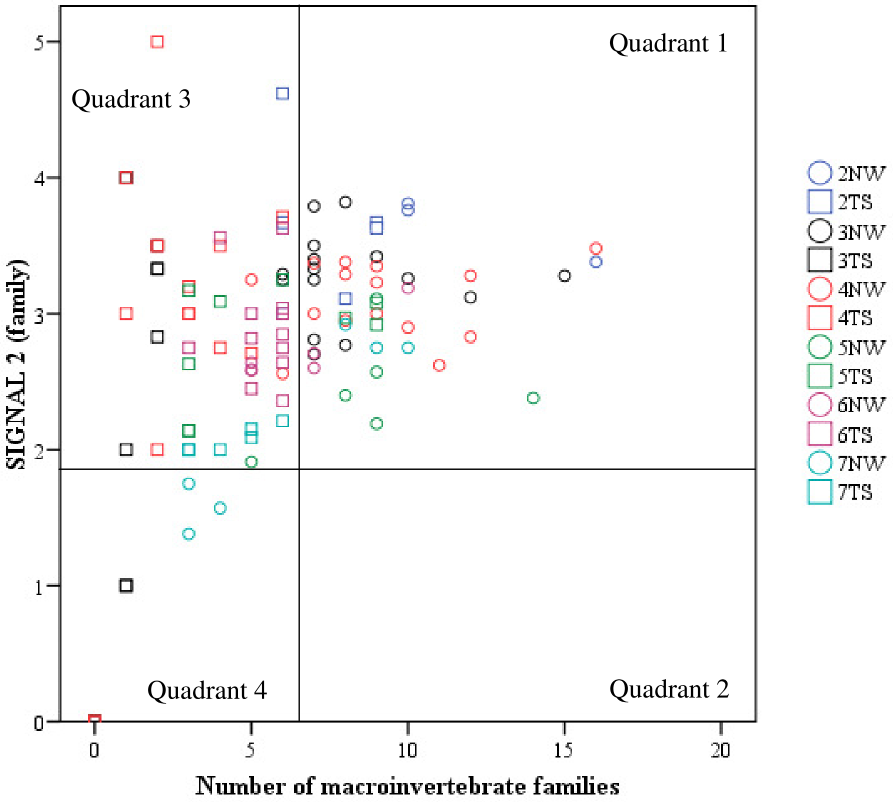


Fig. 5.17: A bi-plot of the SIGNAL 2 scores and number of macroinvertebrate taxa for the temporal sampling programme. O = natural wetlands, □ = typical storages and numbers refer to sampling time; 2 = November 2005, 3 = February 2006, 4 = March 2006, 5 = December 2006, 6 = January 2007, 7 = February 2007.

Table 5.24: Summary of ANOSIM results comparing SIGNAL scores among waterbody types (NW = natural wetlands, TS = typical storages) and sampling times. 2 = November 2005, 3 = February 2006, 4= March 2006, 5 = December 2006, 6 = January 2007, 7 = February 2007.

Waterbody	Factor	Global R	*p*	Significant pairwise tests
NW	Time	0.344	0.001	2-4 (0.018), 2-5 (0.012), 2-6 (0.012), 2-7 (0.024), 3-5 (0.001), 3-6 (0.002), 3-7 (0.001), 4-5 (0.002, 4-6 (0.01), 4-7 (0.003), 5-6 (0.052), 6-7 (0.035)
TS	Time	0.128	0.007	2-5 (0.032), 2-6 (0.01) 2-7 (0.002), 3-6 (0.002), 4-7 (0.072), 5-7 (0.009), 6-7 (0.001)
NW & TS	Waterbody	0.133	0.004	
NW & TS	Time	0.228	0.001	2-5 (0.003), 2-6 (0.002), 2-7 (0.001), 3-5 (0.001), 3-6 (0.001), 3-7 (0.002), 4-5 (0.004), 4-6 (0.02), 4-7 (0.001), 5-7 (0.004), 6-7 (0.001)

5.3.3 Zooplankton

Pooled data over the five sampling occasions from February 2006 to February 2007 comprised of 928,365 individuals from 28 taxa. Zooplankton were present in every sample collected. The majority of taxa were rotifers (21) along with 3 copepods and 4 cladocerans. The most abundant taxon was copepod nauplii contributing 28% to the total number of individuals (Table 5.25). *Filinia* sp., with a total density of 143,335 and contributing 15% to the total number of zooplankton, was the most abundant rotifer.

There was considerable variation in the number of individuals in each replicate sample ranging from 163 to 4,160 animals L^{-1} in natural wetlands and 56 to 10,112 animals L^{-1} in typical storages. However, in general, typical storages had less individuals per volume than natural wetlands and were less diverse, 22 taxa compared with 27 taxa. *Dicranophorus* sp. (Rotifera) was only present in one sample from typical storages while *Testudinella* sp., *Platyias* sp., *Colurella* sp., *Lepadella* sp., and *Trichotria* sp. (Rotifera) and *Bosminidae* sp. (Cladocera) were specific to natural wetlands.

From the 47 samples collected from natural wetlands, copepods were the most abundant, contributing 47% of individuals, closely followed by rotifera (46%). Cladocerans only contributed 7% of individuals. Copepod nauplii were by far the most abundant taxon in natural wetlands, with 159,740 individuals. The two most abundant rotifera taxa were *Polyarthra* sp. and *Keratella* sp. contributing 16% and 12% respectively. No other taxon contributed more than 10% to the total number of individuals.

In typical storages, rotifera accounted for 61% of individuals collected from 51 samples. Copepods and cladocerans made up the other 39% (29% and 10% respectively). Unlike natural wetlands, the most abundant taxon was *Filinia* sp. (Rotifera), contributing 30% of the individuals. Copepod nauplii only contributed 22% of individuals in typical storages. Other abundant taxa included *Hexarthra* sp. (Rotifera) and Daphnidae (Cladocera).

Table 5.25: Zooplankton taxa and percentage contribution to total abundance collected in temporal samples between February 2006 and February 2007, NW = natural wetland, TS = typical storage.

Taxon	NW	Percent of Total	TS	Percent of Total	All samples	Percent of total
Rotifera						
Bdelloidea sp.	740	0.2	720	0.2	1460	0.2
Conochilus sp.	2075	0.5	9075	1.9	11150	1.2
Filinia sp.	5055	1.1	138280	29.5	143335	15.4
Hexarthra sp.	23785	5.2	61430	13.1	85215	9.2
Testudinella sp.	80	0.0	0	0.0	80	0.0
Asplanchna sp.	2625	0.6	3595	0.8	6220	0.7
Brachionus sp.	18830	4.1	41095	8.8	59925	6.5
Keratella sp.	54070	11.8	16695	3.6	70765	7.6
Plationus sp.	560	0.1	30	0.0	590	0.1
Platyias sp.	15	0.0	0	0.0	15	0.0
Colurella sp.	45	0.0	0	0.0	45	0.0
Lepadella sp.	95	0.0	0	0.0	95	0.0
Dicranophorus sp.	0	0.0	15	0.0	15	0.0
Epiphanes sp.	11390	2.5	3845	0.8	15235	1.6
Euchlanis sp.	680	0.1	30	0.0	710	0.1
Lecane sp.	2180	0.5	905	0.2	3085	0.3
Cephalodella sp	5995	1.3	595	0.1	6590	0.7
Polyarthra sp.	74125	16.1	7125	1.5	81250	8.8
Trichocerca sp.	8280	1.8	820	0.2	9100	1.0
Trichotria sp.	70	0.0	0	0.0	70	0.0
Wolga sp.	275	0.1	245	0.1	520	0.1
Copepoda						
Nauplii	159740	34.7	104130	22.2	263870	28.4
Calanoida	14815	3.2	26145	5.6	40960	4.4
Cyclopoida	42680	9.3	8265	1.8	50945	5.5
Cladocera						
Bosminidae	30	0.0	0	0.0	30	0.0
Chydoridae	370	0.1	20	0.0	390	0.0
Daphniidae	31315	6.8	45305	9.7	76620	8.3
Macrothricidae	20	0.0	60	0.0	80	0.0
Total	459940		468425		928365	
Total Taxa	27		22		28	

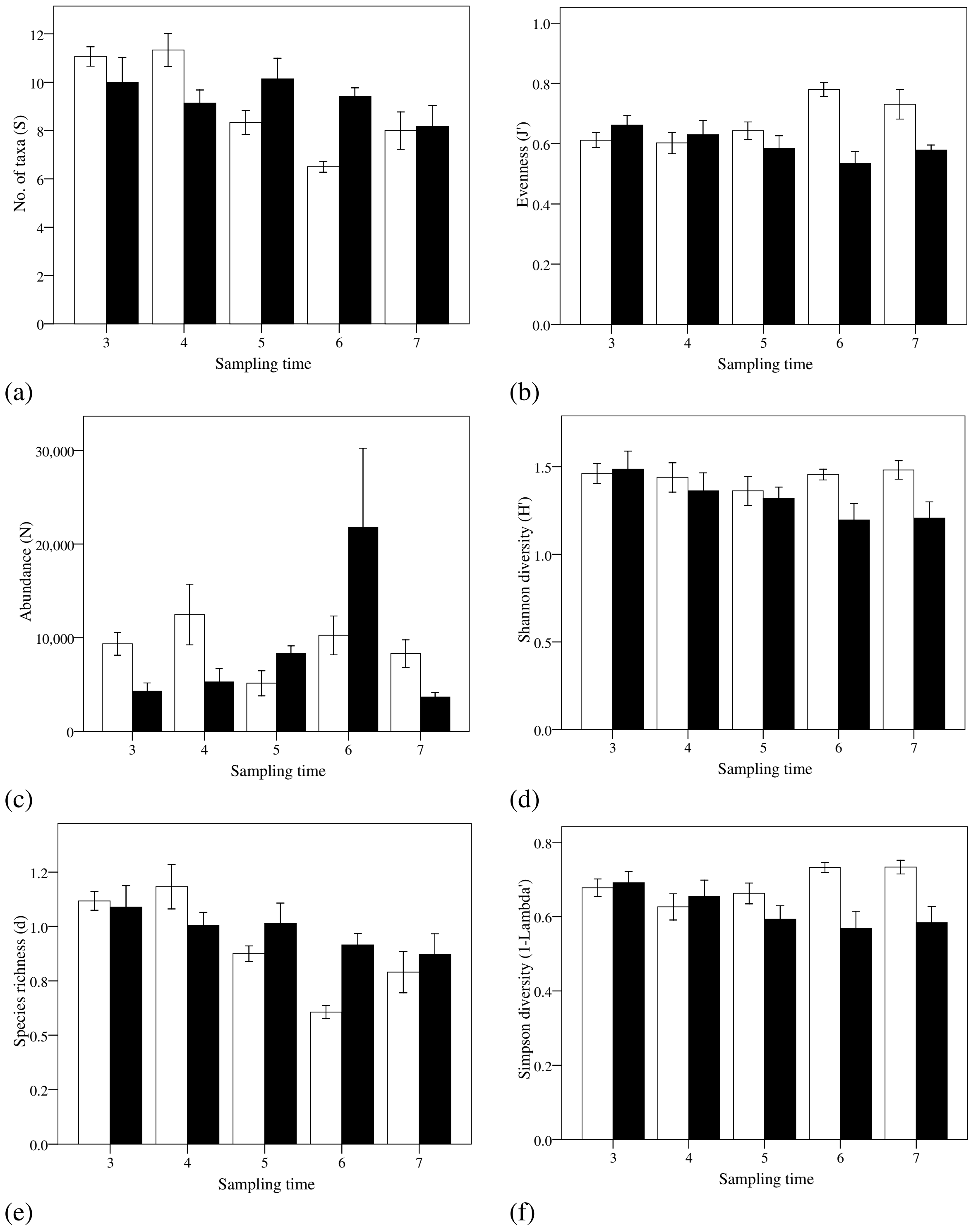


Fig. 5.18: Mean number of taxa, abundance, richness, evenness, Shannon diversity and Simpson diversity (± s.e.) of zooplankton across all sampling times for natural wetlands (clear bars) and typical storages (black bars). 3 = February 2006, 4 = March 2006, 5 = December 2006, 6 = January 2007, 7 = February 2007.

There was a significant difference across all times between the two waterbody types in both mean log of abundance (N) and mean Shannon diversity (H') ($p < 0.05$). Higher diversity and abundance were recorded in natural wetlands. When considering the natural wetlands separately, ANOVA detected significant differences between times in number of taxa (S) ($F_{4,42} = 10.668$, $p < 0.001$), the square root of species richness (d) ($F_{4,42} = 9.233$, $p < 0.001$) and Pielou's evenness (J') ($F_{4,42} = 4.162$, $p < 0.01$). Post hoc tests suggested increased number of taxa (S) in March 2006 compared with December 2006 and January and February 2007 (Fig. 5.18a) and higher species richness (d) in February and March 2006 compared with January and February 2007 (Fig. 5.18e). Post hoc tests also suggested lower evenness (J') in February and March 2006 in comparison with January 2007 (Fig. 5.18e). In typical storages the only summary variable that was significantly different between times was log of abundance ($F_{4,46} = 4.304$, $p <0.01$). Typical storages had increased abundance of zooplankton in January 2007 compared to all other sampling times (Fig. 5.18c), however, post hoc tests did not separate the times into different groups.

Highly significant spatial and temporal differences in zooplankton assemblages were detected using ANOSIM and two-way crossed ANOSIM (Appendix 27-31). Ordination also suggested temporal and spatial variation in zooplankton assemblages of sampling sites (Fig. 5.19). Over all sampling times, natural wetlands clustered low on Axis 2. Samples from natural wetlands in February and March 2006 were positioned high on Axis 1, while those samples from December 2006 and January and February 2007 were found low on Axis 1. Typical storages in February and March 2006 were in a similar position to natural wetlands, clustering low on Axis 2 and high on Axis 1. Position of waterbody centroids (± s.e), in December 2006 and January and February 2007, suggested greater temporal variation in zooplankton assemblage composition within typical storages than natural wetlands. Both natural wetlands and typical storages followed a similar trajectory through the ordination space over time; moving from a position low on Axis 2 and high on Axis 1 in February 2006 in an anti-clockwise direction to a position low on Axis 1 and in the middle of Axis 2 in February 2007 (Fig. 5.19).

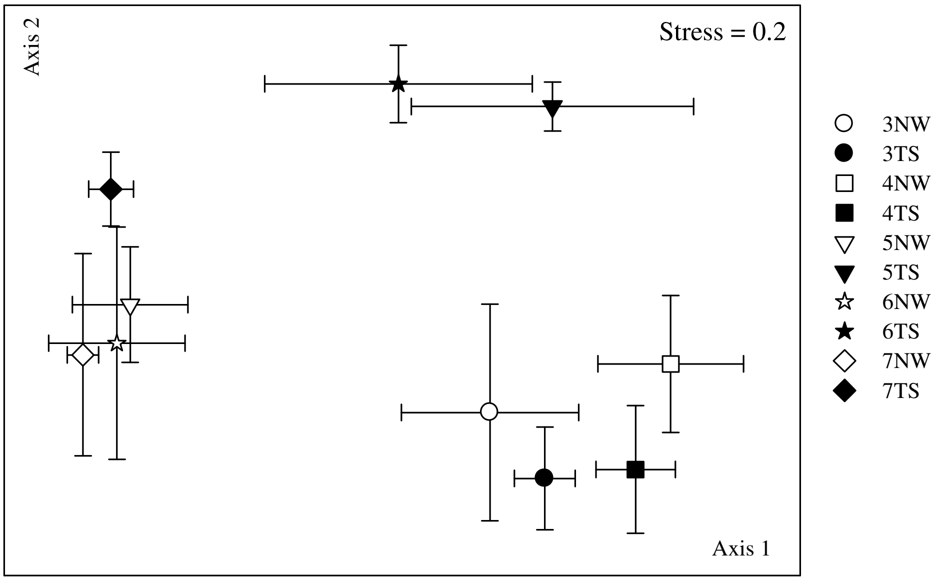


Fig. 5.19: Non-metric multi-dimensional scaling plots for zooplankton assemblage structure based on $Log_{10}(x+1)$ data standardised for relative abundance, across all temporal sampling times for natural wetlands (NW) and typical storages (TS). Samples collected from each waterbody on each sampling occasion are represented as centroids (mean x-y coordinates) ± s.e bars. 3 = February 2006, 4 = March 2006, 5 = December 2006, 6 = January 2007, 7 = February 2007.

For both natural wetlands and typical storages, similarity was highest within sites in February 2007 (Table 5.26) when water levels were at their lowest (see Fig. 5.4). The similarity between the two waterbody types was also at its highest in February 2007 (Table 5.27). However, there was no obvious trend in similarity within waterbody types or dissimilarity between waterbody types from February 2006 to February 2007. The taxa contributing to differences between natural wetlands and typical storages were similar at all sampling times. Six taxa were identified as major contributors to differences between waterbody types on all sampling occasions; Daphniidae, *Polyarthra* sp., Calanoida, *Filinia* sp., Cyclopoida, *Brachionus* sp. (Table 5.27).

Table 5.26: Summary of SIMPER results for similarity within waterbody types (NW = natural wetland, TS = typical storage) on each different sampling occasion (3 = February 2006, 4 = March 2006, 5 = December 2006, 6 = January 2007, 7 = February 2007). Based on $Log_{10}(x+1)$ data standardised for relative abundance listing taxa contributing >10% to the similarity of the waterbodies.

NW			TS		
Time	% Similarity	Contributing taxa	Time	% Similarity	Contributing taxa
3	55	Nauplii (31%) Cyclopoida (17%) *Hexarthra* sp. (15%), Daphniidae (11%)	3	67	Nauplii (32%) *Hexarthra* sp. (21%), Cyclopoida (16%)
4	57	Nauplii (28%), *Polyarthra* sp. (24%),	4	61	Nauplii (36%), *Hexarthra* sp. (28%), *Keratella* sp. (14%),
5	68	Nauplii (38%), Daphniidae (22%), Calanoida (20%)	5	59	Nauplii (24%), *Brachionus* sp. (16%), *Filinia* sp. (16%), Calanoida (10%), *Hexarthra* sp. (10%)
6	64	Nauplii (35%), Daphniidae (27%), Calanoida (11%)	6	53	Nauplii (26%), *Filinia* sp. (17%), Calanoida (16%), Daphniidae (15%),
7	68	Nauplii (34%), Daphniidae (22%), *Brachionus* sp. (16%), Cyclopoida (11%)	7	84	Nauplii (33%), Calanoida (23%), Daphniidae (20%), *Brachionus* sp. (12%)

Table 5.27: Summary of SIMPER results of dissimilarity between waterbody types (NW = natural wetland) across sampling occasions (3 = February 2006, 4 = March 2006, 5 = December 2006, 6 = January 2007, 7 = February 2007). Based on $Log_{10}(x+1)$ data standardised for relative abundance listing taxa contributing >5% to the dissimilarity between waterbodies.

Time	Comparison	% dissimilarity	Contributing taxa
All times	NW v. TS	51	*Hexarthra* sp. (10.1%), *Polyarthra* sp. (9.9%), Daphniidae (9.2%), Calanoida (9.2%), Cyclopoida (8.7%), *Keratella* sp. (8.3%), *Filinia* sp. (7.7%), *Brachionus* sp. (7.4%), Nauplii (5.6%)
3	NW v. TS	44	*Hexarthra* sp. (10.0%), Daphniidae (9.4%), *Polyarthra* sp. (8.8%), *Filinia* sp. (8.1%), *Keratella* sp. (7.7%), Cyclopoida (7.3%), *Brachionus* sp. (6.6%), *Trichocerca* sp. (6.5%), Calanoida (6.4%)
4	NW v. TS	48	*Polyarthra* sp. (12.2%), *Hexarthra* sp. (11.4%), *Keratella* sp. (10.7%), *Conochilus* sp. (6.5%), Nauplii (6.4%), Cyclopoida (6.3%), *Filinia* sp. (6.2%), Calanoida (6.1%), *Brachionus* sp. (5.6%), Daphniidae (5.4%),
5	NW v. TS	47	*Brachionus* sp. (12.8%), *Filinia* sp. (12.2%), Cyclopoida (9.9%), Calanoida (9.7%), Daphniidae (9.7%), *Hexarthra* sp. (9.0%), *Keratella* sp. (7.6%), Nauplii (7.4%), *Epiphanes* sp. (6.1%), *Polyarthra* sp. (5.2%),
6	NW v. TS	51	*Filinia* sp. (14.1%), Cyclopoida (11.5%), Daphniidae (10.9%), Calanoida (10.6%), *Polyarthra* sp. (10.0%), *Brachionus* sp. (8.3%), *Hexarthra* sp. (7.9%), Nauplii (7.6%), *Epiphanes* sp. (6.5%)
7	NW v. TS	33	Calanoida (18.0%), *Polyarthra* sp. (15.7%), Cyclopoida (15.1%), *Brachionus* sp. (8.1%), *Cephalodella* sp. (7.9%), *Epiphanes* sp. (7.6%), Daphniidae (6.1%), *Filinia* sp. (5.1%), Nauplii (5.0%),

Table 5.28: Summary of BIO-ENV results based on Spearman rank correlations (r_s) between zooplankton assemblage structure, habitat availability variables and water chemistry variables. Results presented for best possible solution only.

		Transformation	**Best Variable Combination**	**Correlation**
Site	**Habitat Availability Variables**			
NW	Species Diversity	Presence/absence	BB	0.234
	Absolute Abundance	$\text{Log}_{10}(x+1)$	BB	0.299
	Relative Abundance	$\text{Log}_{10}(x+1)$	BB	0.335
TS	Species Diversity	Presence/absence	BB	0.362
	Absolute Abundance	$\text{Log}_{10}(x+1)$	BB	0.357
	Relative Abundance	$\text{Log}_{10}(x+1)$	BB	0.088
NW&TS	Species Diversity	Presence/absence	SN, AQV	0.308
	Absolute Abundance	$\text{Log}_{10}(x+1)$	SN, BB, AQV	0.313
	Relative Abundance	$\text{Log}_{10}(x+1)$	BB	0.145
	Water Chemistry Variables *			
NW	Species Diversity	Presence/absence	Conductivity, turbidity	0.263
	Absolute Abundance	$\text{Log}_{10}(x+1)$	Conductivity	0.240
	Relative Abundance	$\text{Log}_{10}(x+1)$	Conductivity	0.220
TS	Species Diversity	Presence/absence	Turbidity	0.098
	Absolute Abundance	$\text{Log}_{10}(x+1)$	pH	0.129
	Relative Abundance	$\text{Log}_{10}(x+1)$	pH, conductivity	0.292
NW&TS	Species Diversity	Presence/absence	Conductivity, turbidity, temp	0.226
	Absolute Abundance	$\text{Log}_{10}(x+1)$	pH, temp	0.226
	Relative Abundance	$\text{Log}_{10}(x+1)$	pH, temp	0.304

SN = snags, BB = bare bank, AQV = aquatic vegetation, temp = water temperature *Results for December 2006 removed

As with the macroinvertebrate assemblages (see Table 5.17), environmental data failed to explain the variability in the observed zooplankton assemblage structure (Table 5.28). There was a stronger association between assemblage data and the available habitat than between the assemblage data and water quality results, but all r_s values were low (< 0.4).

5.4 Discussion

5.4.1 Aquatic assemblages in waterbodies of the Border Rivers Catchment

The macroinvertebrate taxa across both the spatial and temporal studies included 109 taxa, 102 of which were found at natural wetlands and 72 at storages. The diversity of storages was comparable to that described for dryland rivers elsewhere (Boulton and Lloyd, 1991; 95 taxa; Sheldon and Walker, 1998; 80 taxa; Sheldon *et al.*, 2002; 70 taxa; Marshall *et al.*, 2006; 70 taxa) but lower than that found in Australia's wet/dry tropics (Outridge, 1988; 133 taxa; Leigh and Sheldon, 2009, 124 taxa). In comparison the diversity of natural wetlands in the Border Rivers Catchment was slightly higher than recorded for dryland areas but lower than the wet/dry tropics. However, comparing diversity between studies can be risky due to possible differences in sampling methods and sampling times.

Over all sampling occasions a total of 28 zooplankton taxa were collected, 27 from natural wetlands and 22 from typical storages. The diversity of both typical storages and natural wetlands was lower than that found in other semi-arid floodplain systems (Jenkins and Boulton, 2003, 63 taxa; James *et al.*, 2008, 57 taxa). Again, this variation may be due, in part, to differences in sampling methods and sampling times. Zooplankton densities were highly variable among both sites and sampling times but were within the range reported for similar systems (Jenkins and Boulton, 2003; James *et al.*, 2008).

5.4.2 Spatial variation in aquatic assemblages

Macroinvertebrate assemblages of all three waterbody types were uneven with a few taxa dominating the abundances compared to all other taxa. This unevenness was similar to results found by Sheldon and Walker (1998) in the Murray-Darling River system.

Results from both the spatial and the temporal study displayed considerable variation between waterbody types. Although only evident from the spatial study, as expected natural wetlands and 'no tailwater' storages had increased macroinvertebrate abundances compared with typical storages. As predicted, natural wetlands had more diverse macroinvertebrate assemblages and increased species richness compared with storages. Contrary to my hypothesis, 'no tailwater' and typical storages had similar diversity. Although the presence of tailwater did not appear to be having an adverse affect on aquatic assemblages within storages, very few 'no tailwater' sites were included in the study due to low water levels throughout the catchment.

The zooplankton assemblages of both natural wetlands and typical storages were also uneven with a few taxa dominating the abundances. However, the dominant taxa varied between the two waterbody types. Higher diversity and abundance of zooplankton were also recorded in natural wetlands in comparison to typical storages. Many forms of zooplankton have the ability to persist in a resting, dormant, stage in dry sediment and emerge upon inundation (Boulton and Lloyd, 1992; Jenkins *et al.*, 2002; Jenkins and Boulton, 2003). Not only will natural wetlands have zooplankton washed into them from other areas during a flood but more will emerge from this 'egg bank' of dormant eggs (Jenkins *et al.*, 2002; Jenkins and Boulton, 2003). Compared with the much older natural wetlands, storages, due to their young age, may not have built up this egg bank in their sediment and so, have reduced diversity and abundance. Cole and Fenton (1995) found that continuous flooding reduced the diversity of zooplankton emerging from sediments while other studies have found that abundance and diversity is greater for sediments taken from more frequently flooded areas than from those flooded more rarely (Boulton and Lloyd, 1992; Jenkins and Briggs, 1997). How the storages are managed in terms of water levels is likely to affect zooplankton assemblages within them (Crome and Carpenter, 1988). If the storages sampled had been kept continuously wet this could account for the reduced diversity and abundance recorded.

These spatial differences in aquatic assemblages may also be due to greater morphological and hydrological variability within the natural wetlands. It is known that channel complexity, habitat availability, temperature and depth all affect the macroinvertebrate assemblages of rivers and wetlands (Pringle *et al.*, 1988; O'Connor, 1990; Thoms *et al.*, 2006). However, in this study, neither measured habitat, nor morphology variables explained the spatial variation in assemblages suggesting that parameters not measured could be influencing spatial variation in assemblages. A more in depth assessment of habitat availability or the measurement of additional morphology variables, such as cross-sectional shape or bank slope, may be warranted to identify contributing factors.

It is also possible that connection history could be driving spatial variation in assemblages (Marshall *et al.*, 2006). Invertebrate assemblages of waterholes have been found to be most similar at either end of the hydrological connectivity continuum (Ward *et al.*, 1999; Leigh, 2008). That is, waterholes which are highly connected (high water levels) have invertebrate communities with similar compositions (Sheldon and Thoms, 2006) and after extended disconnection (low water levels) waterholes once again become similar and species poor (Sheldon *et al.*, 2003), with maximum diversity occurring at some intermediate level of connectivity (Ward *et al.*, 1999). It is possible that the spatial patterns observed in this study have been influenced by recent temporal trends.

Although there was also no significant relationship between water quality and assemblage structure, some trends observed in water quality may be leading to differences in aquatic assemblages between storages and natural wetlands. In this study conductivity and salinity levels were within the range previously recorded in lagoons in the Border Rivers Catchment (Medeiros, 2004) but higher than those recorded at river sites (Gordon, 2001; DIPNR, 2003). Although most conductivity levels were below ANZECC (1992) guidelines for medium salinity waters (0.8 mS cm^{-1}) occasional values did exceed this limit. The highest conductivity and salinity levels were recorded in typical storages (0.353 to 1.453 mS cm^{-1} and 0.01 to 0.06, respectively). At the same time natural wetlands with similar water depths recorded conductivity readings of 0.335 to 0.493 mS cm^{-1} and had salinity values of 0.00% to 0.02%. It is likely that the practice of recycling tailwater has lead to a concentration of salts in storages which could be having an adverse effect on their assemblages (Hart *et al.*, 1991; Nielsen *et al.*, 2003; Velasco *et al.*, 2006; Kefford *et al.*, 2007).

Many chemical equilibria in water are influenced by pH levels; for example, an increase in pH can increase the toxicity of other pollutants (DIPNR, 2003). Most natural freshwaters have a pH in the range of 6.5 to 9.0 (DIPNR, 2003). In this study, mean values of pH across all study sites were slightly alkaline and some sites fell above 9.0, with the highest readings recorded in storages without tailwater (9.15) and typical storages (9.88). These higher pH levels in storages could compound the effects of pollutants (Alabaster and Lloyd, 1980; Lechleitner *et al.*, 1985; Nodvin *et al.*, 1986).

Temperature also plays an important role in determining the distribution of aquatic organisms due to its influence on survival, growth, reproduction and behaviour (Gordon, 2001). The conceptual model hypothesised that, due to the lack of riparian vegetation and the shading that it provides, storages would have increased water temperatures compared with natural wetlands. Contrary to this, the maximum temperature throughout the study was recorded in a natural wetland. This is most likely due to the increased depth and area of storages compared with natural wetlands (see Chapter 3), causing the water in natural wetlands to heat up more quickly despite increased shading.

5.4.3 Temporal variation of aquatic assemblages

Although there was spatial variation in assemblages, temporal changes in macroinvertebrate and zooplankton assemblage composition followed a similar pattern in both natural wetlands and typical storages as they dried. Fluctuations in water levels are known to influence macroinvertebrate assemblages in reservoirs (Kaster and Jacobi, 1978), billabongs (Nielsen *et al.*, 1999) and floodplain habitats (Sheldon *et al.*, 2002). Abundances were higher during the second irrigation season when water levels were lowest but, as Nielson *et al.* (1999) suggested, this may only be a perceived increase in abundance as water levels decrease and there is a concentration of animals into a smaller area.

Despite the work of certain researchers (e.g. Crome and Carpenter, 1988), there is a limited understanding of zooplankton communities in semi-arid systems and how they change in response to flooding and drying and the associated changes in water quality (Shiel *et al.*, 2006). However, the life cycles of zooplankton are relatively short. Rotifers may have only a 4-5 day life cycle at 25-30 ^{o}C (Shiel, 1995). The monthly sampling carried out in this study may have missed some species and any population responses to environmental cues such as fluctuating water levels.

These findings are in contrast to the prediction that due to irrigation activities aquatic assemblages in storages would not have the same temporal patterns as those found in natural wetlands. Over the course of our sampling the region was in a severe drought and there were limited opportunities for storages to be refilled. In this respect they may have been acting more like natural wetlands than is usual. It may take a 'wet' year to truly assess how temporal variations of aquatic communities compare between the two waterbody types. At the height of summer, cotton crops may be irrigated as often as every 10 days. It may therefore be necessary to sample more frequently in order to identify any effects of fluctuations in water levels on aquatic communities.

Storages and natural wetlands were least similar to each other at times when water levels were high and most similar when they were low. Similarity within waterbody type was also lowest when water levels were high. This may be due to increased water temperatures and decreased water quality in both waterbody types when water levels are low, which override any benefits that increased habitat diversity may be having on macroinvertebrate and zooplankton assemblages in natural wetlands.

Conductivity and salinity values were highest on the last two sampling occasions when water levels were at their lowest throughout the catchment. This is to be expected as evaporation leads to a concentration of the dissolved salts (Hart and McGregor, 1980; Mackey, 1991). Decreasing water levels can lead to an increase in pH (Osbourne *et al.*, 1987). This was generally the case in typical storages with the highest pH levels noted in January 2007 when water levels were very low. However, they did decrease again in February 2007 even as water levels continued to drop. Though a similar pattern was observed in some natural wetlands, it was not true in all cases.

Turbidity is commonly used as an indicator of the amount of suspended solids in the water column, such as clay, silt, organic particles and microscopic organisms (Preece, 1998). These suspended solids can interfere with the feeding, breathing and reproduction of certain organisms (Arthington, 1995; Wylie, 1995; DIPNR, 2003). There was considerable spatial and temporal variation in turbidity readings, ranging from 16 NTU to 1590 NTU. While these results are similar to other studies in this area (Boey *et al.*, 1997; Medeiros, 2004), the majority are well over the trigger value of 50 NTU for the protection of aquatic ecosystems (Bek and Robinson, 1991; ANZECC, 2000); only one site had a mean turbidity level of under 50 NTU. Although a natural feature of Australian freshwater systems (Preece, 1998; Medeiros, 2004), McCosker (1996) recognised turbidity as one of the most significant contaminants threatening the Border Rivers Catchment. It has also been suggested that high turbidity levels at downstream sites compared with upstream sites in the region are responsible for the lower diversity and abundance of macroinvertebrates and macrophytes (Boddy and Bales, 1996). As reported in other studies (Hart and McGregor, 1980; Medeiros, 2004), highest turbidity values were recorded when water levels were at their lowest.

It would be expected that the unfavourable conditions created by decreasing water levels would reduce the assemblages to a reduced number of taxa that can tolerate these conditions (Sheldon *et al.*, 2003). This was the case in the natural wetlands with a decreasing trend in the number of taxa over time. In comparison, in the typical storages the lowest numbers of taxa were recorded in February and March 2006 when water levels were still high. However, this could also be an effect of dilution.

5.4.4 Variation in functional mobility groups between natural wetlands and storages

There was significant spatial variation in the functional composition of macroinvertebrate assemblages. As expected, during the spatial study (T1), typical storages were dominated by taxa from the zero and low FMG's suggesting that any highly mobile taxa that had been pumped into these sites may have moved on to nearby natural wetlands. However, although the natural wetlands sampled had increased abundances of highly mobile taxa compared with typical storages they were also dominated by low and zero mobility taxa. This may be a reflection of the time of year that the sites were sampled. Macroinvertebrates are more active during the warmer months, with recruitment of certain taxa coinciding with rising temperatures and water levels (Nielsen *et al.*, 1999). Sampling during the summer months may have identified different spatial patterns in the structure and function of macroinvertebrate assemblages. In contrast to our hypothesis, the fauna at the 'no tailwater' storages was dominated by highly mobile taxa. There are a number of possibilities as to why this was the case; the habitat features and conditions at these storage sites were favourable, these sites were isolated so there were no natural wetlands for mobile taxa to relocate to or these sites were in such close proximity to natural wetlands that mobile taxa were frequenting both waterbodies. Before any conclusions can be drawn, further sampling to see if these patterns were repeated would be necessary.

Despite significant spatial differences and increased variation between typical storages at particular sampling times, macroinvertebrate functional composition followed a similar trend over time in both natural wetlands and typical storages. Over the course of the temporal study (T2-T7), the macroinvertebrates collected moved from being dominated by low and zero mobility taxa to being dominated by high mobility taxa. These changes coincided with decreasing water levels and reduced water quality (see Section 5.3.1). This is in contrast to my hypothesis, which predicted that unfavourable conditions would lead to assemblages dominated by low mobility taxa. However, it may be the case that the highly mobile taxa collected, were also taxa that were tolerant of the poor water quality conditions. For example the highly mobile Notonectidae and Corixidae are known to be extremely tolerant of poor water quality due to their air-breathing habit (Gooderham and Tsyrlin, 2002; Chessman, 2003).

5.4.5 Waterbody health

During the spatial study (T1), EPT index values were generally very low across all sites. However, results suggested that natural wetlands and ‘no tailwater’ storages supported higher assemblages of EPT taxa than typical storages. During the temporal study, there was a significant difference in EPT index values between the two waterbody types. EPT taxa were recorded from natural wetlands on all sampling occasions apart from February 2007 when water levels were at their lowest. The associated water quality issues may have resulted in the absence of EPT taxa at this time. EPT taxa were only recorded in typical storages in November 2005 and January 2007. It is unclear why this was the case but increased sampling effort may have increased the occurrence of EPT taxa. During temporal sampling, the highest EPT index value was five for natural wetlands and three for typical storages. These values are very low compared to other studies using EPT index values; for instance Marchant *et al.* (1994) recorded up to 74 EPT taxa in one river site in Victoria over seven sampling occasions. The highest number of EPT taxa recorded in one natural wetland was seven over three sampling occasions. While in the typical storages a maximum of three EPT taxa were recorded in one site over three sampling occasions.

Although there may be differences in taxonomic resolution and number of samples between the studies it is clear that all waterbodies, including the natural wetlands, in the Border Rivers Catchment have low EPT values. However, EPT index values were developed for the bioassessment of streams. It is known that, in comparison to streams, wetlands have reduced numbers of EPT taxa (King and Richardson, 2002; Chessman, 2003) and do not contain other coarse-level taxa that are as sensitive to human activity that could be used in place of the EPT taxa (King and Richardson, 2002). For example, families such as Dytiscidae and Chironomidae, which are common in wetlands, tolerate a wide range of environmental variables (Cranston, 1994; Gooderham and Tsyrlin, 2002). In his study of macroinvertebrate communities in Victoria, Australia, Metzeling *et al.* (2006) advised that the EPT index could not be used in all regions due to natural variations in macroinvertebrate distributions, resulting in very low numbers of EPT families (6-10) at certain reference sites. Therefore, perhaps this is not a valid approach for evaluating the ‘health’ of wetland sites in the Border Rivers Catchment.

SIGNAL scores were also very low throughout the sampling sites ranging from 1.5 to 4.21. Only two samples, both from the same ‘no tailwater’ storage, Barra Storage (TW1), had signal scores greater than 4, suggesting that all other sites, both storages and natural wetlands, were severely polluted (Table 5.29). However, like the EPT index values, SIGNAL scores were developed for streams. Chessman (2003) warns that SIGNAL scores for wetlands are likely to be lower than streams in the same region.

Table 5.29: General key to SIGNAL scores (Metzeling *et al.*, 2006)

SIGNAL score	Water Quality
>7	Excellent
6-7	Clean water
5-6	Mild pollution
4-5	Moderate pollution
<4	Severe Pollution

If we simply use the signal scores to compare between waterbody types in the Border Rivers Catchment, rather than with other studies, it is clear that natural wetlands and storages are generally found in different areas of the graph (see Fig. 5.16 and 5.17). During the temporal study there were significant differences in SIGNAL 2 scores between natural wetlands and typical storages. If we assume that most natural wetlands in the catchment have favourable conditions with good habitat and low concentrations of chemicals (Quadrant 1) then the majority of storages appear to have harsh conditions and toxic levels of pollutants (Quadrants 3 and 4). It is likely that these pollutants are a result of the widespread agricultural industry in the catchment and the storages close proximity to agricultural land and associated activities. Results also indicate that some natural wetlands may be similarly polluted. Wylie (1995) highlighted the high concentration of nutrients and pesticides as two of the four main water quality issues related to agricultural practices in the region. Despite the cotton industries efforts to reduce the amount of chemicals used during a season (APVMA, 1999; Williams, 1999), it would be fair to say that, due to their position, storages are more likely to have high levels of chemicals and therefore lower signal scores than natural wetlands and that those natural wetlands which are closer to irrigation areas are at higher risk of chemical contamination.

5.4.6 Conclusions

Differences in aquatic assemblages were apparent between storages and natural wetlands. Although not obvious from the fish data, natural wetlands were generally more diverse and productive than storages in the Border Rivers Catchment. Storages also supported lower numbers of the sensitive EPT taxa and had less favourable SIGNAL scores than natural wetlands. As on-farm storages are a permanent part of the floodplain in the Border Rivers Catchment and other irrigation areas of Australia, it is likely that the long term preservation of the biodiversity and health of aquatic areas will be a balanced approach between the conservation of natural wetlands and the improvement of artificial waterbodies as aquatic habitat.

In order to maintain the aquatic diversity provided by natural wetlands it will be essential to maintain or improve upon the current flooding regime in the region. Although they are creating a form of aquatic habitat, increased water diversions into storage dams will only serve to reduce the aquatic biodiversity and aquatic health of the catchment. A number of suggestions have been made as to how storages could be altered in order to increase their biodiversity (see Section 3.4.3), such as providing additional habitat in the form of aquatic vegetation and woody debris, incorporating gently sloping sides, central islands and shallow areas into the structure of storages and minimising the number of storages that contain recycled tailwater (Lutton, 2005; Kennedy and Jarman, 2006). Although, the results from this study can not confirm whether any of these improvement activities would be beneficial, I would suggest further research before any are discounted. In particular, I recommend sampling an increased number of sites especially ‘no tailwater’ storages, carrying out a more detailed habitat assessment of the study sites and measuring additional morphology variables. However, if any of the proposed improvement activities are implemented it will be essential to monitor the aquatic assemblages both before and after any changes are made.

Chapter 6 Sources of colonisation for storages

6.1 Introduction

Floodplains can be viewed as riparian systems or aquatic-terrestrial ecotones and form an integral part of the natural riverine system (Gregory *et al.*, 1991). Flood pulses drive the transitions between aquatic and terrestrial phases of this ecotone (Tronstad *et al.*, 2007) and create numerous temporary wetlands. These can be regular, seasonal periods of inundation (Adis and Junk, 2002; De Simone *et al.*, 2003; Feyrer *et al.*, 2006) or highly irregular in their timing and predictability (Puckridge *et al.*, 1998; Kingsford *et al.*, 1999; Bunn *et al.*, 2006). Such events are often responsible for a boom of production on inundated floodplains and associated wetlands, especially those in dryland regions (Walker *et al.*, 1997; Puckridge *et al.*, 2000; Bunn *et al.*, 2006; Balcombe and Arthington, 2009). The flow requirements of large floodplain rivers include those which sustain ecosystem processes and biota in what are traditionally considered to be terrestrial systems (Capon, 2003). The notion that a river can be defined as the area between the banks has long been disregarded (Junk *et al.*, 1989; Gregory *et al.*, 1991; Capon, 2003). However, many floodplains in dense urban or intensive agricultural areas have been isolated from their river channels and are functionally extinct (Tockner *et al.*, 2008).

In order to exist in this ever changing floodplain environment, aquatic taxa must be able to colonise newly inundated habitats rapidly and often on an irregular basis (Tronstad *et al.*, 2007). Aquatic invertebrates are able to colonise inundated floodplains by three methods of dispersal: immigration by drifting in the water column or crawling on substrate (Smock, 1994; Jenkins and Boulton, 2003), persistence during the dry phase in a dormant or semiaquatic state (Boulton and Lloyd, 1992; Jenkins *et al.*, 2002; Jenkins and Boulton, 2003; Tronstad *et al.*, 2005) or through aerial colonisation (Tronstad *et al.*, 2007). Flood events allow aquatic taxa to move from the river channel and permanent wetlands onto the floodplain, enabling colonisation of temporary wetlands (Nielsen *et al.*, 1999; Jenkins *et al.*, 2002). Fish are known to move onto the floodplain to take advantage of more diverse food resources and increased feeding opportunities associated with flood events (Ross and Baker, 1983; Balcombe *et al.*, 2005). Inundation of the floodplain also stimulates intense bouts of production in macroinvertebrates (Gladden and Smock, 1990; Sheldon *et al.*, 2002) and waterbirds (Kingsford *et al.*, 1999b).

Floodplain and dry wetland sediments also contain a reservoir (‘egg bank’) of dormant eggs of aquatic invertebrates, such as rotifers and microcrustaceans (Boulton and Lloyd, 1992; Nielsen *et al.*, 2003) allowing recolonisation upon flooding (Hairston Jr., 1996). The number of eggs in the egg bank is often large (Hairston Jr., 1996; Nielsen *et al.*, 2000) and long lived (Boulton and Lloyd, 1992; Hairston Jr. *et al.*, 1995; Jenkins and Briggs, 1997). The viable eggs of calanoid copepods, cladocerans (and other branchiopods) and rotifers are consistently found in numbers ranging between 10^3 and 10^6 m^{-2} and can be up to 332 years old (Hairston *et al.*, 1995). While these groups have desiccation resistant life stages that enable them to persist during these dry terrestrial phases, other taxa employ different strategies, including crayfish which survive drought conditions by burrowing deep enough to reach groundwater (Gooderham and Tsyrlin, 2002; Pintor and Soluk, 2006) and turtles that bury themselves in the mud or burrows of other animals (Kennett and Christian, 1994; Peterson and Stone, 2000).

Aerial dispersal is the final method of colonisation of temporary waterbodies. Oviposition and non-reproductive immigration by flying or springing insects are two mechanisms employed by macroinvertebrates to colonise recently inundated wetlands (Tronstad *et al.*, 2007). These include the winged insects especially those with short development times, such as Chironomidae that can complete their life cycle in < 15 days and emerge before the floodplain dries (Hauer and Benke, 1991). Non-reproductive immigration includes active dispersal by aquatic adult insects, particularly those with good dispersal capabilities, such as Dytiscidae (Schafer *et al.*, 2006). Waterbirds are another example of aquatic fauna which use aerial dispersal as a means of utilising temporary wetlands. Arid zone wetlands are generally only flooded for a limited period of time and waterbirds must move frequently to seek feeding and breeding habitat (Kingsford *et al.*, 1999a; Roshier *et al.*, 2002). Invertebrate taxa can even be transported into wetlands via waterbirds, including mussels attached to the feet of ducks (McMichael, 1958).

In contrast to natural floodplain wetlands, storages built on the floodplain are often separated from natural flooding events by embankments and levee banks. Flood water can only enter storages when it is pumped via irrigation channels. Larger aquatic species can be injured in the process and extraction into irrigation off-takes has led to a serious decline in fish numbers in irrigation districts around the world (Blackley, 2003). These controlled, one-way exchanges of river water and overland flow are likely to reduce the abundance and biodiversity of aquatic biota entering the storages and in turn limit the continued development of an egg bank in the storage sediment. To compound this further, the bottoms of storages can be utilised for cropping and grazing during dry periods. This use is likely to physically disrupt the egg bank and remove nutrients from the system rendering them unavailable for growth and development of emerging invertebrates (Boulton and Lloyd, 1992).

Floodplains and natural wetlands and their biota have adapted to fluctuating water levels between wet and dry periods and animals and plants have developed physiological, morphological or behavioural traits that allow survival in both the dry and flooded phases (Wiggins *et al.*, 1980; Walker *et al.*, 1995). Inhabitants of floodplains must be able to produce diapausing eggs or other dormant stages, or disperse to other areas during a drying event. When flood waters return they then must be able to recover from dormancy or recolonise the temporary wetland (Tronstad *et al.*, 2007). Manipulation of the natural flooding regime, as would be found in storages, is likely to affect these life cycle traits and change the diversity and abundance of the egg bank in storage sediment (Nielsen *et al.*, 2000).

6.1.1 Aims

In order to gain more of an understanding of how storages are colonised, this chapter investigates two of these potential pathways for recolonisation: (i) the aquatic assemblages present in water pumped into storages and (ii) the egg bank found in the sediments of dry storages. It is believed that the passage of extracted water through a pump station and unnatural flooding regimes in storages along with physical disturbance of storage soil will reduce the abundance and diversity of fauna in storages. The specific aims are to:

1) Determine the size classes of fish in pumped water to test the hypothesis that only small individuals will pass through the pump station unharmed.
2) Determine the diversity and abundance of fish, benthic macroinvertebrate and zooplankton fauna present in extracted river water to test the hypothesis that the assemblage composition will be similar to that of typical storages.
3) Compare the diversity and abundance of zooplankton emerging from the sediment of storages with that of the adjacent floodplain to test the hypothesis that the eggbank of storages will be less diverse and abundant than that of floodplain sediment.

6.2 Methods

6.2.1 Pumping collections

6.2.1.1 Sampling design

Two sites were selected to investigate the nature of the fauna in water that is pumped from a river into a water storage during water harvesting. Jericho pump station (Jericho PS) and Newinga pump station (Newinga PS) are both situated on the Weir River south of Talwood, QLD. Like many Australian dryland rivers (Sheldon et al., 2002; Bunn et al., 2003), the Weir River flows only after significant rainfall in the catchment area and during dry periods becomes a series of disconnected waterholes. There are a number of private weirs along this river that fill up during the course of a flood: examples include those located at Newinga and Jericho. Depending on the size of the flow and type of licence held, property holders may be given permission to extract water at the peak of the hydrograph and, under certain conditions, can pump out water stored in their weir once the river has ceased to flow.

Table 6.1: Fish sampling dates of pumped water at Newinga and Jericho pump stations.

Date	Site	Time	Rep	River Condition	Sampling Time
02 Feb 2006	Newinga	1	1	Flowing	2pm-3pm
03 Feb 2006	Jericho	1	1	Flowing	10am-11am
07 Feb 2006	Jericho	2	1	Flowing	12pm-5pm
07-08 Feb 2006	Jericho	2	2	Flowing	5.45pm-8am
08 Feb 2006	Jericho	2	3	Flowing	10am-4pm
19 March 2006	Jericho	3	1	Ceased to flow	10am-4pm
20 March 2006	Jericho	3	2	Ceased to flow	8am-4.45pm
22 March 2006	Jericho	3	3	Ceased to flow	10am-4pm

Jericho PS was sampled 3 times from February 2006 to March 2006, while Newinga PS was sampled only once in February 2006 (Table 6.1). The first and second sampling occasions in early February 2006 occurred after the peak of the hydrograph but while the river was still flowing. The last set of samples was collected after a flow at the end of February once the river had stopped flowing and the private weirs were being pumped down (Fig. 6.1). Newinga PS pumped into a wide (> 6 m), deep (2.5 m) supply channel and on the one sampling occasion, water had been stored in the channel before pumping commenced. In contrast the supply channel at Jericho was narrower (5 m) and shallow (< 1 m) and water continuously flowed away from the pump site. Water quality parameters of pumped water at both sites were within the range recorded for natural wetlands and storages in the Border Rivers Catchment (see Chapter 5). Extracted water at both sites was slightly alkaline with moderate turbidity levels, low salinity and conductivity readings and temperatures as expected for the time of year.

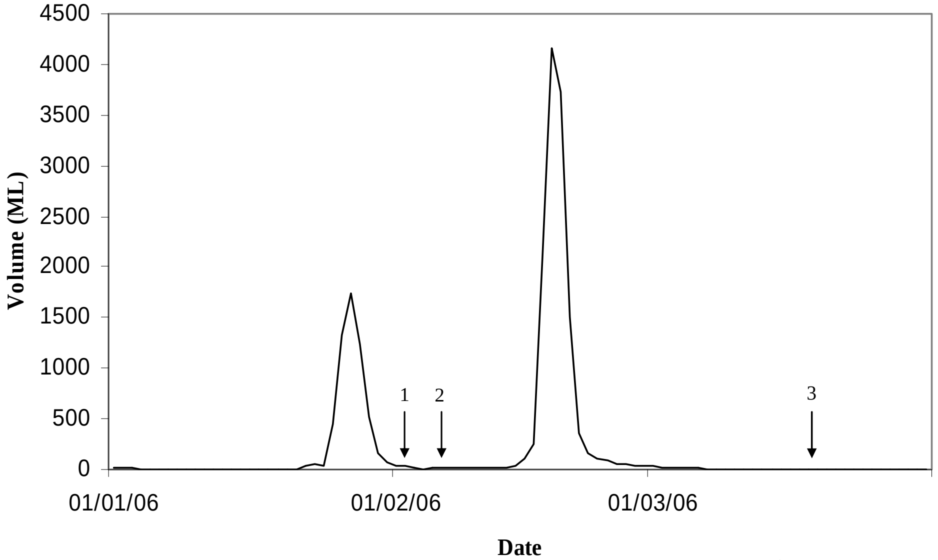


Fig. 6.1: Flow volumes recorded at the Weir River at Talwood (416202A) from 1st January 2006 to the 31st March 2006. Arrows indicate the three sampling times.

6.2.1.2 Fish collection

Fish were sampled by extending a large fyke net across the full width of the channel in front of the pump station outlet pipe (Fig. 6.2 and 6.3). As there were no standards to follow for this type of sampling, trial and error had to be used when deciding on sampling effort. Due to the depth and width of the supply channel and the force of water at Newinga PS, the fyke net had to be held in place by three people. Thus, the fyke net was only left set for one hour and on one occasion at Newinga PS. Both the wing width and set duration were recorded for subsequent calculation of catch per unit effort (CPUE). After net retrieval, all fish were identified to species, counted and, with the exception of exotic species, returned to the river after a subsample of each species was measured and weighed. Fish counts were standardised to provide data on the CPUE and allow comparison between sites and times.

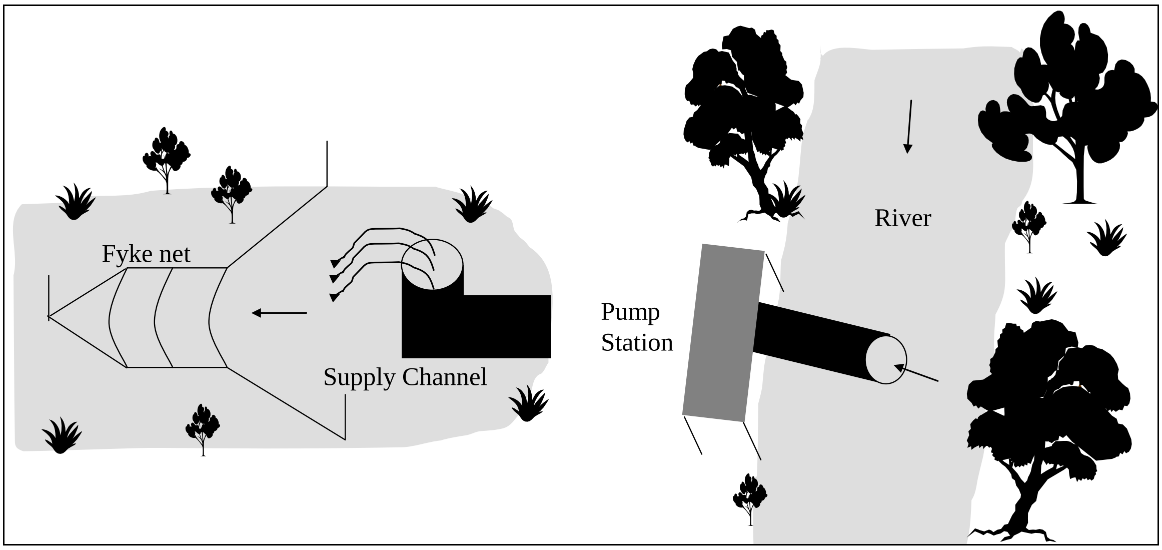


Fig. 6.2: Position of fyke net in a supply channel at a river side pump station. Arrows show the direction of water flow.

Fig. 6.3: a) Fyke net set up for sampling pumped water at Jericho PS, b) pump station showing the first water coming through the outlet pipe.

6.2.1.3 Macroinvertebrate collection

Macroinvertebrates were collected by sweeping a 250 µm mesh dip net in the area in front of the outlet pipe for 30 s. Three replicate samples were taken on each sampling occasion (Table 6.2). Samples were preserved in 70% ethanol in the field and later washed through nested sieves with mesh sizes of 4000, 2000, 1000 and 250 µm. The organisms were sorted, counted and identified as far as practical using various keys listed in Hawking (2000). Unidentified specimens were recorded as separate taxa (e.g. tiny unidentified Chironomid).

Table 6.2: Macroinvertebrate, zooplankton and water quality sampling dates of pumped water at Newinga PS and Jericho PS.

Date	Site	Time	River Condition
02 Feb 2006	Newinga	1	Flowing
03 Feb 2006	Jericho	1	Flowing
07 Feb 2006	Jericho	2	Ceased to flow
19 March 2006	Jericho	3	Ceased to flow

6.2.1.4 Zooplankton collection

Zooplankton samples were obtained by collecting 10 litres of water from in front of the outlet pipe and pouring it through a 53 µm net. Three replicate samples were taken on each sampling occasion (Table 6.2) and preserved in 70% ethanol in the field. In the laboratory, samples were coloured with Rose Bengal Dye and made up to a known volume. A Sedgewick Rafter counting chamber was used to count and identify the organisms to genus using the key of Shiel (1995).

6.2.2 Egg bank study

6.2.2.1 Sediment Collection

Sediments were collected in May 2006 from five typical storages and five adjacent floodplain sites (Table 6.3). Storages were located along the Macintyre River from east of Boggabilla to Bungunya (Fig. 6.4) and floodplain sites were selected in consultation with landholders in close proximity to these storages. At each storage and floodplain location, three sites (20 m x 20 m) were randomly selected. At each of these sites 3 sediment cores of 4.5 cm depth and 10 cm diameter were collected using a metal ring and aggregated in a single plastic bag to form a sample. Individual cores were collected from random locations within the 20 m^2 quadrat. In the laboratory any live or dead vegetation was removed and the sediment was lightly crushed to break up clods, mixed, air dried and stored until the egg bank trials commenced.

Table 6.3: Sites sampled in May 2006 for sediment collection.

Floodplain Site	Site Code	Typical Storage	Site Code
Floodplain Wirindi	F1	Wirindi Storage	TS10
Floodplain Taraba	F2	Taraba Storage 2	TS11
Floodplain Mullala	F3	Mullala Back Dam	TS12
Floodplain Eumorella	F4	Eumorella Storage 2	TS13
Floodplain Mundine	F5	Mundine Storage 3	TS14

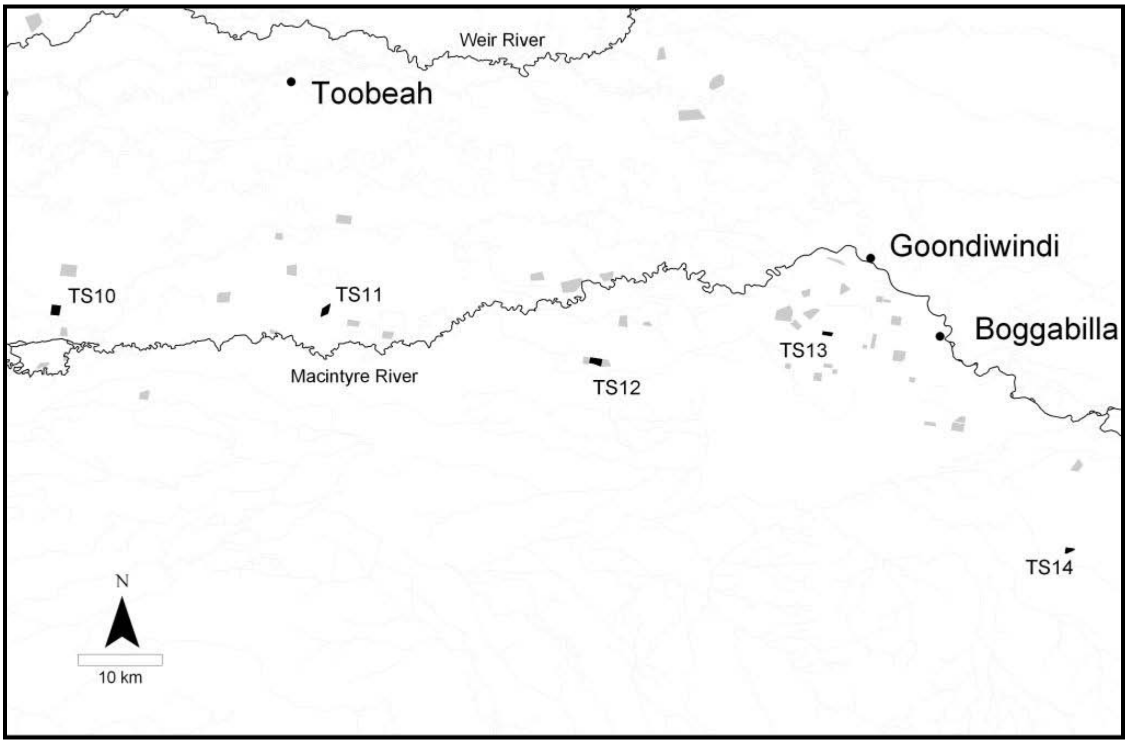


Fig. 6.4: Typical storages sampled for the eggbank study in May 2006 are shown in black. Grey objects represent other storages in the area.

6.2.2.2 Laboratory

Before the experiment began, half a cup of soil from each aggregate sample was removed for analysis of organic matter content using the Ash-Free Dry Weight (AFDW) technique, combusting the sample in a muffle furnace at $500^{o}C$ for 6 hours. Weight before and after drying was used to calculate the percentage organic content of the soil.

For the egg bank study, replicate microcosms were made from 650 ml plastic "take-away food containers" prepared by piercing 5 holes in the bottom of each container and attaching handles of fishing line so they could be lifted in and out of the water. The remaining soil from each sample was placed into these pots such that the soil was level with the top of the container to allow maximum drainage. The pot was in turn placed into a 4L plastic container serving as a microcosm. All 30 containers (5 wetlands x 3 replicates + 5 typical storages x 3 replicates) plus three controls were arranged randomly on pallets in a makeshift greenhouse in Talwood, QLD. Water from the Macintyre River was filtered through a 35 µm net prior to flooding the sediment. A small amount of this filtered river water was retained as a blank for the aquatic fauna or propagules it contained. Conductivity (mS cm^{-1}), pH, dissolved oxygen (DO) (mg L^{-1}), turbidity (NTU), water temperature (^{o}C) and salinity (%) of the filtered river water were measured using a Horiba multiprobe U-10. Flooding was initiated by sprinkling the river water from a watering can to minimise disturbance of the soil. Approximately 3 litres of the filtered river water was added to a depth of 3 inches over the soil. Filtered river water was added, when necessary, over the course of the study to maintain the water at this level. The three controls were filled only with filtered river water to monitor possible aerial contaminants. Minimum and maximum air temperatures were recorded daily over the first four weeks of the experiment and final readings were taken on day 72, from two gauges placed at either end of the green house.

After 2, 7, 10, 14, 17, 21, 24, 28 and 73 days the pots were gently raised out of the water to minimise soil disturbance. A Horiba multiprobe U-10 was used to measure conductivity (mS cm^{-1}), pH, dissolved oxygen (DO) (mg L^{-1}), water temperature (^{o}C) and salinity (%) prior to pouring the water through a fine mesh (35 µm) (Table 6.4).

The filtered water was returned to the 4L container and the pot was carefully lowered back into the water. The net was washed into a sample bottle using 70% ethanol. All equipment was washed in filtered river water between samples. In the laboratory, samples were coloured with Rose Bengal Dye and made up to a known volume. A Sedgewick Rafter counting chamber was used to count and identify the organisms to genus using the key of Shiel (1995). Samples from Day 10 and Day 14 were amalgamated and counted as one sample. The same was done for Day 17 and Day 21 and for Day 24 and Day 28 (Table 6.5). After Day 73 the filtered water was returned to the microcosm and left to dry down naturally. The dry soil was analysed for organic matter content using the AFDW technique as before.

Table 6.4: Sampling times for water quality data during the egg bank study from 24th March 2007 to 5th June 2007.

Date	Day	Time
24 March 2007	0	0
26 March 2007	2	1
31 March 2007	7	2
03 April 2007	10	3
07 April 2007	14	4
10 April 2007	17	5
14 April 2007	21	6
17April 2007	24	7
21 April 2007	28	8
5 June 2007	73	9

Table 6.5: Sampling times for zooplankton assemblages during the egg bank study from 24th March 2007 to 5th June 2007.

Date	Day	Time
26 March 2007	2	1
31 March 2007	7	2
03 & 07 April 2007	10 & 14	3
10 & 14 April 2007	17 & 21	4
17 & 21 April 2007	24 & 28	5
5 June 2007	73	6

6.3 Data analysis

6.3.1 Pumping collections

6.3.1.1 Fish assemblages

Standardised catch-per-unit-effort (CPUE) data (sum of the three fyke net catches with a wing width of 6.5 m and sampling duration of 20 h) were used to obtain measures of total fish abundance, the abundance of individual species and the total number of species per wetland site. As there were a low number of replicates these results merely serve to give an insight into the species and size of fish that are surviving the pumping process. Descriptive methods (e.g. pie charts and histograms) were used to explore the fish assemblages of pumped water.

6.3.1.2 Macroinvertebrate assemblages

Summary variables for macroinvertebrate assemblage composition (number of taxa (S), species richness (d), abundance (N), Shannon diversity (H'), Simpson diversity (1-Lambda') and Pielou's evenness (J')) were calculated for each sample. All variables were normally distributed except abundance (N) which required log transformation. Analysis of variance (ANOVA) was used to explore temporal differences (between times at Jericho PS) in each summary variable, while an independent samples 2 tailed *t*-test was used to detect any significant spatial differences (between sampling sites at sampling time 1).

6.3.1.3 Zooplankton assemblages

The same summary variables as for macroinvertebrate assemblages were calculated for each sample. All variables were normally distributed so no transformations were necessary. Analysis of variance (ANOVA) and an independent samples 2 tailed *t*-test were used to explore temporal (between times at Jericho PS) and spatial (between sampling sites at sampling time 1) differences respectively in each summary variable.

Patterns of variation in zooplankton assemblage structure across sampling locations and times were visualised using ordinations. Non-standardised $\log_{10}(x+1)$ (absolute abundance), standardised $\log_{10}(x+1)$ (relative abundance) and species presence/absence data with all taxa included were used. Bray-Curtis similarity was used as the measure of difference between samples. Ordination plots were produced using non-metric multi-dimensional scaling (MDS) with default PRIMER settings (PRIMER-E version 5.2.9; Clarke and Gorley, 2001) and 100 random starts. Solutions were calculated in two and three dimensions and the two dimensional solution is presented if its stress was less than or equal to 0.2.

6.3.2 Egg bank study

6.3.2.1 Water quality and soil organic content

Water quality data were range standardised and tested for normality. The data was transformed accordingly prior to analysis (Table 6.6). The data for all pot types were analysed to detect any significant spatial differences (between pot types). One-way ANOVA tests were used to detect any significant spatial differences in those parameters which were normal or could be rendered normal via transformation. Those water quality variables that were not normal were investigated using non-parametric Kruskal-Wallis tests to detect any spatial or temporal differences.

To detect any significant differences in percentage organic content between pot types before and after the experiment, independent samples 2-tailed *t*-tests were used.

Table 6.6: Transformations performed on water quality parameters prior to analysis. Ref. = reflected, C.N. = couldn't be rendered normal.

	All Pots	**Control**	**Floodplain**	**TS**
Water temp.	Square root	Normal	Normal	Square root
pH	Ref. square root	C.N.	Normal	Ref. square root
Conductivity	Normal	Normal	Fourth root	Normal
DO	Normal	Normal	Normal	Normal
Salinity	C.N.	C.N.	C.N.	C.N.

6.3.2.2 Zooplankton assemblages

To avoid including taxa that may have entered the mesocosm via aerial colonisation or from contaminated equipment, prior to analyses any taxa that were found in the control pots over the course of the experiment were removed (see Table 6.12).

The same summary variables as used previously were calculated for each sample and tested for normality. The data was transformed accordingly prior to analysis (Table 6.7). The data for all pot types were analysed to detect any significant spatial differences (between pot types). For those variables which were normal or could be rendered normal, an independent samples 2-tailed *t*-test was used to test for differences between the two pot types (floodplain and typical storage). For those summary variables which could not be rendered normal via transformation, a Kruskal-Wallis test was used to explore spatial differences in this variable.

The two data sets were also analysed separately to detect any temporal variations (between sampling times) in the two pot types. For those variables that were normal or were made normal via transformation (Table 6.7), a one-way ANOVA was used, while those variables which could not be rendered normal were explored using Kruskal-Wallis tests.

One-way and 2-way crossed analysis of similarities (ANOSIM) based on Bray-Curtis similarity matrices obtained from non-standardised $\log_{10}(x+1)$ (absolute abundance), standardised $\log_{10}(x+1)$ (relative abundance) and species presence/absence data, were used to identify significant spatial and temporal differences in zooplankton assemblage structure. Default PRIMER settings with 10,000 permutations were used (PRIMER-E version 5.2.9; Clarke and Gorley, 2001).

Table 6.7: Transformations performed on summary variables prior to analysis. Ref. = reflected, C.N. = couldn't be rendered normal. TS

Variable	All pots	Floodplain	TS
S: number of taxa	C.N.	C.N.	C.N.
d: species richness	Fourth root	Normal	Square root
N: abundance	Fourth root	Square root	Fourth root
H': Shannon diversity	Normal	C.N.	Normal
1-Lambda': Simpson diversity	Normal	Ref. fourth root	Normal
J': Pielou's evenness	Ref. log	Normal	C.N.

To visualise the results, non metric MDS plots were created using the data for the floodplain and typical storage pots only. Ordination analysis, based on the same Bray-Curtis similarity matrices, were performed using non-metric multi-dimensional scaling (PRIMER-E version 5.2.9; Clarke and Gorley, 2001) to identify any patterns in absolute abundance, relative abundance and species diversity between pot types. Those samples that did not contain any taxa had to be removed before similarity matrices were created in PRIMER. MDS plots were created using default settings in PRIMER and 100 random starts (PRIMER-E version 5.2.9; Clarke and Gorley, 2001). Solutions were calculated in two and three dimensions and the two dimensional solution is presented if its stress was 0.2 or less.

SIMPER based on $\log_{10}(x+1)$ data standardised for relative abundance were used to determine which taxa were most responsible for differences between pot types and to compare similarity within and between pot types.

6.3.3 Colonisation of storages by zooplankton

To examine the relative role of importation and emergence to the colonisation of typical storages we compared the zooplankton assemblages of extracted water and emerging from typical storage pots with that found in typical storages during the temporal study (see Chapter 5). Dendrograms for all three zooplankton assemblage data sets (extracted water, eggbank and typical storages) were generated from presence/absence data using hierarchical agglomerative cluster analysis (UPGMA) in the Primer 5 software package (PRIMER 5.2.9; Clarke and Gorley, 2001). Bray-Curtis was used as the measure of similarity, with group averaging used to generate the clusters. The dendrograms were then compared using manual two-way cluster analysis; one to compare the zooplankton assemblages of the eggbank with those of typical storages and a second to compare the assemblages of extracted water with those of typical storages.

6.4 Results

6.4.1 Pumping collections

6.4.1.1 Fish assemblages

Although a number of individuals were injured or fatally wounded fish were collected from both sites and in all samples. Fish fauna across all pumping samples comprised of eight different species, of which three were exotic, *Cyprinus carpio*, *Carassius auratus* and *Gambusia holbrooki* (Table 6.8). All eight species were collected from Jericho PS while only four species were present in the sample from Newinga PS. The most common species across all replicate samples was *Nematolosa erebi*, which was caught in all eight nets, closely followed by *Leiopotherapon unicolor* and *Macquaria ambigua* (7 out of 8).

At Jericho PS, mean CPUE ranged from 25 at time 1 (3rd February) to 1420 (±1234 s.e.) at time 2 (7th – 8th February) and 420 (±190 s.e.) at time 3 (19th - 22nd March). The standard error was high, particularly at time 2. The total CPUE at Newinga was 320.

As was found during fish sampling of typical and 'no tailwater' storages (see Fig. 4.4), the contribution of exotic species to the total CPUE of pumped water was extremely low (< 1%) compared to native species. In fact, the sample collected from Newinga PS did not contain any exotic individuals (Table 6.8).

Table 6.8: Fish fauna of pumping samples collected from 2nd February 2006 to 22nd March 2006 at Newinga PS and Jericho PS. Exotic species are marked with an asterisk.

Species	Common Name	No. of replicates with species present (max. 8)	CPUE across pump site		
			Newinga	Jericho	
				Mean	s.e.
Retropinna semoni	Australian smelt	4	21.3	4.9	3.2
Nematolosa erebi	bony bream	8	21.3	481.6	356.5
Macquaria ambigua	Golden perch	7	255.7	255.0	169.1
Leiopotherapon unicolor	spangled perch	7	21.3	45.6	12.6
Tandanus tandanus	eel-tailed catfish	1	0.0	0.2	0.2
*Gambusia holbrooki**	mosquito fish	2	0.0	2.9	2.0
*Cyprinus carpio**	European carp	3	0.0	1.3	0.7
*Carassius auratus**	Goldfish	2	0.0	0.9	0.6

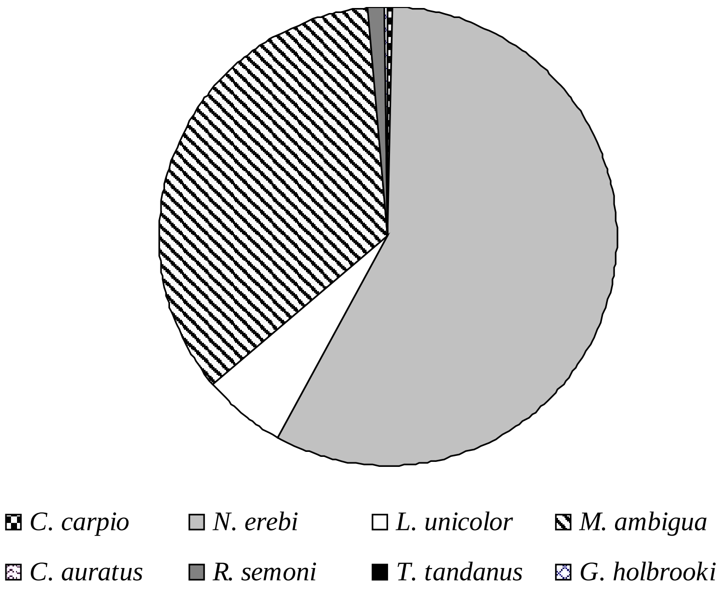


Fig. 6.5: Percentage contribution of each species to the total CPUE across all pumping samples.

The combined catch over all sampling times and sites was dominated by *N. erebi* (Fig. 6.5), contributing 58% of the total CPUE. However, at Newinga PS, *M. ambigua* was the most abundant species contributing 80% of the total CPUE, while *N. erebi* and *M. ambigua* contributed 61% and 32%, respectively at Jericho PS (Fig. 6.5). *M. ambigua* contributed more than a third (35%) of the total CPUE across all pumping samples (Fig. 6.5).

The size of fish surviving the journey through the pump stations was surprisingly large, with the standard length of individuals of *N. erebi* and *M. ambigua* recorded at over 200 mm (Appendix 32). As expected, small fish also made it through the pump intact with the smallest individual only 5 mm. The size distributions of fish collected in the pumping samples were positively skewed with the exception of *R. semoni* which exhibited a normal size distribution (Fig. 6.6).

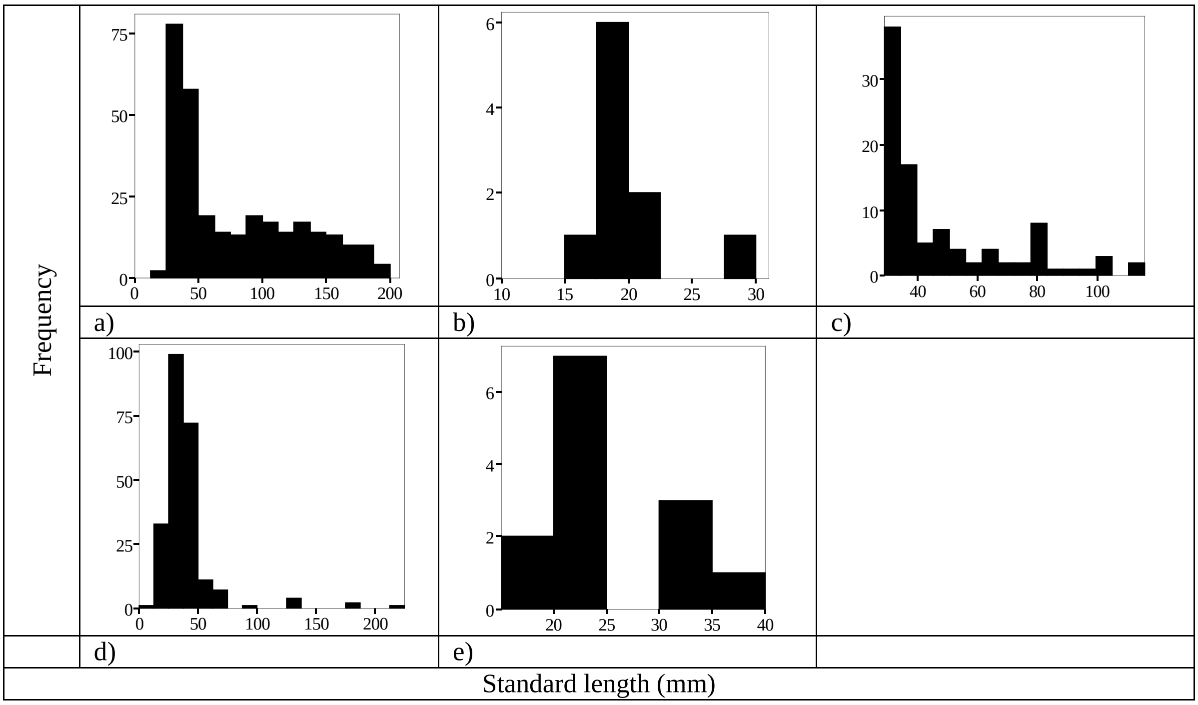


Fig. 6.6: Frequency of occurrence histograms in relation to the standard length (mm) of five of the fish species collected across all pumping samples a) *N. erebi*, b) *G. holbrooki*, c) *L. unicolor*, d) *M. ambigua* and e) *R. semoni*.

6.4.1.2 Macroinvertebrate assemblages

The pooled data from all sampling occasions comprised of 22 macroinvertebrate taxa and 332 individuals (Appendix 33). Seven taxa were common to both sampling sites while eight taxa were specific to Jericho PS and seven taxa were specific to Newinga PS. Fourteen taxa were collected from Newinga PS compared with 15 taxa from Jericho PS. Insects comprised 73% of the taxa but only 38% of the total individuals. The most abundant order was Nematoda (57%) but the majority of these were collected from Newinga PS (Table 6.9). Of the insects, Diptera were the richest (55%) and most abundant order (37%) (Table 6.9). Six taxa occurred only once.

Table 6.9: Taxa and their contribution to the macroinvertebrate assemblage as a percentage of the total individuals and a percentage of the total taxa collected across all pumping samples.

	% Individuals	**% Taxa**
Crustacea	3.61	13.64
Gastropoda	0.30	4.55
Nematoda	56.63	4.55
Bryozoa	1.20	4.55
Insecta	38.25	72.73
Diptera	36.75	54.55
Ephemeroptera	0.30	4.55
Hemiptera	0.60	9.09
Odonata	0.60	4.55

The number of individuals per sample was higher at Newinga PS (33 to 139) compared with Jericho PS (3 to 26). The increased abundance of macroinvertebrates at Newinga PS was mainly due to the high numbers of nematodes. No significant differences in mean number of taxa (S), mean log of abundance (N), mean species richness (Margalef d), mean Pielou's evenness (J'), mean Shannon diversity (H') or mean Simpson diversity (1-Lambda') were detected either between sites at time 1 only or between times at Jericho PS (p in all cases > 0.05).

6.4.1.3 Zooplankton assemblages

Across all sampling occasions 25 taxa of zooplankton were identified, consisting of 18 rotifera, 4 copepods and 3 cladocerans. Rotifera made up 50% of the total individuals closely followed by the copepods (41%), while the cladocerans contributed only 9%. The most abundant species of rotifer was *Hexarthra* sp. (Table 6.10). Copepod nauplii, with a total of 14,525 individuals, was the most abundant taxon, contributing 31.2% to the total number of zooplankton collected. The remainder of copepods collected were dominated by Cyclopoida.

At Newinga PS, 19 zooplankton taxa were collected compared with 21 taxa at Jericho PS. Six taxa were collected from Jericho PS only while 4 taxa were specific to Newinga PS. The most abundant taxon overall at Newinga PS was Chydoridae (16%) which was not found at Jericho PS (Table 6.10). The most prevalent rotifer at Newinga PS was *Cephalodella* sp. (15%). Copepod nauplii was the most common taxon at Jericho PS and *Hexarthra* sp. was the most abundant rotifer, contributing 38% and 26% respectively to the total number of zooplankton.

At Jericho PS, one-way ANOVA detected a significant difference in mean number of individuals (N) ($F_{2,6} = 56.888$, $p < 0.001$) between times. Post hoc tests suggested lower abundance at time 1 compared to both time 2 and 3 (Fig. 6.7).

Table 6.10: Zooplankton taxa and percentage contribution to the total abundance collected in pumping samples between February and March 2006.

	Newinga	**% Total**	**Jericho**	**% Total**	**All Samples**	**% Total**
Rotifera						
Bdelloidea	800	8.6	15	<0.1	815	1.8
Conochilus sp.	0	0.0	20	0.1	20	<0.1
Filinia sp.	585	6.3	1240	3.3	1825	3.9
Hexarthra sp.	250	2.7	9530	25.6	9780	21.0
Asplanchna sp.	15	0.2	410	1.1	425	0.9
Brachionus sp.	820	8.8	0	0.0	820	1.8
Keratella sp.	0	0.0	1770	4.7	1770	3.8
Plationus sp.	30	0.3	25	0.1	55	0.1
Lepadella sp.	0	0.0	90	0.2	90	0.2
Dicranophorus sp.	105	1.1	0	0.0	105	0.2
Epiphanes sp.	15	0.2	290	0.8	305	0.7
Euchlanis sp.	575	6.2	20	0.1	595	1.3
Lecane sp.	780	8.4	65	0.2	845	1.8
Cephalodella sp.	1430	15.4	65	0.2	1495	3.2
Polyarthra sp.	1180	12.7	2485	6.7	3665	7.9
Testudinella sp.	0	0.0	30	0.1	30	0.1
Trichocerca sp.	500	5.4	25	0.1	525	1.1
Wolga sp.	25	0.3	0	0.0	25	0.1
Copepoda						
Nauplii	530	5.7	13995	37.5	14525	31.2
Calanoida	0	0.0	250	0.7	250	0.5
Cyclopoida	105	1.1	4245	11.4	4350	9.3
Harpacticoida	35	0.4	120	0.3	155	0.3
Cladocera						
Chydoridae	1480	16.0	0	0.0	1480	3.2
Daphniidae	15	0.2	2560	6.9	2575	5.5
Simocephalus	0	0.0	40	0.1	40	0.1
Total Zooplankton	9275		37290		46565	

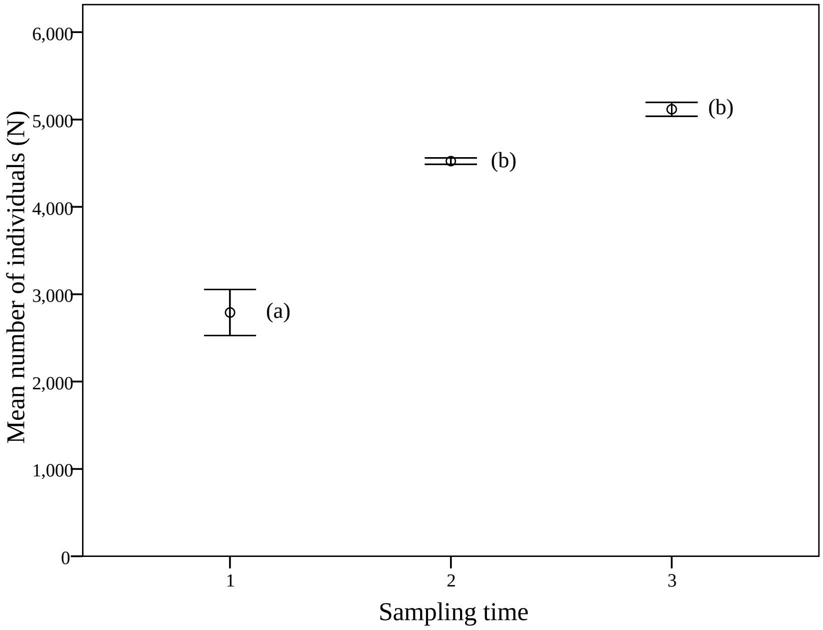


Fig. 6.7: Mean (± s.e.) number of zooplankton individuals for the three sampling times at Jericho PS. Groups suggested by post hoc tests are shown, (a) and (b).

6.4.2 Egg bank study

6.4.2.1 Greenhouse temperatures

Over the first four weeks of the study from the 24th March 2007 to the 21st April 2007, the mean maximum daily temperatures were between 44°C and 50°C and minimum daily temperatures were between 11°C and 22°C (Appendix 34). Temperatures dropped considerably by the final day of the study; 4th June 2007, with a mean night time temperature of 3°C and mean day time temperature reaching only 34°C (Appendix 34). Minimum temperatures in the greenhouse were comparable to historical data of air temperatures recorded in Goondiwindi for the time of year. However, maximum greenhouse temperatures exceeded the historical average for air temperatures in Goondiwindi by as much as 20°C.

6.4.2.2 Water quality and soil organic content

The river water used to flood the pots was slightly alkaline, with a pH of 8.75, and had low salinity and conductivity readings of 0.01% and 0.268 mS cm^{-1}, respectively (Table 6.11). On each sampling occasion the typical storage samples had the highest mean conductivity readings ranging from 0.294 mS cm^{-1} to 0.426 mS cm^{-1} and were the only sites to record salinity levels > 0.01% (Table 6.11 and Appendix 35).

Over all sampling occasions there was no significant difference in the square root of water temperature between the three different pot types ($F_{2,294}$ = 2.003, $p > 0.05$). There was a significant difference in mean salinity (Kruskal-Wallis, $p < 0.001$). One-way ANOVA also detected significant differences in the reflected square root of pH ($F_{2,294}$ = 3.331, $p < 0.05$), conductivity ($F_{2,294}$ = 42.051, $p < 0.001$) and DO ($F_{2,294}$ = 4.798, $p < 0.01$) between pot types. Post hoc tests suggested higher conductivity at typical storage pots than both the control pots and the floodplain pots (Fig. 6.8).

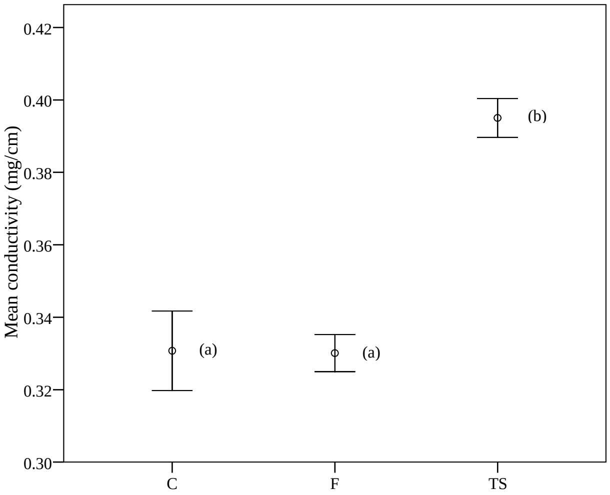


Fig. 6.8: Mean (± s.e.) conductivity (mS cm^{-1}) of control sediment pots (C), floodplain sediment pots (F) and typical storage sediment pots (TS) during the microcosm study from March 2007 to June 2007. Groups detected by post hoc tests are shown in brackets.

Table 6.11: Water quality data for river water (RW), control pots (C), floodplain samples (F) and typical storage samples (TS) from the microcosm sampled between 24th March 2007 and 5th June 2007. Data represents the mean of each site (±s.e.) for each sampling time (0 = Day 0, 1 = Day 2, 2 = Day 7, 3 = Day 10, 4 = Day 14, 5 = Day 17, 6 = Day 21, 7 = Day 24, 8 = Day 28 and 9 = Day 73).

Time	**Site**	**pH**	**Conductivity (mS cm^{-1})**	**Dissolved oxygen (mg l^{-1})**	**Water temperature (^{o}C)**	**Salinity (%)**
0	RW	8.75	0.268	7.55	26.4	0.01
1	C	9.05	0.251	6.71	29.07	0.01
		(±0.06)	(±0.006)	(±0.43)	(±3.67)	(±0.00)
1	F	8.78	0.267	6.51	33.51	0.01
		(±0.09)	(±0.008)	(±0.22)	(±1.41)	(±0.00)
1	TS	9.03	0.294	6.51	31.81	0.01
		(±0.06)	(±0.007)	(±0.16)	(±1.38)	(±0.00)
2	C	8.47	0.284	8.02	23.40	0.01
		(±0.05)	(±0.014)	(±0.44)	(±3.36)	(±0.00)
2	F	8.36	0.339	8.11	26.24	0.01
		(±0.10)	(±0.017)	(±0.26)	(±1.65)	(±0.00)
2	TS	8.36	0.386	8.39	23.66	0.01
		(±0.08)	(±0.012)	(±0.23)	(±1.44)	(±0.00)
3	C	8.28	0.296	7.70	27.00	0.01
		(±0.23)	(±0.016)	(±0.34)	(±2.56)	(±0.00)
3	F	8.46	0.338	8.94	28.27	0.01
		(±0.06)	(±0.015)	(±0.30)	(±1.47)	(±0.00)
3	TS	8.43	0.396	8.90	25.51	0.01
		(±0.10)	(±0.011)	(±0.25)	(±1.37)	(±0.00)
4	C	8.51	0.323	8.08	24.30	0.01
		(±0.01)	(±0.023)	(±0.35)	(±2.24)	(±0.00)
4	F	8.65	0.352	9.26	26.82	0.01
		(±0.08)	(±0.018)	(±0.25)	(±1.25)	(±0.00)
4	TS	8.77	0.412	9.70	25.25	0.01
		(±0.07)	(±0.012)	(±0.30)	(±1.16)	(±0.00)
5	C	8.53	0.330	8.24	23.23	0.01
		(±0.03)	(±0.023)	(±0.50)	(±2.99)	(±0.00)
5	F	8.73	0.343	9.56	25.09	0.01
		(±0.06)	(±0.015)	(±0.28)	(±1.60)	(±0.00)
5	TS	8.82	0.408	9.82	23.73	0.01
		(±0.07)	(±0.013)	(±0.28)	(±1.39)	(±0.00)
6	C	8.31	0.353	8.27	21.00	0.01
		(±0.07)	(±0.028)	(±0.31)	(±2.10)	(±0.00)
6	F	8.51	0.335	9.44	22.82	0.01
		(±0.06)	(±0.014)	(±0.24)	(±1.15)	(±0.00)
6	TS	8.58	0.411	9.51	21.79	0.01
		(±0.07)	(±0.016)	(±0.20)	(±0.99)	(±0.00)
7	C	8.36	0.366	8.06	22.33	0.01
		(±0.07)	(±0.035)	(±0.24)	(±1.92)	(±0.00)
7	F	8.62	0.331	9.35	24.25	0.01
		(±0.07)	(±0.013)	(±0.26)	(±1.20)	(±0.00)
7	TS	8.67	0.410	9.34	23.31	0.01
		(±0.06)	(±0.017)	(±0.21)	(±0.99)	(±0.00)

Table 6.11: Continued.

Time	Site	pH	Conductivity (mS cm^{-1})	Dissolved oxygen (mg l^{-1})	Water temperature (^{o}C)	Salinity (%)
8	C	8.53	0.378	8.38	19.50	0.01
		(±0.03)	(±0.030)	(±0.21)	(±1.51)	(±0.00)
8	F	8.71	0.331	9.62	21.51	0.01
		(±0.08)	(±0.013)	(±0.35)	(±1.19)	(±0.00)
8	TS	8.65	0.411	9.45	20.46	0.01
		(±0.07)	(±0.016)	(±0.22)	(±0.90)	(±0.00)
9	C	8.58	0.396	10.58	14.03	0.01
		(±0.09)	(±0.027)	(±0.27)	(±0.79)	(±0.00)
9	F	8.67	0.335	11.11	14.55	0.01
		(±0.06)	(±0.016)	(±0.15)	(±0.55)	(±0.00)
9	TS	8.70	0.426	11.00	14.47	0.01
		(±0.07)	(±0.014)	(±0.29)	(±0.58)	(±0.00)

There was no significant spatial difference (between pot types) in soil organic content before the experiment began ($p > 0.05$). However, *t*-test did detect a significant spatial difference in the organic content of soil once the experiment had finished ($p < 0.05$). At the end of the microcosm experiment, the organic content of soil was higher in typical storage pots than floodplain pots (Fig. 6.9).

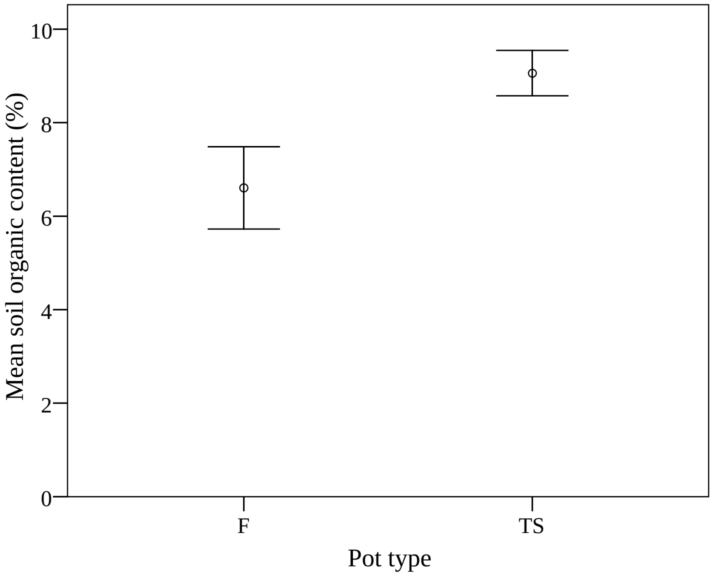


Fig. 6.9: Mean percentage soil organic content (± s.e.) of floodplain (F) and typical storage (TS) pots after the microcosm experiment.

6.4.2.3 Zooplankton assemblages

Of the three samples of filtered river water used to flood the microcosms, one contained no zooplankton and the other two contained only the rotifer, *Lecane* sp. Seventeen taxa from 63,303 individuals were collected from the control replicates over the period of the study (Table 6.12). These consisted of 13 rotifera, 2 collembola, 1 copepod and 1 cladoceran. These taxa have been excluded from further analyses or discussion.

Pooled data from the floodplain pots and typical storage pots consisted of 117,063 individuals from 23 taxa. Rotifers were the most abundant and diverse taxa contributing 86% of the individuals and 66% of the taxa. The most abundant taxon overall was the rotifer, *Epiphanes* sp., contributing approximately 30% of the individuals. The next most abundant taxon was the rotifer, *Trichotria* sp. making up 19% of the total individuals.

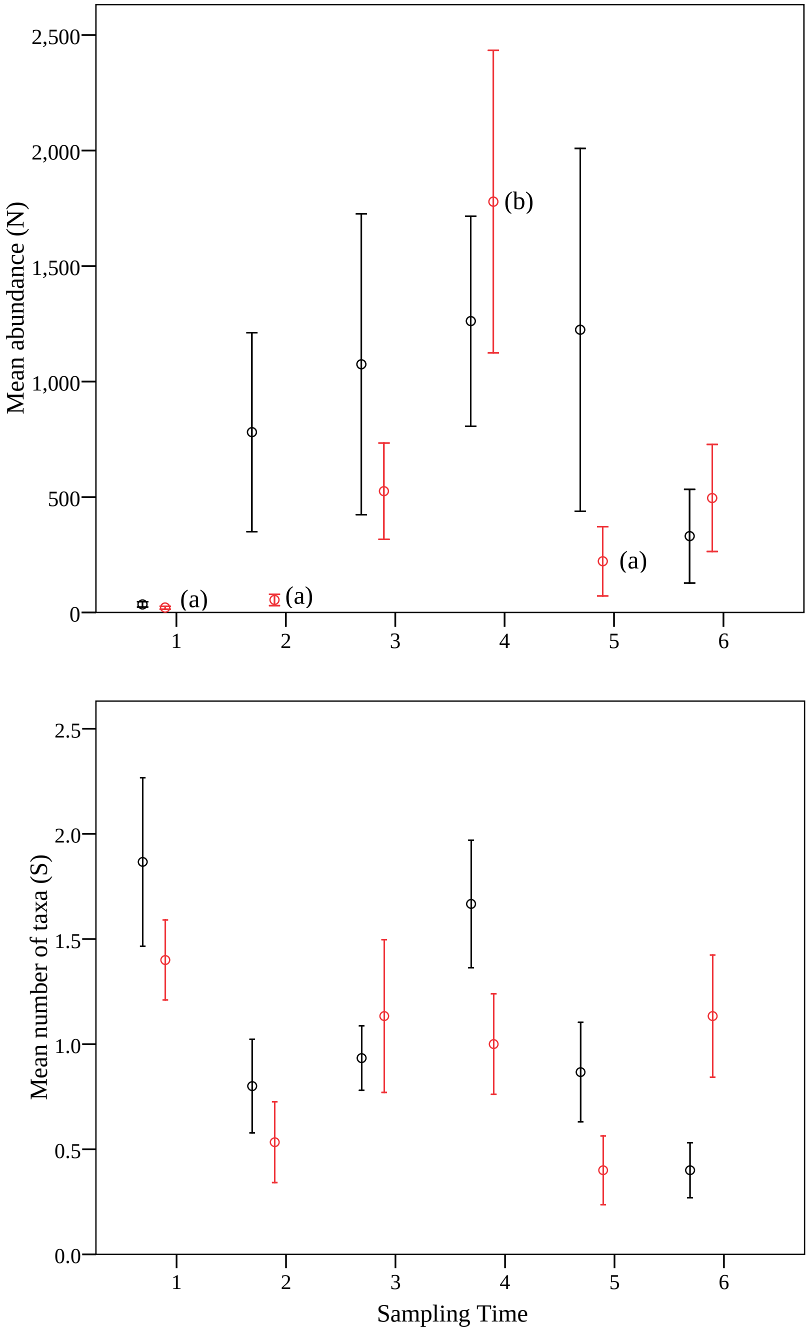
Floodplain pots were more diverse but produced fewer individuals than typical storage pots. In 90 samples from floodplain pots, 46,463 individuals from 20 taxa were collected. In an equal number of samples from typical storage pots, there were 70,600 individuals from 16 taxa. The variation in total individuals between replicate samples was considerable ranging from 0 to 8,520 in floodplain pots and 20 to 11,850 in typical storage pots. Seven taxa were specific to floodplain samples, while three taxa were found only in samples from typical storage pots. The most abundant taxon in floodplain pots was *Eosphora* sp. (46% of individuals) followed by Chydoridae (18% of individuals). In the storage pots the rotifer, *Epiphanes* sp. (47% of individuals) was the most abundant taxon followed by *Trichotria* sp. (25% of individuals) (Table 6.12). The most abundant cladoceran was Chydoridae, while Harpacticoda were the most abundant of the copepod taxon. Collembola were only present in typical storage samples, while Ostracoda were only associated with samples from floodplain pots.

Table 6.12: Zooplankton taxa and percentage contribution to the total abundance collected over the course of the egg bank study; TS = typical storage, F = floodplain. Shaded taxa were those found in the control pots over the course of the study.

	Control	**Percent of Total**	**TS**	**Percent of Total**	**F**	**Percent of Total**
Rotifera						
Bdelloidea sp.	16500	26.1	73135	10.1	55899	7.4
Conochilus sp.	0	0.0	5	<0.1	1300	0.2
Filinia sp.	0	0.0	175	<0.1	29	<0.1
Floscularia sp.	0	0.0	0	0.0	180	<0.1
Ptygura sp.	0	0.0	0	0.0	80	<0.1
Hexarthra sp.	14	0.0	332	<0.1	101	<0.1
Asplanchna sp.	0	0.0	145	<0.1	125	<0.1
Brachionus sp.	1090	1.7	113414	15.7	26983	3.6
Keratella sp.	20	0.0	100	<0.1	45	<0.1
Plationus sp.	0	0.0	75	<0.1	718	0.1
Platyias sp.	0	0.0	3380	0.5	30	<0.1
Colurella sp.	3970	6.3	11848	1.6	2785	0.4
Lepadella sp.	4350	6.9	23270	3.2	57900	7.6
Dicranophorus sp.	180	0.3	0	0.0	1440	0.2
Epiphanes sp.	0	0.0	33245	4.6	1597	0.2
Euchlanis sp.	2650	4.2	39265	5.4	176941	23.3
Gastropus sp.	0	0.0	30	<0.1	0	0.0
Lecane sp.	26560	42.0	299298	41.3	307358	40.5
Lindia sp.	0	0.0	6190	0.9	2240	0.3
Cephalodella sp.	6630	10.5	37703	5.2	52974	7.0
Eosphora sp.	0	0.0	580	0.1	21380	2.8
Monommata sp.	0	0.0	1515	0.2	804	0.1
Scaridium sp.	0	0.0	0	0.0	2760	0.4
Polyarthra sp.	270	0.4	6653	0.9	4010	0.5
Trichocerca sp.	830	1.3	33340	4.6	21702	2.9
Trichotria sp.	0	0.0	17900	2.5	4675	0.6
Wolga sp.	0	0.0	1270	0.2	700	0.1
Unidentified	120	0.2	10170	1.4	1590	0.2
Collembola						
Sminthuridae	59	0.1	418	0.1	374	<0.1
Hypogastruridae	20	<0.1	5	<0.1	2	<0.1
Isotomidae	0	0.0	40	<0.1	0	0.0
Copepoda						
Nauplii	30	<0.1	740	0.1	2589	0.3
Calanoida	0	0.0	80	<0.1	35	<0.1
Cyclopoida	0	0.0	0	0.0	320	<0.1
Harpacticoida	0	0.0	0	0.0	760	0.1
Cladocera						
Bosminidae	0	0.0	425	0.1	0	0.0
Chydoridae	0	0.0	5545	0.8	8370	1.1
Ceriodaphnia	10	<0.1	3830	0.5	687	0.1
Macrothricidae	0	0.0	0	0.0	60	<0.1
Ostracoda	0	0.0	0	0.0	300	<0.1
Total Abundance	**63303**		**724121**		**759843**	

Rotifers were the dominant taxa on the first four sampling occasions, increasing in abundance from sampling time 1. In the floodplain pots, rotifer numbers dropped considerably at sampling time 5 and again in samples from day 75. This drop in rotifer numbers did not occur until sampling time 6 in the typical storage pots. Collembola were associated only with the typical storage pots at sampling times 2 and 3, contributing less than 0.3% of the total individuals. Ostracoda were present only in the floodplain pots on one sampling occasion, making up less than 4% of the total individuals. Cladocerans were not present in the samples until time 4 when they represented less than 4% of the individuals in both floodplain and typical storage pots. After this, their numbers increased dramatically in samples from day 75, to 92% of individuals in typical storage pots and 78% of individuals in floodplain pots.

Kruskal-Wallis and *t*-tests did not indicate any significant difference in the summary variables between pot types across all sampling times (p in all cases > 0.05). In the floodplain pots there was a significant difference in the square root of abundance (N) ($F_{5,84} = 4.986$, $p < 0.001$) and mean number of taxa (S) ($p < 0.05$) between times. Post hoc tests suggested higher abundance at time 4 compared to times 1, 2 and 5 (Fig. 6.10a). The number of taxa (S) was lower at times 2 and 5 compared to all other times (Fig. 6.10b). In the typical storage pots there was a significant difference in the fourth root of abundance (N) ($F_{5,84} = 2.380$, $p < 0.05$) between times, however, post hoc tests did not separate sampling times based on abundance. Post hoc tests suggested higher species richness and Shannon diversity at time 1 than at times 2, 3 and 6. Kruskal-Wallis also detected significant differences in mean number of taxa (S) between sampling times ($p < 0.01$).



(a)

(b)

Fig. 6.10: Mean (± s.e.) abundance (N) and number of taxa (S) of floodplain pots (red) and typical storage pots (black) during the microcosm study from 24th March 2007 to 4th June 2007 (1= Day 2, 2 = Day 7, 3 = Day 10 & 14, 4 = Day 17 & 21, 5 = Day 24 & 28, 6 = Day 75). Groups detected for floodplain pots by post hoc tests shown in brackets.

ANOSIM and 2-way crossed ANOSIM detected highly significant spatial and temporal differences in the zooplankton egg bank assemblages (Table 6.13 and Appendix 36 to 39). In the floodplain pots, samples taken at time 1 were significantly different from all other sampling times. In the typical storage pots, samples from time 2 were not significantly different from any other sampling time, with the exception of time 4 when the absolute abundance data was analysed (Table 6.13 and Appendix 36 to 39).

Table 6.13: Summary of ANOSIM results comparing zooplankton egg bank assemblage structure (based on $\log_{10}(x+1)$, $\log_{10}(x+1)$ standardised for relative abundance and presence absence data) between sampling sites (F = floodplain, TS = typical storage) and sampling occasions (1= Day 2, 2 = Day 7, 3 = Day 10 & 14, 4 = Day 17 & 21, 5 = Day 24 & 28, 6 = Day 75). * $P = 0.001$, ** $P<0.01$, ***$P<0.05$.

Waterbody	**Transformation**	**Factor**	**Global R**	***P***	**Significant pairwise tests**
F	$\text{Log}_{10}(x+1)$Std	Time	0.291	< 0.001	1-4, 1-5, 1-6 < 0.001 2-6, 3-6 * 1-3 ** 1-2, 3-5, 4-5 ***
F	$\text{Log}_{10}(x+1)$	Time	0.289	< 0.001	1-4, 1-5, 1-6, 3-6, 4-6 < 0.001 2-6 * 1-3 ** 1-2, 3-5, 4-5 ***
F	Presence/Absence	Time	0.279	< 0.001	1-5, 1-6 < 0.001 1-4, 2-6, 3-6, 4-6 * 1-2, 1-3, 3-5, 4-5, ***
TS	$\text{Log}_{10}(x+1)$Std	Time	0.146	< 0.01	1-4, 4-6 < 0.001 1-6 * 1-5 ** 3-6, 4-5 ***
TS	$\text{Log}_{10}(x+1)$	Time	0.162	< 0.001	1-4, 4-6 < 0.001 1-5, 1-6 * 2-4, 3-6, 4-5 ***
TS	Presence/Absence	Time	0.154	< 0.001	1-4, 4-6 < 0.001 1-6 * 1-5 ** 3-6, 4-5 ***
F & TS	$\text{Log}_{10}(x+1)$Std	Waterbody	0.136	< 0.001	
F & TS	$\text{Log}_{10}(x+1)$	Waterbody	0.130	< 0.001	
F & TS	Presence/Absence	Waterbody	0.133	= 0.001	
F & TS	$\text{Log}_{10}(x+1)$Std	Time	0.214	< 0.001	1-4, 1-5, 1-6, 2-6, 3-6, 4-6 < 0.001 1-3, 4-5 ** 1-2, 3-5 ***
F & TS	$\text{Log}_{10}(x+1)$	Time	0.222	< 0.001	1-4, 1-5, 1-6, 2-6, 3-6, 4-6 < 0.001 1-2, 1-3, 4-5 ** 2-4 ***
F & TS	Presence/Absence	Time	0.213	< 0.001	1-4, 1-5, 1-6, 2-6, 3-6, 4-6 < 0.001 1-3, 4-5 ** 1-2 ***

Ordination also suggested temporal variation in emerging zooplankton assemblages. In general, both pot types moved in a linear fashion through the ordination space, from a position high on both Axis 1 and Axis 2 at sampling time 1 to a position low on Axis 1 and Axis 2 at sampling time 6 (Fig. 6.10). Plots for relative abundance and presence/absence zooplankton data show a similar pattern (Appendix 40). The arrangement of centroids suggested that temporal variation was greater than spatial variation between floodplain pots and typical storage pots.

In the samples from floodplain sites, similarity was highest within sites at time 1 (41%), while the samples from typical storage sites were most similar at time 4 (34%) (Table 6.14). However, both floodplain and typical storage samples had low within site similarity at all sampling times (less than 42%). There was high dissimilarity between sampling times in both floodplain pots and typical storage pots (greater than 70%) (Table 6.15 and 6.16). Dissimilarity was highest between times 2 and 6 in the floodplain pots and between times 3 and 6 and times 4 and 6 in the typical storage pots. The highest similarity was between times 5 and 6 in the floodplain pots (Table 6.15) and between times 2 and 3 and times 3 and 4 in the typical storage pots (Table 6.16).

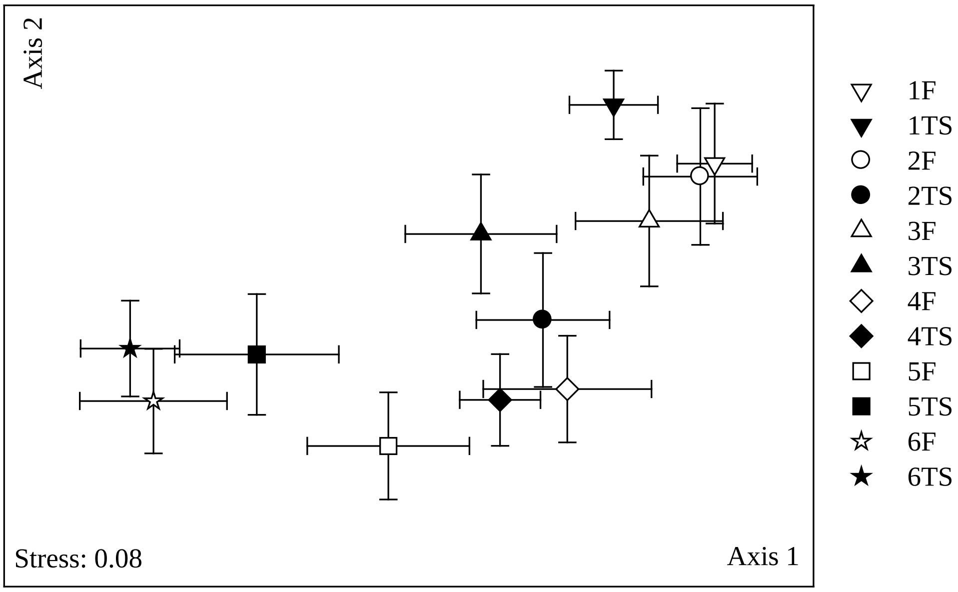


Fig. 6.11: Non-metric multidimensional scaling plot based on log(x+1) transformed data of zooplankton assemblages emerging from soil collected from floodplain sites (F) and typical storage sites (TS) during the egg bank study. 1= Day 2, 2 = Day 7, 3 = Day 10 & 14, 4 = Day 17 & 21, 5 = Day 24 & 28, 6 = Day 75. Samples collected from each sites and time are represented as centroids (average x-y coordinates ± s.e.).

Table 6.14: Summary of SIMPER results for similarity within waterbody types (NW = natural wetland, TS = typical storage) on each different sampling occasion (1= Day 2, 2 = Day 7, 3 = Day 10 & 14, 4 = Day 17 & 21, 5 = Day 24 & 28, 6 = Day 75). Based on $\text{Log}_{10}(x+1)$ data standardised for relative abundance listing taxa contributing >10% to the similarity.

F			**TS**		
Time	**% Similarity**	**Contributing taxa**	**Time**	**% Similarity**	**Contributing taxa**
1	41	*Epiphanes* sp. (89%),	1	29	*Filinia* sp. (47%), *Epiphanes* sp. (15%), *Asplanchna* sp. (15%), *Monommata* sp. (12%)
2	12	*Plationus* sp. (55%), *Eosphora* sp. (29%), *Wolga* sp. (16%)	2	24	*Epiphanes* sp. (70%), *Monommata* sp. (21%),
3	7	*Conochilus* sp. (37%), *Epiphanes* sp. (30%), *Eosphora* sp. (18%), *Plationus* sp. (14%),	3	24	*Epiphanes* sp. (89%)
4	28	*Eosphora* sp. (95%)	4	34	*Epiphanes* sp. (81%)
5	7	Chydoridae (100%)	5	14	*Wolga* sp. (44%), *Monommata* sp. (29%), *Trichotria* sp. (15%), *Epiphanes* sp. (12%)
6	34	Chydoridae (91%)	6	13	*Monommata* sp. (50%), Chydoridae (50%)

Dissimilarity between floodplain and typical storage pots was highest at time 2 (92%) and lowest at time 6 (73%) (Table 6.17). The taxa contributing to the differences between sites were very similar to the taxa contributing to the differences between sampling times for both floodplain sites and typical storage sites. Ten taxa were identified as major contributors to differences between sampling sites and sampling times; *Monommata* sp., *Epiphanes* sp., Chydoridae, *Eosphora* sp., *Plationus* sp., *Trichotria* sp, *Wolga* sp., *Filinia* sp., *Asplanchna* sp. and *Lindia* sp. (Table 6.15, 6.16 and 6.17).

Mean abundances of these taxa displayed considerable spatial and temporal variation (Fig. 6.12). A number of species including *Filinia* sp. and *Plationus* sp. were found during the early sampling times while Chydroidae, *Trichotria* sp. and *Lindia* sp. were only evident in samples from later in the study period. *Plationus* sp. was more abundant in the typical storage pots compared with *Epiphanes* sp. which was more prevalent in samples from floodplain sites (Fig. 6.12).

Table 6.15: Summary of SIMPER results for dissimilarity between times (1= Day 2, 2 = Day 7, 3 = Day 10 & 14, 4 = Day 17 & 21, 5 = Day 24 & 28, 6 = Day 75) at floodplain sites. Based on $Log_{10}(x+1)$ data standardised for relative abundance, listing taxa contributing >5% to the dissimilarity.

Time	**% Dissimilarity**	**Contributing taxa**
1-2	82	*Epiphanes* sp. (31%), *Plationus* sp. (21%), *Asplanchna* sp. (15%), *Eosphora* sp. (13%), *Wolga* sp. (9%), *Filinia* sp. (6%)
1-3	86	*Epiphanes* sp. (28%), *Plationus* sp. (13%), *Conochilus* sp. (12%), *Filinia* sp. (10%), *Asplanchna* sp. (8%), *Monommata* sp. (7%), *Eosphora* sp. (7%), *Lindia* (6%)
1-4	90	*Epiphanes* sp. (29%), *Eosphora* sp. (27%), *Plationus* sp. (9%), *Asplanchna* sp. (8%), *Monommata* sp. (8%), *Filinia* sp. (5%)
1-5	99	*Epiphanes* sp. (30%), Chydoridae (15%), *Monommata* sp. (10%), *Trichotria* sp. (9%), *Wolga* sp. (9%), *Plationus* sp. (8%), *Asplanchna* sp. (7%)
1-6	99	*Epiphanes* sp. (28%), Chydoridae (26%), *Monommata* sp. (9%), *Asplanchna* sp. (8%), *Plationus* sp. (7%)
2-3	89	*Eosphora* sp. (27%), *Plationus* sp. (19%), *Epiphanes* sp. (14%), *Wolga* sp. (9%), *Asplanchna* sp. (9%), *Monommata* sp. (7%), *Lindia* (6%)
2-4	84	*Plationus* sp. (18%), *Eosphora* sp. (15%), *Epiphanes* sp. (13%), *Conochilus* sp. (11%), *Wolga* sp. (8%), *Asplanchna* sp. (8%), *Filinia* sp. (6%), *Lindia* (6%), *Monommata* sp. (6%)
2-5	97	Chydoridae (17%), *Plationus* sp. (16%), *Wolga* sp. (14%), *Eosphora* sp. (12%), *Monommata* sp. (10%), *Trichotria* sp. (10%), *Asplanchna* sp. (8%), *Epiphanes* sp. (8%)
2-6	100	Chydoridae (27%), *Plationus* sp. (15%), *Eosphora* sp. (11%), *Monommata* sp. (9%), *Asplanchna* sp. (8%), *Wolga* sp. (8%), *Epiphanes* sp. (7%)
3-4	87	*Eosphora* sp. (25%), *Epiphanes* sp. (12%), *Conochilus* sp. (11%), *Monommata* sp. (10%), *Lindia* (10%), *Plationus* sp. (9%), *Trichotria* sp. (6%), *Filinia* sp. (6%)
3-5	97	Chydoridae (15%), *Monommata* sp. (12%), *Trichotria* sp. (11%), *Conochilus* sp. (10%), *Wolga* sp. (9%), *Plationus* sp. (7%), *Epiphanes* sp. (7%), *Eosphora* sp. (7%)
3-6	98	Chydoridae (25%), *Monommata* sp. (11%), *Conochilus* sp. (9%), *Plationus* sp. (7%), *Epiphanes* sp. (7%), *Eosphora* sp. (6%)
4-5	95	*Eosphora* sp. (27%), Chydoridae (17%), *Monommata* sp. (13%), *Trichotria* sp. (12%), *Wolga* sp. (10%), *Epiphanes* sp. (7%)
4-6	95	Chydoridae (27%), *Eosphora* sp. (25%), *Monommata* sp. (12%), *Epiphanes* sp. (6%)
5-6	76	Chydoridae (29%), *Monommata* sp. (18%), *Trichotria* sp. (13%), *Wolga* sp. (12%), *Ptygura* sp. (6%), Calanoida (6%), Macrothricidae (6%)

Table 6.16: Summary of SIMPER results for dissimilarity (DisSim) between times (1= Day 2, 2 = Day 7, 3 = Day 10 & 14, 4 = Day 17 & 21, 5 = Day 24 & 28, 6 = Day 75) at typical storage sites. Based on $Log_{10}(x+1)$ data standardised for relative abundance listing taxa contributing >5% to the dissimilarity.

Time	**% DisSim**	**Contributing taxa**
1-2	77	*Epiphanes* (23%), *Filinia* sp. (22%), *Monommata* (19%), *Asplanchna* sp. (13%), *Plationus* sp. (10%), Isotomidae (7%)
1-3	80	*Epiphanes* (23%), *Filinia* sp. (22%), *Monommata* (16%), *Asplanchna* sp. (12%), *Plationus* sp. (10%)
1-4	85	*Epiphanes* (20%), *Filinia* sp. (17%), *Monommata* (13%), *Asplanchna* sp. (10%), *Lindia* sp. (9%), *Plationus* sp. (8%)
1-5	91	*Filinia* sp. (17%), *Monommata* (14%), *Epiphanes* (13%), *Wolga* sp. (12%), *Asplanchna* sp. (9%), *Trichotria* sp. (9%), *Plationus* (8%)
1-6	93	*Filinia* sp. (19%), *Monommata* (18%), Chydoridae (14%), *Epiphanes* (12%), *Asplanchna* sp. (10%), *Plationus* sp. (9%), Calanoida (7%), Bosminidae (7%)
2-3	71	*Epiphanes* (32%), *Monommata* (22%), *Filinia* sp. (18%), Isotomidae (11%)
2-4	74	*Epiphanes* (26%), *Monommata* (18%), *Lindia* sp. (12%), *Filinia* sp. (9%), Isotomidae (8%), *Trichotria* sp. (7%), *Wolga* sp. (6%)
2-5	87	*Epiphanes* (22%), *Monommata* (18%), *Wolga* sp. (15%), *Trichotria* sp. (11%), *Filinia* sp. (8%), Isotomidae (7%), Chydoridae (7%)
2-6	91	*Epiphanes* (24%), *Monommata* (22%), Chydoridae (17%), *Filinia* sp. (9%), Calanoida (9%), Bosminidae (9%), Isotomidae (8%)
3-4	71	*Epiphanes* (27%), *Monommata* (14%), *Lindia* sp. (14%), *Trichotria* sp. (11%), *Filinia* sp. (9%), *Platyias* (8%), *Wolga* sp. (6%)
3-5	86	*Epiphanes* (24%), *Wolga* sp. (16%), *Monommata* (15%), *Trichotria* sp. (13%), *Platyias* (9%), *Filinia* sp. (8%), Chydoridae (7%)
3-6	95	*Epiphanes* (25%), *Monommata* (19%), Chydoridae (17%), *Filinia* sp. (9%), Calanoida (8%), Bosminidae (8%)
4-5	79	*Epiphanes* (25%), *Wolga* sp. (16%), *Monommata* (14%), *Trichotria* sp. (14%), *Lindia* sp. (13%), *Platyias* (8%), Chydoridae (8%)
4-6	95	*Epiphanes* (27%), *Monommata* (16%), Chydoridae (15%), *Lindia* sp. (10%), Calanoida (7%), Bosminidae (7%), *Trichotria* sp. (6%)
5-6	89	*Monommata* (20%), Chydoridae (18%), *Wolga* sp. (16%), *Trichotria* sp. (11%), *Epiphanes* (9%), Calanoida (8%), Bosminidae (8%), *Platyias* sp. (6%)

Table 6.17: Summary of SIMPER results of dissimilarity (DisSim) between sampling sites (F = floodplain, TS = typical storage) across sampling times (1= Day 2, 2 = Day 7, 3 = Day 10 & 14, 4 = Day 17 & 21, 5 = Day 24 & 28, 6 = Day 75). Based on $Log_{10}(x+1)$ data standardised for relative abundance listing taxa contributing >5% to the dissimilarity between waterbodies.

Time	**Comparison**	**% DisSim**	**Contributing taxa**
All times	F v. TS	87	*Epiphanes* sp. (20%), *Monommata* sp. (13%), Chydoridae (9%), *Filinia* sp. (8%), *Eosphora* sp. (8%), *Plationus* sp. (7%), *Trichotria* sp. (6%)
1	F v. TS	74	*Epiphanes* sp. (27%), *Filinia* sp. (21%), *Asplanchna* sp. (15%), *Plationus* sp. (15%), *Monommata* sp. (14%)
2	F v. TS	92	*Epiphanes* sp. (23%), *Plationus* sp. (17%), *Monommata* sp. (13%), *Eosphora* sp. (12%), *Asplanchna* sp. (10%), *Wolga* sp. (9%), *Filinia* sp. (8%)
3	F v. TS	89	*Epiphanes* sp. (23%), *Filinia* sp. (12%), *Monommata* sp. (11%), *Conochilus* sp. (11%), *Plationus* sp. (8%), *Eosphora* sp. (8%), *Lindia* sp. (8%), *Trichotria* sp. (7%)
4	F v. TS	85	*Epiphanes* sp. (26%), *Eosphora* sp. (25%), *Lindia* sp. (13%), *Monommata* sp. (11%), *Trichotria* sp. (8%), *Wolga* sp. (6%)
5	F v TS	82	Chydoridae (20%), *Wolga* sp. (20%), *Monommata* sp. (18%), *Trichotria* sp. (17%), *Epiphanes* sp. (9%), *Platyias* sp. (6%), *Ptygura* sp. (6%)
6	F v. TS	73	Chydoridae (33%), *Monommata* sp. (24%), Calanoida (15%), Bosminidae (10%), Macrothricidae (7%), Harpacticoida (6%)

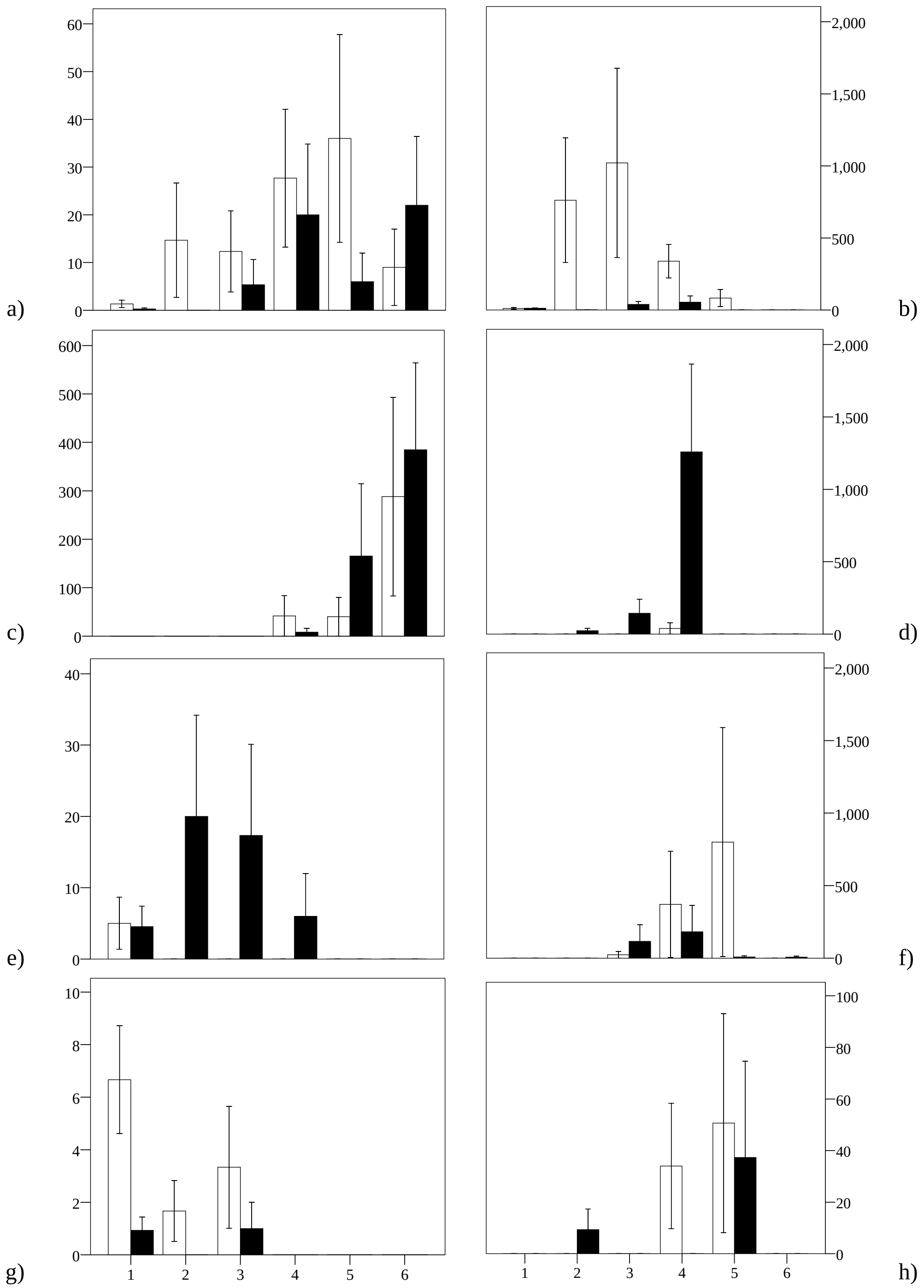


Fig. 6.12: The mean abundance (± s.e.) at each sampling time of ten taxa that were identified by SIMPER as making large contributions to faunal differences between sampling times and pot types (Black bars = typical storages sites, White bars = floodplain sites) a) *Monommata* sp., b) *Epiphanes* sp., c) Chydoridae d) *Eosphora* sp., e) *Plationus* sp., f) *Trichotria* sp., g) *Filinia* sp., h) *Wolga* sp., i) *Asplanchna* sp., j) *Lindia* sp. 1= Day 2, 2 = Day 7, 3 = Day 10 & 14, 4 = Day 17 & 21, 5 = Day 24 & 28, 6 = Day 75.

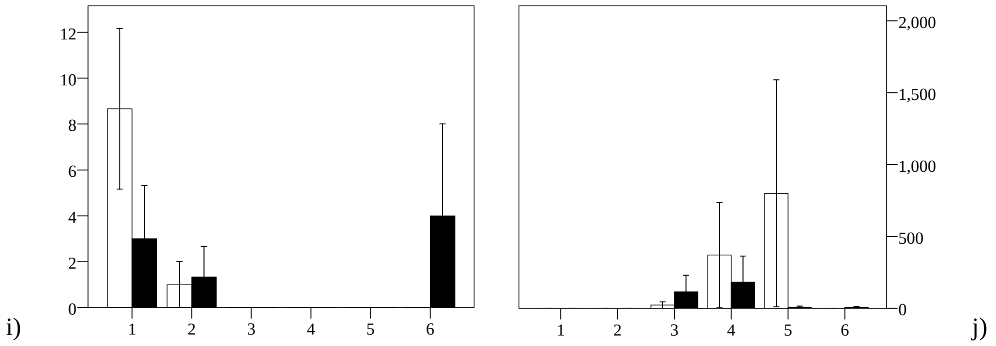


Fig. 6.12: Continued

6.4.3 Colonisation of storages by zooplankton

Two methods of colonisation of typical storages by zooplankton where examined during this study; those species imported with extracted water and those emerging from the sediment eggbank. Pooled data from extracted water, the eggbank of typical storages and samples collected from typical storages during the temporal study (see Chapter 5) consisted of 34 taxa (Table 6.18). Of these, only 8 taxa were common to all three sets of samples, while 13 taxa were found in both extracted water and typical storages but not the eggbank of typical storages. None of the zooplankton taxa collected from typical storages during the temporal study were only found in the eggbank of typical storages and not extracted water. Eight taxa were specific to the eggbank of typical storages and four taxa were only found in samples of extracted water. Only one taxon was specific to samples from typical storages (Table 6.18).

Table 6.18: List of zooplankton comparing presence in extracted water, the eggbank of typical storages and samples from typical storages (TS) during temporal study (see Chapter 5).

Rotifera	
Bdelloidea sp.	▥
Conochilus sp.	■
Filinia sp.	■
Hexarthra sp.	▥
Asplanchna sp.	■
Brachionus sp.	▥
Keratella sp.	▥
Plationus sp.	■
Platyias sp.	░
Lepadella sp.	▨
Dicranophorus sp.	▥
Epiphanes sp.	■
Euchlanis sp.	▥
Gastropus sp.	░
Lecane sp.	▥
Lindia sp.	░
Cephalodella sp.	▥
Eosphora sp.	░
Monommata sp.	░
Polyarthra sp.	▥
Testudinella sp.	▨
Trichocerca sp.	▥
Trichotria sp.	░
Wolga sp.	■
Collembola	
Isotomidae	░
Copepoda	
Nauplii	▥
Calanoida	■
Cyclopoida	▥
Harpacticoida	▨
Cladocera	
Bosminidae	░
Chydoridae	■
Daphniidae	▥
Simocephalus	▨
Macrothricidae	▓

Key:

■	Present in all 3
░	Eggbank only
▨	Extracted water only
▓	TS only
▥	Extracted water & TS

In all three sets of samples, cluster failed to separate the zooplankton taxa into meaningful groups with most groups comprised of only one or two taxa (Table 6.19, Table 6.20 and Table 6.21). The exceptions to this were group ‘A’ created by cluster using the zooplankton data for typical storages during the temporal study (Table 6.20) and groups ‘f’ and ‘m’ created by cluster using the zooplankton data from extracted water (Table 6.19). The manual two-way cluster analysis combining output comparing the zooplankton assemblages found in typical storages with that from the eggbank of typical storages did not identify any definite groups of zooplankton emerging from the eggbank into storages (Fig. 6.13). The second manual two-way cluster analysis comparing the zooplankton assemblages of typical storages with that sampled from extracted water (Fig. 6.14) did identify one group, cluster group ‘A’ (Table 6.20), which appeared to be imported with extracted water. Only two of the taxa in this group, *Filinia* sp. and Calanoida, were also found in eggbank samples from typical storage sediment (Fig. 6.13).

Table 6.19: Species groups created by cluster of zooplankton data from extracted water.

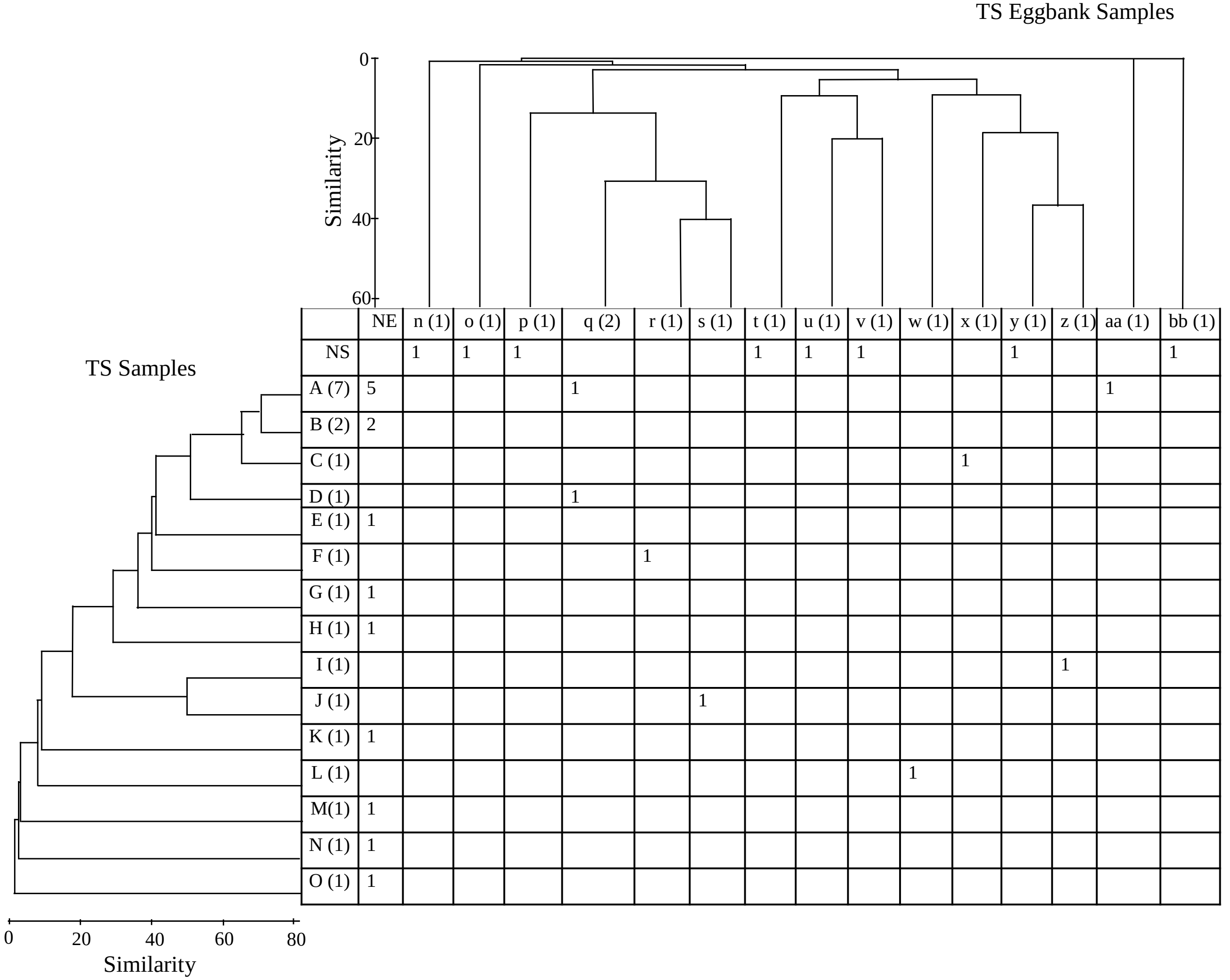
Group	Number of Species	Zooplankton Species
a	1	*Plationus* sp.
b	1	*Testudinella* sp.
c	1	*Conochilus* sp.
d	1	Simocephalus
e	3	*Keratella* sp. *Asplanchna* sp. Calanoida sp.
f	4	*Bdelloidea* sp. *Lecane* sp. *Cephalodella* sp. *Trichocerca* sp.
g	1	*Wolga* sp.
h	2	*Euchlanis* sp. Chydoridae
i	1	Harpacticoida sp.
j	2	*Brachionus* sp. *Dicranophorus* sp.
k	1	*Lepadella* sp.
l	1	*Epiphanies* sp.
m	6	*Hexarthra* sp. Nauplii *Fillinia* sp. *Polyarthra* sp. Cyclopoida Daphnidae

Table 6.20: Species groups created by cluster of zooplankton data from typical storages collected during the temporal study (Chapter 5).

Group	Number of Species	Zooplankton Species
A	7	*Keratella* sp. *Brachionus* sp. Calanoida Cyclopoida Daphnidae *Filinia* sp. Nauplii
B	2	*Hexarthra* sp. *Polyarthra* sp.
C	1	*Epiphanes* sp.
D	1	*Asplanchna* sp.
E	1	*Lecane* sp.
F	1	*Conochilus* sp.
G	1	*Trichocerca* sp.
H	1	*Cephalodella* sp.
I	1	*Wolga* sp.
J	1	*Plationus* sp.
K	1	*Euchlanis* sp.
L	1	Chydoridae
M	1	Macrothricidae
N	1	*Bdelloidea* sp.
O	1	Dicranophorus

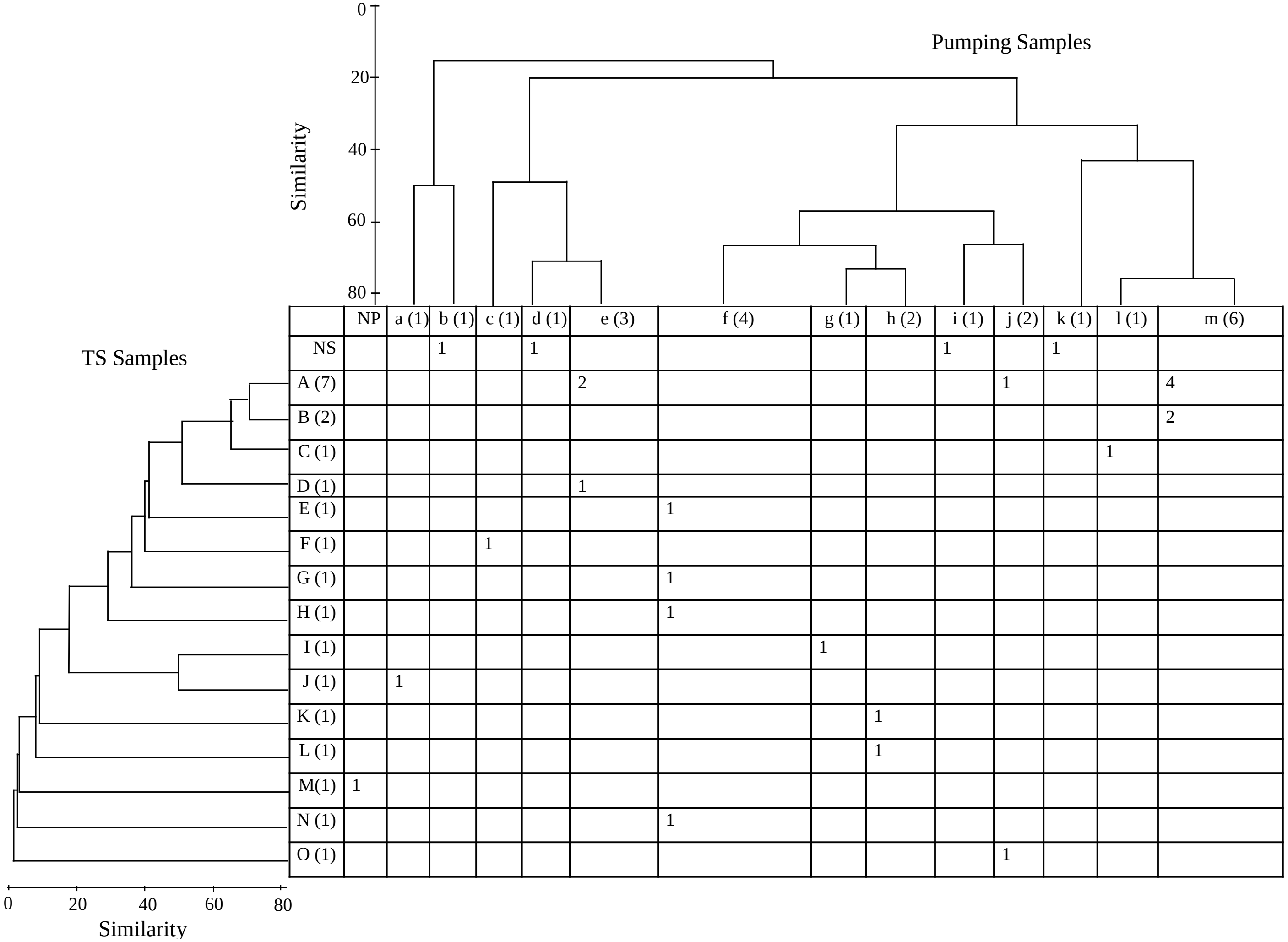
Table 6.21: Species groups created by cluster of zooplankton data from the eggbank of typical storages.

Group	Number of Species	Zooplankton Species
n	1	*Eosphora* sp.
o	1	Isotomidae
p	1	*Gastropus* sp.
q	2	*Filinia* sp. *Asplanchna* sp.
r	1	*Conochilus* sp.
s	1	*Plationus* sp.
t	1	*Trichotria* sp.
u	1	*Platyias* sp.
v	1	*Monommata* sp.
w	1	Chydoridae
x	1	*Epiphanes* sp.
y	1	*Lindia* sp.
z	1	*Wolga* sp.
aa	1	Calanoida
bb	1	Bosminidae



	NE	n (1)	o (1)	p (1)	q (2)	r (1)	s (1)	t (1)	u (1)	v (1)	w (1)	x (1)	y (1)	z (1)	aa (1)	bb (1)
NS		1	1	1				1	1	1			1			1
A (7)	5				1										1	
B (2)	2															
C (1)												1				
D (1)					1											
E (1)	1															
F (1)						1										
G (1)	1															
H (1)	1															
I (1)														1		
J (1)							1									
K (1)	1															
L (1)											1					
M(1)	1															
N (1)	1															
O (1)	1															

Fig. 6.13: Two-way cluster output of the dendrogram based on zooplankton assemblages of eggbank samples and dendrogram based on zooplankton assemblages of typical storages (TS) in the Border Rivers Catchment. NE = not present in eggbank samples; NS = not present in storage samples.



	NP	a (1)	b (1)	c (1)	d (1)	e (3)	f (4)	g (1)	h (2)	i (1)	j (2)	k (1)	l (1)	m (6)
NS			1		1					1		1		
A (7)						2					1			4
B (2)														2
C (1)													1	
D (1)						1								
E (1)							1							
F (1)				1										
G (1)							1							
H (1)							1							
I (1)								1						
J (1)		1												
K (1)									1					
L (1)									1					
M(1)	1													
N (1)							1							
O (1)											1			

Fig. 6.14: Two-way cluster output of the dendrogram based on zooplankton assemblages of extracted water and dendrogram based on zooplankton assemblages of typical storages (TS) in the Border Rivers Catchment. NP = not present in pumped samples; NS = not present in storages samples

6.5 Discussion

6.5.1 Colonisation of storages via extraction of river water

As predicted a number of fish were injured or fatally wounded during the pumping process. Nevertheless, many fish did survive, some with a standard length greater than 200 mm (Appendix 32). However, nets were set fairly close to the pump site. It is possible that if samples were collected further along the channel mortality rates would increase as a result of stress suffered during the extraction process. It is also likely that the number of fish surviving the process will depend on their size and the type of pump used (Baumgartner *et al.*, 2007) and may be species specific (Patrick and McKinley, 1987). The effect of the extraction process on individuals of macroinvertebrates and zooplankton is unknown from this study.

Only five native and three exotic fish species were collected from extracted water compared with the fourteen species of native fish and three exotic species that have been recorded in the Border Rivers Catchment (Moffatt and Voller, 2002). The reduced diversity may have been due to low replication effort and for reasons outlined in Section 4.5. However, diversity was similar to that observed in storages and natural wetlands during the spatial study (see Chapter 4).

An interesting finding was the huge variation in total CPUE between replicate samples at Jericho PS (from 26 to 3,888). The net containing 3,888 fish was the only net set overnight. Increased nocturnal movements of freshwater fish have been demonstrated previously (Pierce *et al.*, 2001; Carol *et al.*, 2007; Rypel and Mitchell, 2007; Koster and Crook, 2008). Although one sample is certainly not conclusive, if further sampling also found increased CPUE at night, limiting water extraction to during the day may be an option for reducing the impact on fish populations.

Only 22 macroinvertebrate taxa were collected across all sampling occasions of extracted water with no significant spatial or temporal differences detected. This represented much lower diversity than that found in any of the waterbody types sampled during the spatial study or other dryland rivers in Australia (see Section 5.4.1). Drift rates of most insect species in streams are consistently higher at night compared to day (Benson and Pearson, 1987; Brewin and Ormerod, 1994; Svendsen *et al.*, 2004). These nocturnal habits may have reduced the number of macroinvertebrate species being extracted from the water during the day. Macroinvertebrates may also be able to avoid extraction by remaining in the benthic zone or swimming near the water surface.

A total of 25 taxa of zooplankton were collected over all sampling occasions from the two pump stations. Significantly lower abundances of zooplankton at sampling time 1 compared to the other two sampling times may reflect a change in flow conditions. At sampling times 1 and 2, the river was flowing. However, by sampling time 2 the water level had dropped and water was only trickling over the weir. At sampling time 3 the river had ceased to flow and landholders had commenced pumping down their private weirs. It is possible that if zooplankton were concentrated near the surface they would be carried with the flow over the weir rather than being extracted from the channel. However, diel migration of zooplankton to lower depths during the day to avoid predation (Zaret and Suffern, 1976; Vuorinen, 1987; Haney, 1988; Lampert, 1989) and deleterious chronic effects of ultra-violet radiation (Williamson, 1995; Casper and Thorp, 2007) is well documented. Shiel *et al.* (1982) found plankton densities in the River Murray, Australia, to be higher at 5 m depths than in subsurface samples. In contrast, rotifers and small zooplankton often exhibit a reverse migration upward during the day and downward at night to avoid predation by larger zooplankton (Ohman, 1990; Williamson, 1990, 1993). Viroux (1999) reported an increased abundance of rotifers in the subsurface layer compared to deeper samples in the River Meuse, Belgium. As the majority of zooplankton collected during this study were rotifers (18 out of 25 taxa), it is possible that they would have occupied the upper layers of the water column.

6.5.2 Colonisation of storages via the egg bank

Over the course of the egg bank study, a total of 20 zooplankton taxa were collected from floodplain pots and 16 from typical storage pots. Only rotifer taxa had begun emerging within 2 days of flooding (sampling time 1), indicating that the response of this group to external stimuli is rapid. In comparison, cladocerans did not appear until sampling time 4, between 17 and 21 days after flooding, and copepods were not encountered until sampling time 6, between 29 and 75 days after flooding. This is in contrast to results from artificial billabongs constructed out of soil from the River Murray floodplain, where all rotifer and microcrustacean taxa began emerging within 1 day of flooding (Nielsen *et al.*, 2000). However, Nielsen *et al.* (2000) did point out that the rapid emergence witnessed in their study may have been a result of controlled conditions and that such rapid emergence would be rare in natural billabongs. The results from this study also suggest that emergence rates of zooplankton were species specific, with some species displaying a rapid response to inundation (e.g. *Plationus* sp., *Filinia* sp.), while others were not encountered until over a week after flooding (e.g. *Lindia* sp., Chydoridae).

In both pot types diversity of emerging zooplankton communities was highest 2 days after flooding (time 1), while abundance was lowest on this sampling occasion. Abundance followed a similar pattern in both floodplain and typical storage pots, increasing up to 21 days after flooding (time 4) and then decreasing. This initial increase in abundance of emerging zooplankton over time was similar to results observed from sediments of the Darling River, Australia (Jenkins *et al.*, 2002; Jenkins and Boulton, 2003).

Although resting stages of many invertebrates are resistant to extremes and can survive high temperatures (Brendonck and De Meester, 2003), hatchling abundance has been shown to decline at temperatures as low as 20 ^{o}C (Vandekerkhove *et al.*, 2005). However, despite the extreme temperatures of over 50^{o}C recorded in the greenhouse, the number of animals that emerged from sediments of both floodplain sites and typical storage sites were comparable with those recorded previously in the Murray-Darling Basin (Skinner *et al.*, 2001) and appeared higher than the estimated number of resting stages in floodplain sediment of the River Murray calculated by Nielsen *et al.* (2000). However, it is generally accepted that no two wetlands are the same (Hillman, 1986) and that most Australian wetlands contain biota unique to that particular wetland (Brock *et al.*, 2003). Therefore, this spatial variation in abundance of zooplankton resting stages is not unexpected.

As expected, the egg banks associated with floodplain sites were more diverse than that found in sediment from typical storages but, in contrast to our hypothesis, were less productive. This could be a result of their more natural variation in inundation.

If storages were managed such that they were permanently inundated, this may have resulted in the absence of environmental cues necessary to trigger the emergence of large numbers of resting stages (Nielsen *et al.*, 2000). As a result, emergence is suppressed over time (Delorme, 1991) and the number of resting stages in the egg bank increases (Nielsen *et al.*, 2000). Nielsen *et al.*, (2000) also suggested that a loss of variability in water levels may result in long term loss of biodiversity. In order to make firm conclusions about the effects of water regimes, it would be necessary to know the flooding history of the storage sites. During the mesocosm study, salinity and conductivity readings were higher in typical storage pots compared with floodplain pots. This was similar to water quality measurements recorded during the spatial study of aquatic assemblages and could be the reason for a less diverse egg bank at storages (see Section 5.3.1). The differences in soil organic content may also be the cause of the variation in species richness observed between floodplain and typical storage sites.

The constructed nature of storages may be contributing to the differences between floodplain sites and typical storage sites. Unlike natural wetlands, the majority of storages are manmade; soil is removed from around the inside perimeter of the storage, forming a borrow pit, and used to build up the embankment wall. The depth of this borrow pit and disturbance to the remainder of floodplain forming the bottom of the storage may have affected the survival and diversity of the egg bank of storages. Garcia-Roger *et al.*, (2005) found a decreasing viability of eggs with depth of soil while Briggs and Jenkins (1997) recorded fewer rotifers emerging from soil collected in parts of lakebeds that had been cropped than soil from undisturbed parts of lakebeds.

6.5.3 Comparison of the egg bank and assemblages of extracted water with the aquatic assemblages of storages

It is likely that extracted river water and the sediment egg bank are the sources of colonisation for fish and zooplankton assemblages found in storages. With only a few exceptions the fish populations of storages were comparable to those sampled from extracted water and the zooplankton assemblages of extracted water and emerging from the egg bank could account for the zooplankton assemblages of storages.

G. holbrooki, which was collected at Jericho PS, was not encountered during the spatial sampling programme but has been recorded previously throughout the Border Rivers Catchment (Moffatt and Voller, 2002). In contrast, *Hypseleotris* spp. and *M. fluviatilis* were recorded in storages in the region during the spatial study (see Chapter 4) but not in extracted water. These differences may be a result of low sampling effort during both the spatial study and the colonisation study.

Abundances of exotic species collected from extracted water were very similar to those found at both types of storage sampled during the spatial study, contributing only one per cent to the total CPUE. This suggests that exotic fish assemblages found in storages are a result of what is being pumped into the storage and has little to do with predation or their breeding capabilities (see Section 4.5). Although *C. carpio* have been identified as facultative drifters (Humphries and King, 2003) they do not appear to be more susceptible to pumping as suggested in Section 4.5. Native fish may have a natural tendency to follow the flow of a waterway (Blackley, 2003). If exotic species do not display this same tendency this could explain the low numbers of exotics collected from extracted water.

The position of pipes in the river channel may also affect the fish species being extracted from the river. In this case, pump stations were extracting water from a private weir. To maximise the amount of water removed from the weir, the end of the suction pipe is positioned as low as possible in the river channel. It would therefore be fair to say, that fish which are normally found in deep water are more likely to be extracted during the pumping process. However, *C. carpio* are renowned for their bottom feeding behaviour (Moffatt and Voller, 2002) but results of this study suggest that they are less susceptible to extraction.

As was found during the spatial study, *N. erebi* was the most common species, present in all 8 nets and contributing over 58% of the total CPUE, which is not unexpected given that it is widespread in central, eastern and northern Australia (Allen *et al.*, 2002; Pusey *et al.*, 2004) and may have become more prolific since European settlement (Moffatt and Voller, 2002). *M. ambigua* was also present in high numbers throughout the study, contributing more than a third (35%) of the total CPUE across all pumping samples (Fig. 6.5). This was in contrast to the fish assemblages recorded at all three waterbody types, where *M. ambigua* made up less than 11% of the total CPUE at each waterbody type; natural wetlands (10%), typical storage sites (5%) and ‘no tailwater’ storages (10%) (see Fig. 4.4). Although *M. ambigua* respond rapidly (< 5 min) to stressors such as netting or confinement in tanks, they are considered to have the ability to recover rapidly from both acute and chronic stress (Carragher and Rees, 1994). This, along with the fact that *M. ambigua* can tolerate a wide range of water quality conditions (Pusey *et al.*, 2004), suggests that those individuals which live through the pumping process should survive in storages. The reduced contribution of *M. ambigua* to fish assemblages of storages, compared to numbers being extracted from the river, may reflect their inability to breed in storages. Even though a rise in water level in the storage after filling could be enough to initiate spawning (Lake, 1967), the age and length at which this species reaches sexual maturity may reduce their breeding potential. Females do not mature until four years at a minimum length of 39.7 cm, while males mature at three years (occasionally two years) at a minimum length of 32.5 cm (Pusey *et al.*, 2004). Although individuals of *M. ambigua* of up to 20 cm were collected in the extracted water, mean length was 4 cm (Appendix 32) indicating that these fish were far from sexual maturity.

The size of fish species collected from natural wetlands generally followed a normal distribution (see Fig. 4.6). The fact that size distributions of fish in extracted water were positively skewed (Fig. 6.6) suggests that smaller fish are more likely to be extracted from the river channel. If fish continue to be extracted during water harvesting it could lead to a fish population in the river dominated by older individuals. Size distributions of fish collected in pumping samples were comparable to samples collected in typical storages where *N. erebi*, *M. ambigua* and *L. unicolor* were all positively skewed (see Fig. 4.6).

Zooplankton assemblages in both extracted water and the egg bank of typical storages were comparable to the zooplankton populations associated with typical storages during the temporal study but lower than that observed in other semi-arid floodplain systems (see Section 5.4.1). The only taxa present in storages that could not be accounted for in pumped water or emerging from the egg bank was the cladoceran, Macrothricidae. In fact there were a number of zooplankton taxa that were recorded in harvested water and emerging from the egg bank but were not present in typical storages. These included several rotifers, Simocephalus and Bosminidae (Cladocera), Harpacticoida (Copepoda) and Isotomidae (Collembola). The short life cycles of zooplankton combined with monthly sampling during the temporal study may explain why these taxa were not encountered in typical storages. Or it may be that after being imported with extracted water or emerging from the eggbank, conditions in the typical storages were unsuitable for these taxa. A number of taxa collected from typical storages (13) were only found in extracted water suggesting that this is their sole mode of colonisation. In comparison the eggbank did not appear to be the only mode of colonisation for any of the zooplankton found in typical storages, as all taxa present in typical storages and emerging from the sediment were also found in extracted water.

In contrast to fish and zooplankton, the macroinvertebrate assemblages of storages could not be accounted for by what is coming in via extracted river water. Only 22 macroinvertebrate taxa were recorded in harvested water compared to the 72 taxa present in typical and ‘no tailwater’ storages. Two orders, Coleoptera and Trichoptera, found in storages were completely missing from samples of extracted water. These results would suggest that many of the macroinvertebrates associated with storages are colonising storages by aerial dispersal from nearby natural wetlands rather than being pumped into storages during water harvesting. It is likely that the highly vagile groups, such as Coleoptera, Hemiptera, Diptera, Ephemeroptera, and Odonata (Nicolet *et al.*, 2004; Schafer *et al.*, 2006; Tronstad *et al.*, 2007), are able to colonise storages in this way. Tronstad *et al.* (2007) found that aerial colonisation and rapid growth rates alone were enough to establish a diverse assemblage of aquatic invertebrates in inundated floodplains.

6.5.4 Conclusions

Due to the limited number of sites and sampling times of extracted water, the results of this study only provide a limited insight into the fauna that were removed from the river channel as a result of water extraction. However, the results did suggest that high extraction rates of fish, zooplankton and, to a lesser extent, macroinvertebrates are associated with water harvesting. Even those fish which survive passage through the pumps and are entrained into the storage are guaranteed mortality, as the only outlets are onto irrigated fields (NMFS, 1994; Moffatt and Voller, 2002). There are a number of fish screens which are used extensively in the USA and other parts of the world to prevent the movement of fish into irrigation offtakes, although they are uncommon in Australia (Blackley, 2003). Managers should investigate the possibility of implementing these fish screens to minimise the effect of water extraction on fish populations in the Border Rivers Catchment and other irrigation areas. Although some invertebrate stages will be able to disperse to natural wetlands once pumped into storages, the truly aquatic taxa will meet with the same fate as fish.

In order to fully understand the effect of water extraction on aquatic communities of rivers, this needs to be the subject of further research. In the future, it may also be necessary to use different sampling techniques; for example, fyke nets with a smaller mesh size may be needed to ensure that numbers of smaller species of fish passing through the pumps are not being underestimated.

To limit evaporation and seepage losses many landholders with multiple storage sites, only fill one storage at a time. Over a number of years this management option could result in this storage being permanently inundated while others on the property remain dry, reducing the heterogeneity of wetlands in the landscape (Brock *et al.*, 1999). Extended periods of dry or more frequent and shorter periods of inundation can lead to either temporary or permanent changes in biodiversity (Nielsen and Brock, 2009). In order to preserve the biodiversity of the egg bank in storages it may be necessary to rotate which storages are holding water so that every storage undergoes fluctuations in water levels, similar to natural floodplain wetlands (Brock *et al.*, 2003).

Only two methods of colonisation of storages were explored in this study: those taxa drifting in the water column (via pump stations) and those emerging from the egg bank. The third method of colonisation, aerial colonisation, was not investigated. Future research in this area will need to examine all three methods of colonisation to provide comprehensive insight into the sources of aquatic assemblages in storages.

Chapter 7 Stable isotope analysis of energy sources supporting the food webs of storages and floodplain wetlands in the Border Rivers Catchment

7.1 Introduction

In floodplain rivers and their associated wetlands, debate remains as to the importance of energy derived from inputs of organic matter from upstream or surrounding riparian vegetation (allochthonous sources) versus the importance of energy derived from local instream production by phytoplankton, benthic algae and other aquatic plants (autochthonous sources). The three main models of river function propose different sources of organic matter fuelling the food webs of large rivers and their floodplains. The river continuum concept (RCC: Vannote *et al.*, 1980), developed to explain function in small streams, proposed the major input of organic matter to aquatic food webs in lowland river systems was terrestrially derived fine particulate and dissolved organic matter (FPOM; DOM) transported downstream from the headwaters. However, later modifications suggested this was applicable only to constricted-channel rivers as distinct from floodplain rivers (Thorp *et al.*, 1998). More specifically for large floodplain rivers, Junk *et al.* (1989) in the Flood Pulse Concept (FPC) argued that the flood pulse was the principle driving force in productivity of the major biota, allowing aquatic consumers access to terrestrial detritus and aquatic macrophytes on the floodplain. In contrast, the riverine productivity model (RPM) identified autochthonous autotrophic carbon as the major contributor fuelling metazoan production (Thorp and Delong, 1994; revised by Thorp and Delong, 2002). In recent years there has been growing support for the RPM concept, namely that algal production is the primary source of organic carbon to aquatic food webs in rivers and in the floodplains of most riverine landscapes throughout the world (Hamilton *et al.*, 1992; Lewis Jr. *et al.*, 2001; Bunn *et al.*, 2003; reviewed in Thorp *et al.*, 2006; Leigh, 2008).

Traditionally, carbon sources often considered important in freshwater wetlands included phytoplankton, benthic and epiphytic algae and detritus originating from aquatic vascular plants (Hamilton *et al.*, 1992; Hecky and Hesslein, 1995; Keough *et al.*, 1996). However, more recent stable isotope studies discounted this theory revealing that macrophytes contribute little carbon to aquatic food webs even when abundant (Hamilton *et al.*, 1992; Keough *et al.*, 1996; Kwak and Zedler, 1997). Food webs of floodplain wetlands are now thought to be largely sustained by algal carbon and inputs of organic matter from riparian vegetation (Hecky and Hesslein, 1995; France, 1997; Herwig *et al.*, 2004; Medeiros, 2004).

In Australia, studies on dryland floodplain rivers and their wetlands have highlighted the importance of autochthonous sources of carbon to the food web. Despite the large amounts of terrestrial carbon available (Medeiros, 2004), autochthonous sources such as algae and zooplankton have been found to be the major source of carbon in these systems (Bunn and Davies, 1999; Bunn *et al.*, 2003; Medeiros, 2004). These sources are also likely to be of particular importance to food webs in floodplain rivers and wetlands of the Border Rivers Catchment.

Although storages are floodplain habitats, their physical shape (see Chapter 3) means they are unlikely to have as much access to allochthonous sources of organic carbon compared with surrounding floodplain wetlands. Raised banks minimise connection with the floodplain reducing access to dissolved organic matter and lack of riparian vegetation around storage banks is likely to limit the energy contributed from terrestrial sources. Storages are therefore expected to be reliant on organic matter that is pumped in during water harvesting and/or autochthonous energy sources. In other words, they are more likely to be acting within the boundaries of the RPM and be reliant on algal production. Medeiros (2004) suggested that, due to high turbidity levels in the Border Rivers Catchment, there would be limited algal production available to food webs. However, even in highly turbid systems such as the Cooper Creek, filamentous algae observed in a narrow band along the shallow littoral zone of waterholes were found to be the major source of energy for aquatic consumers (Bunn *et al.*, 2003).

Food web studies using carbon ($\delta^{13}C$) and nitrogen ($\delta^{15}N$) stable isotopes can identify energy sources and trace the flow of energy through ecosystems (Medeiros, 2004). These methods are based on the fact that certain isotopes fractionate in different ways as elements move through the food web (Fry, 1991) i.e. consumers will have the same isotopic composition as their food source (DeNiro and Epstein, 1978; Fry *et al.*, 1978). The advantage of stable isotope analysis (SIA) over other methods, such as gut content analysis, is that it indicates what an animal is assimilating into the body tissue rather than assuming this from what is ingested. Carbon isotope values ($\delta^{13}C$) show little fractionation across trophic levels (between 0.0 and 1.0‰ per trophic level) (Peterson and Fry, 1987) and so can be used to indicate the sources of carbon which are important to consumers in food webs (Fry, 1991). In comparison nitrogen isotope values ($\delta^{15}N$) exhibit a significant enrichment across trophic levels (between 2.0 and 5.0‰ per trophic level) (Minagawa and Wada, 1984; Fry, 1991) due to the excretion of greater amounts of ^{14}N over ^{15}N in urine (Peterson and Fry, 1987). This predictable fractionation through the food web allows nitrogen isotopes to be used as an indicator of trophic level (Fry, 1991).

The lack of riparian vegetation, bank complexity and hard substrate at storages will ultimately reduce the number and type of carbon sources available to the aquatic food web (see Chapter 1 and Chapter 3). In addition, the high turbidity and steeper banks associated with some storages may restrict the amount of littoral habitat in the photic zone and reduce the availability of benthic sources of algal production (Fellows *et al.*, 2009). Due to this limited access to energy sources it is possible that storages will have food webs that are less variable than those found in natural wetlands. If the diversity of sources is limited at storages in comparison to natural wetlands this may lead to reduced aquatic biodiversity in storage sites. There is also the potential that the mobile aquatic taxa associated with storages may actually be feeding at nearby natural wetlands. If this is the case these taxa are likely to have stable isotope signatures more similar to the food sources present in natural wetlands.

7.1.1 Aims

This chapter aims to (a) investigate the sources of energy supporting food webs within storages using SIA and (b) compare these food webs to those of natural floodplain wetlands. By tracing sources of organic carbon through the food web, SIA will be used to identify the major sources of organic carbon supporting the macroinvertebrate and fish communities in typical storages and natural wetlands. The main objectives of this chapter are:

1) To identify the main source of carbon driving the aquatic food web in natural wetlands and typical storages.
2) To investigate the spatial variation in complexity and variability of food webs and test the hypothesis that the food web of storages will be more dependent on autochthonous food resources than allochthonous food resources due to limited access to energy sources at storages.
3) It is believed that storages will have a limited range of food resources and we aim to explore the possibility that mobile taxa found at typical storages are actually feeding at the more productive natural wetlands.

7.2 Methods

7.2.1 Study Sites

To understand the differences in food webs between artificial and natural waterholes and to identify the main source of organic carbon driving the food webs in each, five natural wetlands and five typical storages were selected for sampling (Table 7.1). The ten sites were all located between Talwood and Mungindi in the Border Rivers Catchment (Fig. 7.1).

Table 7.1: Sites used for stable isotope analyses in April 2008

Site	**Site Code**
Typical Storages	
Jericho Storage 1	TS1
Yattlewondi Storage 4	TS6
Tarrawatta N End Dam	TS7
Yarramildi Feedlot Dam	TS15
Willarie Storage	TS16
Natural Wetlands	
Newinga Weir Waterhole	NW4
Tarrawatta Weir Waterhole	NW7
Kerry’s Creek	NW9
Warril Creek Talwood	NW10
Warril Creek Jericho	NW11

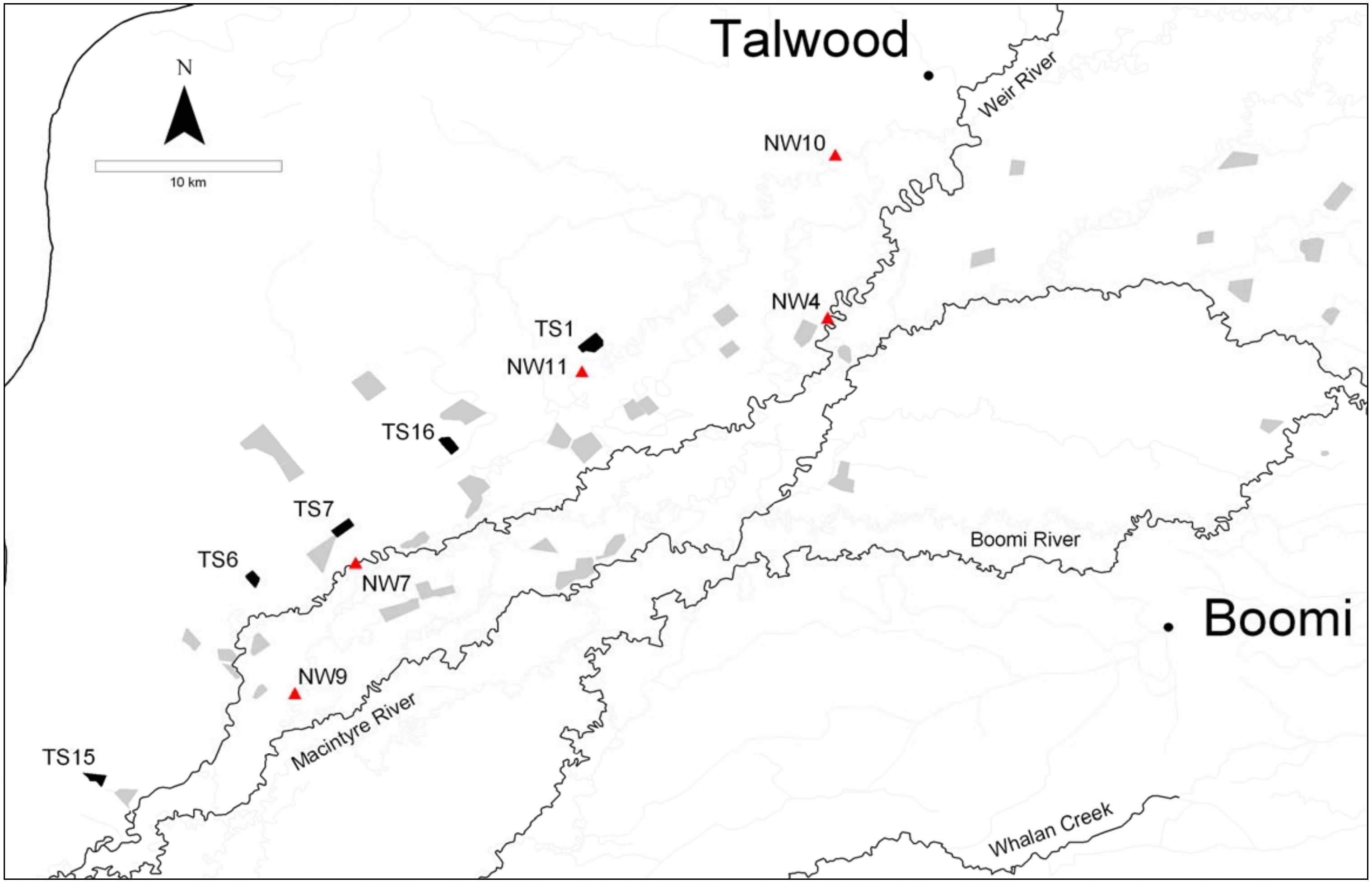


Fig. 7.1: Natural wetlands (triangles) and storage sites (black) used for stable isotope analysis sampling in April 2008. Grey objects represent other storages.

7.2.2 Sampling methods

Sampling for stable isotope analysis was carried out over a week in April 2008. Major sources of aquatic organic carbon and nitrogen were collected along with invertebrate consumers and fish at each study site. The primary sources sampled were seston, algal biofilm attached to woody debris and plants, isolated clumps of filamentous algae and algal scum when present in noticeable quantities; benthic Fine Particulate Organic Matter (FPOM) and benthic Coarse Particulate Organic Matter (CPOM). Consumers included zooplankton, macroinvertebrates and fish.

In most stable isotope studies of food webs, seston is collected from open water (pelagic zone) (e.g. Huryn *et al.*, 2001) while consumers are collected from benthic and littoral zones (e.g. Hadwen and Bunn, 2005). Leigh (2008) did not find any significant differences in stable isotope values between seston collected in the littoral zone to that collected in the pelagic zone. Therefore, in this study it was decided to sample all sources and consumers from the littoral zone. During collection and laboratory processing gloves were worn at all times. Prior to sampling, all equipment was acid washed (10% HCl) and then thoroughly rinsed in site water between sites. These procedures helped to reduce the likelihood of contamination from unknown sources of organic carbon and nitrogen. Unless otherwise stated, three replicates of each source and consumer were collected and all samples upon collection were immediately placed in labelled zip-lock bags and frozen.

7.2.2.1 Primary sources

Seston samples for SIA were collected in buckets within approximately the top 0.5 m of the water column and then filtered through 75 µm mesh to remove large zooplankton, sediment and any other particulates that could contaminate the sample (Leigh, 2008). The water was kept in the shade and stirred manually while known volumes were subsampled from replicate buckets. This subsample was drawn via vacuum (using a hand pump and maintaining pressure below 0.3 – 0.5 atm) onto pre-combusted (550 ^{o}C for 4 hours) 47 mm diameter glass-fibre filters (Whatman GF/F; pore size = 0.7 µm) until no more water would pass through. Each GFF was then placed in acid washed plastic filter paper holders which were wrapped in aluminium foil, sealed in zip-lock bags and frozen until processed in the laboratory.

Benthic POM was sampled using a 250 µm dip net and by taking handfuls of sediment from the bottom of the waterbody. The resulting detritus was sifted through graded sieves in the field into fine (250 µm to 500 µm) and coarse (500 µm to 1 mm) particulate organic matter fractions (FPOM and CPOM).

Algae samples were carefully scraped off submerged wood, plants or leaves using a scalpel. When present, replicate samples of filamentous algal clumps and floating algal scum were also collected.

7.2.2.2 Consumers

Zooplankton (mostly copepod nauplii and calanoid copepods) was collected by towing a 75 µm plankton net slowly through the water just below the surface. Time of collection was noted but all tows were carried out during the day.

Macroinvertebrates were sampled using a 250 µm triangular dip net and a small seine net. The nets were emptied into white sorting trays and individuals were picked out by hand, targeting representatives of each functional feeding group (FFG) when present. Macroinvertebrates were separated into taxa in the field to isolate predators. Trichopteran larvae were removed from their cases upon collection. All macroinvertebrates were placed in aerated water and left in a cool esky for up to 24 hours before freezing to aid gut voiding (Bunn and Boon, 1993). Invertebrates were also collected from baited traps set to target crayfish (*Cherax*) and prawns (*Macrobrachium*). Up to 10 crayfish were collected from each site. As crayfish can grow back their claws, one claw was removed and frozen before the individuals were returned to the waterbody. Mussels (*Velesunio*) and snails (*Physa*) were collected by hand from shallow benthic zones.

Fish were also collected from the small seine net and the baited traps but larger catches were obtained from fyke nets. Three large fyke nets (Body 6 x 1.1 m, wings 7 m long, 12 mm mesh) and three mini fyke nets (Body 1.65 x 0.4 m, wings 2.4 x 1.0

m, 2 mm mesh) were set for a minimum of 6 hours at each site,. Ideally fyke nets should be set overnight but, due to time constraints, this was possible at only half the sites. At the other 5 sites the nets were set early in the morning and retrieved late the same afternoon. Where possible, 5 replicates of each fish species were collected from each site. Fish were frozen whole upon collection. *Nematalosa erebi* were split into three size classes according to their standard length (small < 6 cm, medium 6 to 10 cm and large > 10 cm) and treated as different species. This is due to the fact that juveniles and adult *N. erebi* have different feeding habits (Atkins, 1984; Pusey *et al.*, 2004; Medeiros and Arthington, 2008). Any extra native fish were returned to the waterbody while extra alien species were euthanised using cold-shock and disposed of by burial away from the watercourse edge and human habitation.

7.2.3 Sample preparation

In the laboratory, biofilm, algae, benthic POM and zooplankton samples were all rinsed in distilled water to wash away dirt and debris. They were then examined under a dissecting microscope to remove any other sources of contamination, such as invertebrates. Prior to washing, the benthic POM was sieved again according to the size fractions collected in the field. Samples were then dried at 60^{o}C for a minimum of 24 hours and ground to a fine powder in a ring grinder (Retsch MM200).

In order to remove traces of non-dietary carbonate, certain samples were acid treated. Each zooplankton sample was split in two and half treated in 10% HCl until the sample ceased to effervesce. Subsamples of seston collected on GFF's were acid treated via fumigation (Hamilton *et al.*, 2005). Similar to zooplankton samples, subsamples of FPOM and seston were also acid treated. In all cases the acid treated portion was used for δ^{13}C analysis while the untreated half was used for δ^{15}N analysis (Bunn *et al.*, 1995).

All macroinvertebrate samples were rinsed thoroughly in distilled water Macroinvertebrates were identified to the lowest taxonomic resolution possible using various keys and guides. In the case of the snails, *Physa* sp., the body of the snail was removed from the shell before being dried. Ideally at least three replicates of each taxa from each site were analysed. This allowed average values of δ^{13}C and δ^{15}N for taxa at each site to be calculated. In cases where the sample size was not sufficient to enable isotopic analyses, individuals were pooled.

A section of muscle was removed from the crayfish claw and from the mussel prior to drying. The prawns, *Macrobrachium* sp. were separated into two size classes; those longer than 25 mm from the tip of the rostrum to the base of the carapace (large) and those shorter than 25 mm (small). As the exoskeleton and the digestive tracts could be a source of contamination from unassimilated material (Bunn *et al.*, 2003), these were removed and samples of muscle tissue were taken from the tail prior to drying. After recording standard length, fish were scaled and skinned and a sample of muscle tissue was taken from the area above the lateral line and adjacent to the dorsal fin. For large individuals of *N. erebi* and *Cyprinus carpio* a section of bone was also removed from the spine and dried. Fish samples analysed were from single individuals. All animal samples were dried in an oven at $60^{o}C$ for 24 to 48 hours before being ground to a fine powder using a mortar and pestle.

Percentage carbon, nitrogen and stable isotope ratios ($\delta^{13}C$ and $\delta^{15}N$) were measured using a continuous-flow isotope-ratio mass spectrometer (Micromass Isoprime Eurovector EA3000, Manchester, UK) at Griffith University. Isotopic compositions are expressed in δ values, which is the relative per million (‰) difference between the sample and a standard e.g.:

$$\delta^{13}C \text{ or } \delta^{15}N = [(R_{sample} / R_{standard}) - 1] \text{ x } 1000$$

where R_{sample} is the isotopic ratio for the sample and $R_{standard}$ is the isotopic ratio of the standard (PeeDee belemite carbonate for $\delta^{13}C$ and atmospheric N for $\delta^{15}N$). Samples enriched in ^{13}C or ^{15}N are considered ‘heavy’ and have higher $\delta^{13}C$ and $\delta^{15}N$ values, while samples enriched in ^{12}C and ^{14}N have lower δ values (Fry, 1991). For all sample runs machine precision (standard deviation of standard replicates) ranged from 0.0-0.1‰ for $\delta^{13}C$ and 0.1-0.4‰ for $\delta^{15}N$.

7.3 Data Analysis

7.3.1 Stable isotope analysis

The significance of any differences in δ^{13}C and δ^{15}N values of individual food web components between natural wetlands and typical storages were investigated using a *t*-test (SPSS, independent sample 2 tailed *t*-test). These analyses were based on a mean isotope value for each food web component at a particular waterbody type. The statistical significance of *p*-values for this test was Bonferonni adjusted to reduce Type I errors. This method obtains a new value of significance by dividing the nominated *p*-value (e.g. 0.05) by the number of comparisons performed (Quinn and Keough, 2002). In this case the significance value was reduced to 0.001 (0.05 divided by 36). Descriptive methods (e.g. box plots, bi-plots, scatter plots) were used to explore variation in δ^{13}C and δ^{15}N values between trophic levels, sites and waterbody types. Linear correlations, using Pearson correlation coefficients and the resulting R^2 value, were used to explore the amount of spatial variation in δ^{13}C values for primary, secondary and fish consumers that could be explained by each primary source. The relationship between δ^{13}C values and fish standard length was also examined in this way.

To compare the complexity of food webs between natural wetlands and typical storages, the variance in δ^{13}C and δ^{15}N values were explored. Euclidean distance was used as a measure of similarity of isotope signatures for individual food web components at both waterbody types. The significance of any differences in the similarity of δ^{13}C and δ^{15}N values of individual food web components between natural wetlands and typical storages were investigated using a *t*-test (SPSS, independent sample 2 tailed *t*-test). The statistical significance of *p*-values for this test was Bonferonni adjusted to reduce Type I errors.

Food web analyses were investigated using IsoSource mixing model software (Phillips and Gregg, 2003). This software calculates feasible combinations of primary source stable isotope signatures that explain consumer isotope signatures. In light of other food web studies, carbon signatures were not corrected for trophic fractionation (Hadwen and Arthington, 2007; Hadwen *et al.*, 2007; Leigh, 2008). As the aim of this study was to identify the main source of carbon driving the food web in natural wetlands compared with typical storages, nitrogen isotope signatures were not included in these analyses. The end members used in the IsoSource mixing model were floating algae and algal clumps (when present at sites), biofilm, seston, zooplankton and CPOM. FPOM was excluded on the basis that its isotopic signature was very similar to CPOM. Although zooplankton is technically a microinvertebrate consumer, it was considered one of the primary sources and included as an end member as it is likely that it is providing a proxy isotope signal for the phytoplankton component of the food web (Hadwen, 2002). Zooplankton is also expected to be a major source of energy for macroinvertebrate and fish species in aquatic food webs (Shiel, 1995; Pusey *et al.*, 2004; Balcombe *et al.*, 2005; Medeiros and Arthington, 2008).

An increment of 1% and a tolerance of 0.1% were used in all IsoSource calculations. The minimum and maximum values for percent contributions of primary sources to consumers were reported along with the mean and the standard deviation. Low maximum values indicate that the source is not important while high minimum values indicate that the source may be important. Small ranges between minimum and maximum indicate that the estimates of a sources contribution to the consumer are well constrained (Benstead *et al.*, 2006). These interpretations of the results are based on the assumption that all likely food sources have been collected and included in the subsequent analyses (Benstead *et al.*, 2006). A *t*-test (SPSS, independent sample 2 tailed *t*-test) was used to detect any significant differences in the number of sources making high maximum ($\geq$ 55%) and high minimum ($\geq$ 40%) contributions to individual consumers between natural wetlands and typical storages.

7.4 Results

7.4.1 Comparison of isotope signatures of biota in natural wetland and typical storage sites

In natural wetlands all primary sources had $\delta^{13}C$ values which were lower than those found in typical storages (Fig. 7.2 and 7.3). CPOM, FPOM and zooplankton $\delta^{15}N$ values were higher in typical storages compared with natural wetlands while biofilm and seston $\delta^{15}N$ values were slightly lower in typical storages.

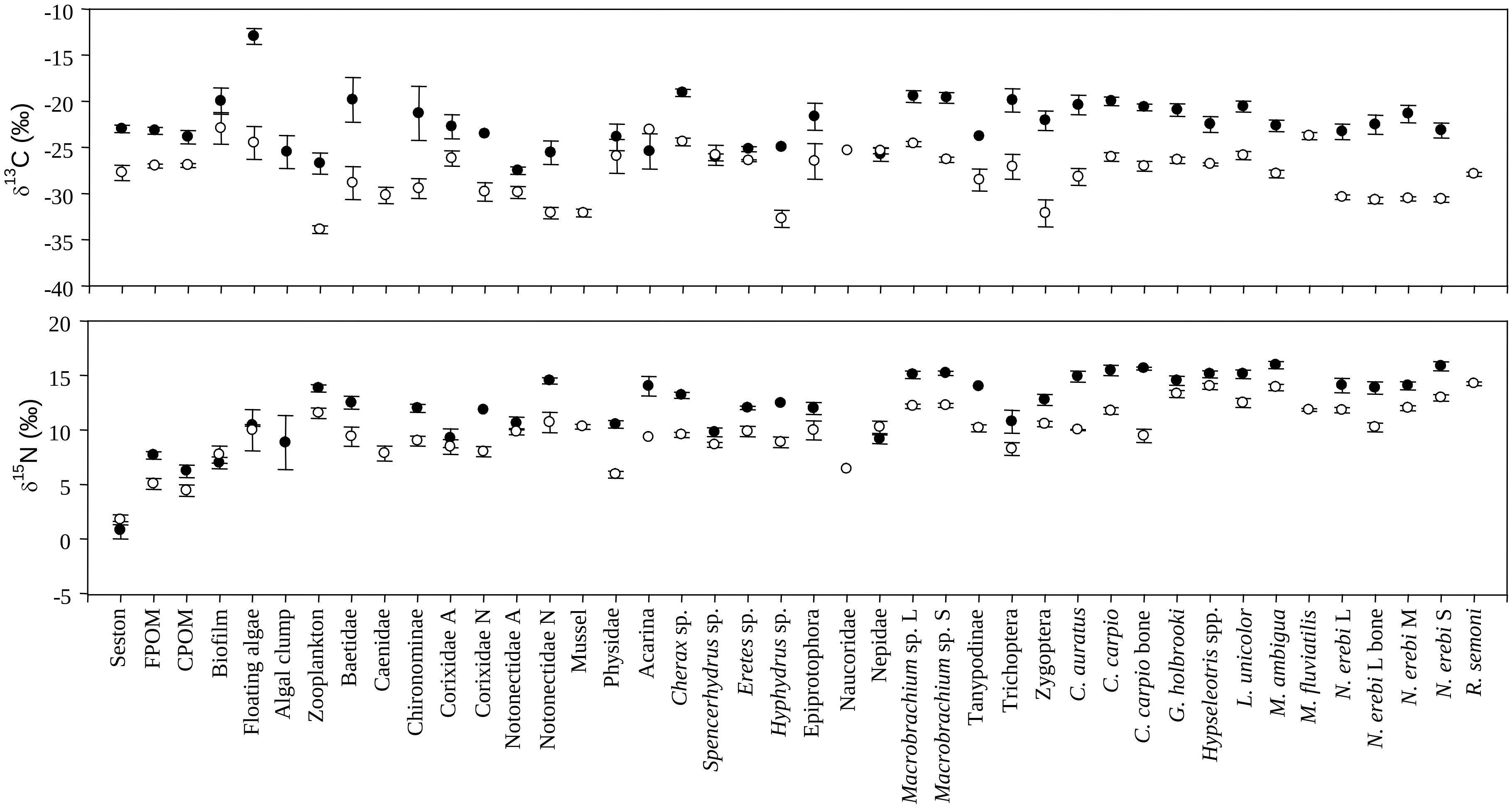


Fig. 7.2: Mean (± s.e.) δ^{13}C and δ^{15}N signatures of all food web components collected from natural wetlands (open symbols) and typical storages (closed symbols) in April 2008.

With the exception of Nepidae, Acarina and *Spencerhydrus* sp., all the food web components were ^{13}C-enriched in typical storages in comparison with natural wetlands (Fig. 7.2). In some cases this enrichment was substantial, with a mean difference in δ^{13}C values of 5.5‰ for primary sources, 5.6‰ for primary consumers, 3.9‰ for secondary consumers and 6.6‰ for the fish. Of the primary sources seston, FPOM and zooplankton were significantly ^{13}C-enriched in typical storages relative to natural wetlands (Table 7.2). *Cherax* sp. along with large and small *Macrobrachium* sp. were the only secondary consumers that displayed a significant enrichment in ^{13}C in the typical storages (Table 7.2). The most obvious variations in ^{13}C values between waterbodies were seen in the fish species. With the exception of *Carassius auratus* and *Hypseleotris* spp. the muscle tissue of all fish species was significantly enriched in ^{13}C in the typical storages (Table 7.2, *t*-test, $p < 0.001$).

In general, δ^{15}N values of the food web components collected were higher in typical storages than natural wetlands (Fig. 7.2); the exception being seston, biofilm and Nepidae. At all trophic levels the mean difference in δ^{15}N values was less than observed for δ^{13}C values; 1.0‰ for primary sources, 2.8‰ for primary consumers, 2.5‰ for secondary consumers and 2.9‰ for the fish. Those differences in δ^{15}N values that were significant were similar to those detected for differences in δ^{13}C values (Table 7.2).

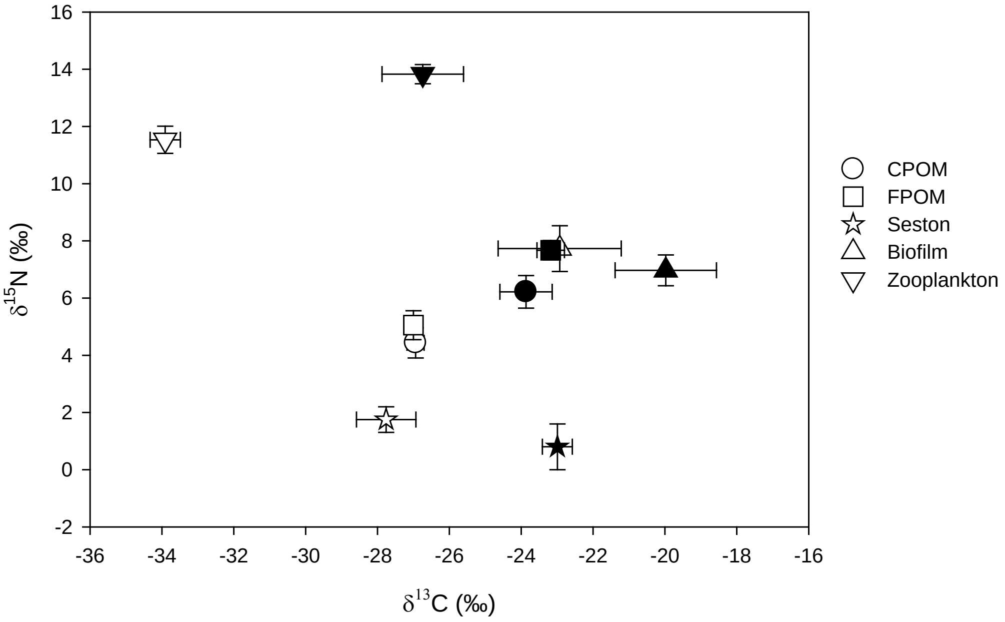


Fig. 7.3: Mean (±s.e.) δ^{13}C and δ^{15}N values for primary sources across all five typical storage sites (closed symbols) and all five natural wetland sites (open symbols)

Table 7.2: Results of *t*-tests comparing mean $\delta^{13}C$ and $\delta^{15}N$ isotope signatures of primary sources and consumers collected from both natural wetlands (NW) and typical storages (TS).

	Sample Size		***P* values for comparisons of $\delta^{13}C$ values from NW & TS sites**	***P* values for comparisons of $\delta^{15}N$ values from NW & TS sites**
	TS	**NW**		
Seston	15	15	0.000*	0.319
FPOM	15	15	0.000*	0.000*
CPOM	15	15	0.001	0.029
Biofilm	11	10	0.201	0.428
Floating algae	3	2	0.007	0.857
Zooplankton	15	15	0.000*	0.000*
Baetidae	3	6	0.022	0.057
Chironominae	3	3	0.061	0.006
Corixidae A	6	9	0.036	0.469
Corixidae N	1	5	0.059	0.031
Notonectidae A	11	14	0.008	0.209
Notonectidae N	9	5	0.001	0.000*
Physidae	3	5	0.411	0.000*
Acarina	2	1	0.612	0.204
Cherax sp.	21	20	0.000*	0.000*
Spencerhydrus sp.	10	5	0.452	0.076
Eretes sp.	2	2	0.067	0.136
Hyphydrus sp.	1	3	0.004	0.081
Epiprotophora	9	8	0.060	0.065
Nepidae	10	2	0.845	0.282
Macrobrachium sp. L	20	20	0.000*	0.000*
Macrobrachium sp. S	25	25	0.000*	0.000*
Tanypodinae	1	5	0.178	0.009
Trichoptera	6	6	0.003	0.068
Zygoptera	4	5	0.001	0.004
C. auratus	10	2	0.003	0.000*
C. carpio	14	15	0.000*	0.000*
C. carpio B	3	2	0.002	0.051
G. holbrooki	21	20	0.000*	0.032
Hypseleotris spp.	3	8	0.033	0.059
L. unicolor	26	10	0.000*	0.000*
M. ambigua	21	21	0.000*	0.000*
N. erebi L	14	14	0.000*	0.006
N. erebi L B	9	12	0.000*	0.000*
N. erebi M	12	23	0.000*	0.000*
N. erebi S	15	21	0.000*	0.000*

*$p < 0.001$ (Significance of *p*-value adjusted according to the Bonferroni method), A=adults, N=nymphs, S=small, M=medium, L=large, B=bone

7.4.2 Primary Sources

In both waterbody types primary sources had $\delta^{15}N$ and $\delta^{13}C$ signatures that spanned a wide range of values (Fig. 7.2 and 7.3). Seston $\delta^{15}N$ values were more depleted than other primary sources (mean site values from -0.6 to 3.1‰ in natural wetlands and from -2.6 to 4.7‰ in typical storages) (Appendix 41). In both waterbody types, but in particular the natural wetlands, CPOM and FPOM had very similar isotopic signatures (Fig. 7.3 and Appendix 41 and 42). Biofilm $\delta^{13}C$ values were generally more enriched than any other primary source even though there was considerable variability across both natural wetlands (mean site values from -15.6 to -28.0‰) and typical storages (mean site values from -14.6 to -24.2‰). Zooplankton samples were more ^{13}C-depleted (-31.5 to -35.9‰ in the natural wetlands; from -21.0 to -32.6‰ in the typical storages) and ^{15}N-enriched (9.5 to 14.6‰ in the natural wetlands; 12.4 to 15.3‰ in the typical storages) (see Appendix 41 and 42) compared with the other primary sources. With the exception of biofilm there was generally less spatial variability in isotopic signatures in natural wetlands compared with typical storages (see error bars in Fig. 7.3).

Seston, CPOM and FPOM had similar ranges of $\delta^{13}C$ values in both typical storages and natural wetlands (Appendix 42). The most obvious differences in $\delta^{13}C$ values were between different types of algae in the typical storage sites (Appendix 42). Mean site values of $\delta^{13}C$ values of biofilm ranged from -14.6‰ to -24.2‰. Although they were both only collected from one site, compared with biofilm, floating algae was more ^{13}C-enriched with a mean site value of -12.9‰ and the algal clump was more ^{13}C-depleted with a mean site value of -25.5‰ (Appendix 42). In comparison, in the natural wetlands there was a wide range in $\delta^{13}C$ values of biofilm (mean site values of -15.6‰ to -28.0‰) and the $\delta^{13}C$ signature for floating algae was within this range. In both waterbody types there was a definite enrichment in ^{15}N from primary sources through the trophic levels to fish (Appendix 41).

7.4.3 Invertebrate Consumers

Isotope signatures for primary and secondary consumers showed considerable spatial variability (see error bars in Fig. 7.4). In general, for both natural wetlands and

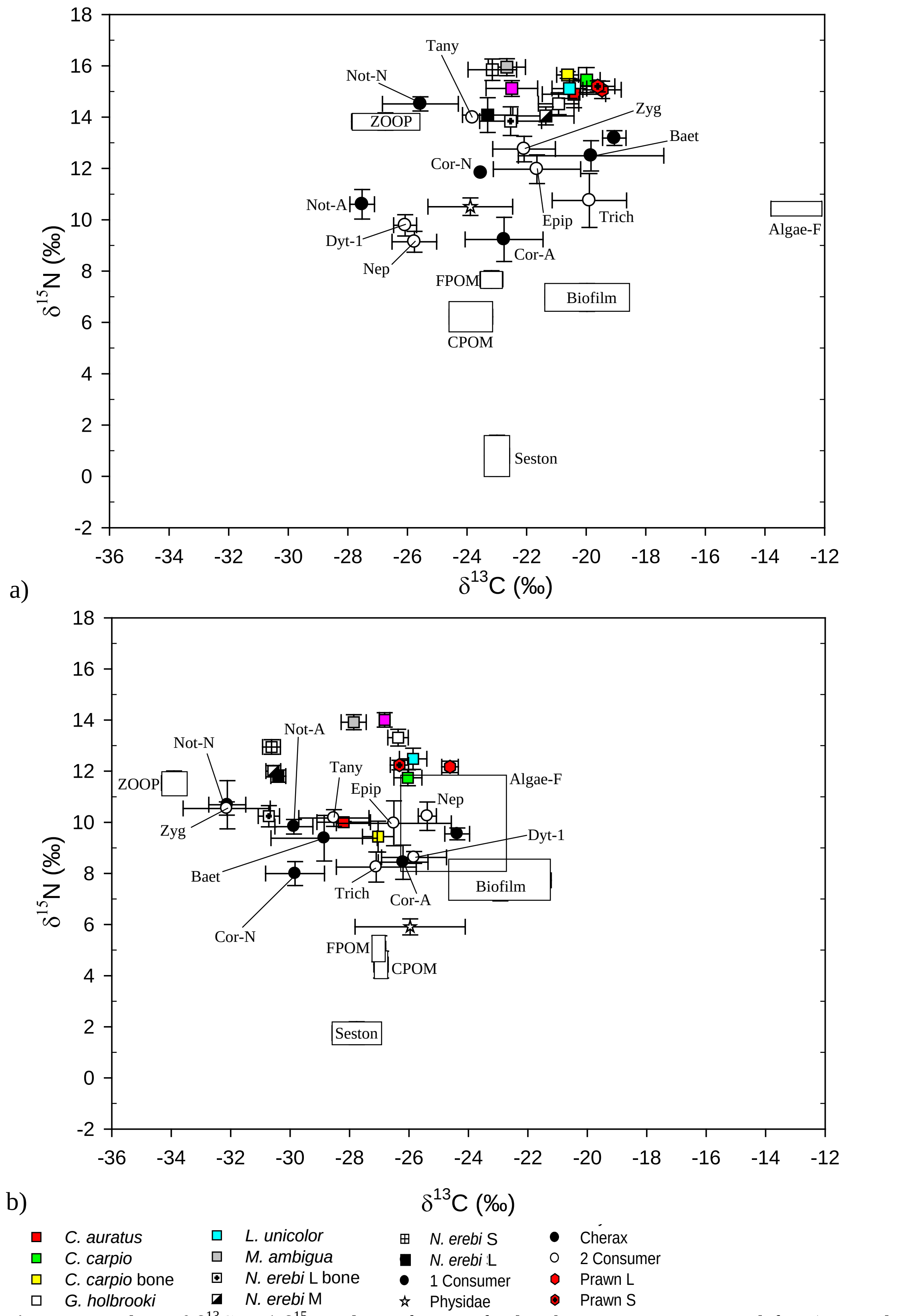


Fig. 7.4: Bi-plots of δ^{13}C and δ^{15}N values of major food web components averaged for a) typical storages and b) natural wetlands. Potential carbon/nitrogen sources are shown as boxes delineated by 1 s.e. while consumers are represented by symbols (mean ± s.e.). Codes represent taxa as follows: Algae F = floating algae, Baet = Baetidae, Cor-A = Corixid adults, Cor-N = Corixid nymphs, Dyt-1 = *Spencerhydrus* sp., Epip = Epiprotophora, Nau = Naucoridae, Nep = Nepidae, Not-A = Notonectid adult, Not-N = Notonectid nymph, Tany = Tanypodinae, Trich = Trichopetera, ZOOP = Zooplankton, Zyg = Zygoptera. L= large, M = medium, S = small.

typical storages invertebrate consumers had $\delta^{15}N$ values which were between the major primary sources and the fish species. In both waterbody types the $\delta^{13}C$ values of invertebrate consumers spanned the range of the major primary sources, but had a wider range of $\delta^{13}C$ than the fish. In general the macroinvertebrate consumers were enriched in both ^{13}C and ^{15}N in the typical storages compared to the natural wetlands. The magnitude of this difference depended on the species (Fig. 7.2).

In both typical storages and natural wetlands large and small prawns, *Macrobrachium* sp., had very similar $\delta^{15}N$ values. However, small prawns were slightly depleted in ^{13}C. In typical storages both large and small prawns had higher ^{13}C and ^{15}N values compared to natural wetlands (Fig. 7.2). See section 7.4.1 for detailed results on differences in isotopic signatures between natural wetlands and typical storages.

The adults and nymphs of both corixids and notonectids had different isotopic signatures. The nymphs of Notonectidae were more ^{15}N-enriched than the adults. This pattern was also apparent for Corixidae nymphs and adults in typical storages but in the natural wetlands the adults were slightly more ^{15}N-enriched (Fig. 7.4). $\delta^{13}C$ values of corixids increased from nymphs to adults at both natural wetlands and typical storages. Notonectids adults were also more ^{13}C-enriched than the nymphs in natural wetlands while in typical storages the opposite was true (Fig. 7.4). Both life stages of corixids and notonectids were enriched in ^{13}C in typical storages compared with natural wetlands. This was also the case for ^{15}N however, the difference between waterbodies was more pronounced in the nymphs of both taxa than the adults (Fig. 7.2). There was no consistent pattern in isotopic signatures between primary macroinvertebrate consumers and secondary macroinvertebrate consumers.

7.4.4 Fish

As expected fish were the most ^{15}N-enriched of the trophic levels collected (Fig. 7.4), ranging from 10.0 to 15.8‰ in the natural wetlands and 12.1 to 17.6‰ in the typical storages for muscle samples across all species (Appendix 41). In general, fish were more ^{13}C-enriched compared to other trophic levels. There was also less variation in isotopic signatures of muscle samples between different fish species in comparison with primary sources and both primary and secondary macroinvertebrate consumers. δ^{13}C values ranged from -16.8 to -27.0‰ in typical storages and -23.1 to -33.1‰ in natural wetlands. This reduced variability in fish signatures compared with primary and secondary consumers was also evident in the bi-plots with fish clustering together in the top right hand corner for both waterbody types (Fig. 7.4).

In natural wetlands, *N. erebi* were the most ^{13}C-depleted fish species collected (Fig. 7.4b), with values ranging from -33.1 to -29.6 ‰ (Appendix 42). This was also true in typical storages (Fig. 7.4a). δ^{13}C values for *N. erebi* muscle also showed more variation in typical storages ranging from -27.0 to -17.1‰ (Appendix 42).

The different size classes of *N. erebi* displayed quite different isotopic signatures (Fig. 7.4). In both waterbody types there was a negative relationship between standard length and δ^{15}N values of *N. erebi* (Table 7.3 and Fig. 7.5). Small *N. erebi* had mean site values for δ^{15}N of between 11.7 and 15.5‰ in natural wetlands and between 14.1 and 17.0‰ in typical storages (Appendix 41). There was no relationship between standard length and δ^{13}C values for *N. erebi* for either waterbody type (Table 7.3). In natural wetlands all three size classes had similar δ^{13}C values (Fig. 7.4b). While in typical storages the medium size class was ^{13}C-enriched in comparison with the large and small size classes. However, there was more spatial variability in typical storage sites (see error bars in Fig. 7.4a).

Leiopotherapon unicolor was the only other species with a negative relationship between standard length and δ^{15}N values but this was only true for typical storage sites (Table 7.3). The standard length of *C. carpio* was positively correlated with δ^{13}C values for both waterbody types (Fig. 7.5), while larger individuals of *Gambusia* *holbrooki*, *Hypseleotris* spp. and *L. unicolor* also tended to be more ^{13}C-enriched but only in natural wetlands (Table 7.3).

At a number of sites bone samples were collected for *C. carpio* and large *N. erebi*. Bone samples of *N. erebi* were more depleted than muscle in ^{15}N in both typical storages and natural wetlands (Fig. 7.4 and Appendix 41). This was also true for *C. carpio* in natural wetlands but in typical storages bone samples of *C. carpio* were slightly more enriched than muscle in ^{15}N. In natural wetlands bone samples of both large *N. erebi* and *C. carpio* were ^{13}C-depleted in comparison with the muscle samples. This was also the case for bone samples of *C. carpio* in typical storages while bone samples of large *N. erebi* were slightly ^{13}C-enriched (Fig. 7.4).

Table 7.3: Pearson's correlation coefficients and associated levels of significance for relationships between isotope values (δ^{13}C and δ^{15}N) and fish standard length (cm) across all natural wetland sites (NW) and typical storage sites (TS). Only relationships based on $n \geq 3$ are shown.

	Length v δ^{13}C				Length v δ^{15}N			
	NW		TS		NW		TS	
Fish Species	*n*	*r*	*n*	*r*	*n*	*r*	*n*	*r*
C. auratus			10	-0.26			10	-0.28
C. carpio	15	0.71**	14	0.71**	15	-0.08	14	0.40
G. holbrooki	20	0.62**	21	0.03	20	-0.34	21	0.12
Hypseleotris spp.	8	0.75*	3	0.97	8	-0.14	3	0.36
L. unicolor	10	0.88**	26	0.17	10	0.58	26	-0.51**
M. ambigua	21	0.34	21	0.23	21	-0.07	21	-0.43
M.fluviatilis	5	0.76			5	-0.37		
N. erebi	58	0.05	41	-0.03	58	-0.31*	41	-0.33*
R. semoni	6	-0.26			6	-0.15		

$*p < 0.05$, $**p < 0.01$, n = sample size

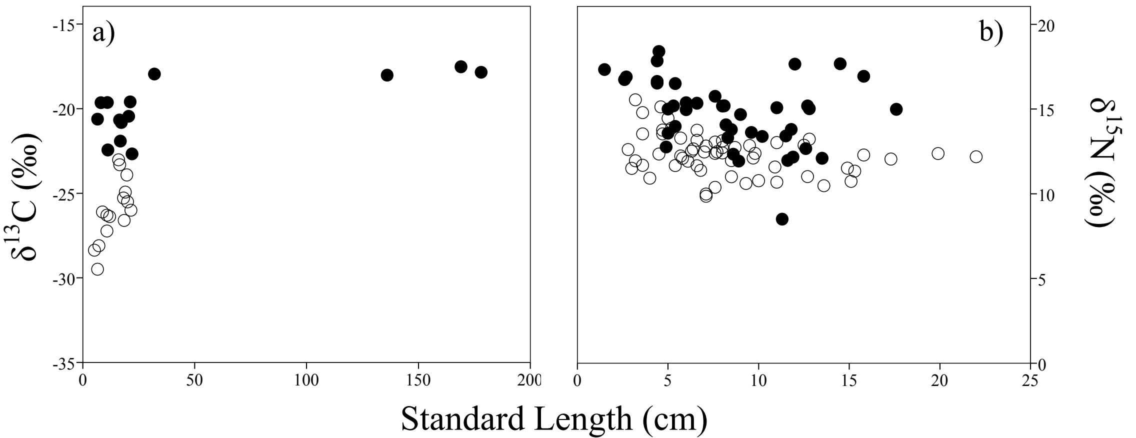


Fig. 7.5: Relationships between isotope values (δ^{13}C and δ^{15}N) and fish standard length are presented for a) *C. carpio* and b) *N. erebi* across natural wetlands (open symbols) and typical storages (closed symbols).

7.4.5 Sources of carbon driving the aquatic food web in typical storages and natural wetlands

IsoSource was used to determine the contribution of primary sources including zooplankton to the diets of the consumer taxa collected using $\delta^{13}C$ values of source and consumer tissues. No solutions were found for a number of consumer taxa because they had $\delta^{13}C$ values that were either too enriched or depleted in relation to the primary sources (Appendix 43).

When considering the results for all consumers, it appeared that seston and CPOM were the two most important sources to consumers in natural wetlands with high minimum (0-95% for seston, 0-92% for CPOM) and maximum (0-100% for seston, 4-100% for CPOM) percentage contributions. This is not unexpected as the bi-plots did suggest that the seston samples are most likely resuspended FPOM from the littoral zone rather than phytoplankton (Fig. 7.4). Biofilm (0-81% min to 7-97% max) and zooplankton (0-84% min to 0-94%) were the next highest contributors to consumer diets. Floating algae (0-74% min to 9-91% max) was also likely to be an important source for consumers at the site where it was present, Tarrawatta Weir Waterhole (NW7).

In contrast, in typical storages, zooplankton (0-77% min to 2-100% max) and biofilm (0-63% min to 7-100% max) were likely to be the most important sources to consumers relative to other sources. CPOM (0-51% min to 9-100% max) and floating algae (0-50% min to 12-63% max) having the next highest combined range of minimum and maximum contributions to consumer diets. Although seston and the algal clump had high maximum scores for percent contributions, 16-99% and 49-100% respectively; neither had high minimum scores (0-0% for both sources).

To identify those sources that were likely to be of most importance for the majority of consumers, sources were ranked based on the frequency that they attained high maximum (> 55%), high minimum (> 40%) and low maximum (< 35%) contributions to consumer diets (Table 7.4) (Leigh, 2008). Unfortunately, using this method, in typical storages no individual source sampled was identified as consistently ranking high based on the three criteria. However, algae, either in the form of biofilm or the algal clump, ranked top for the three criteria. CPOM may have also had an important contribution to diets across all consumers, ranking 2nd twice and 4th once. In natural wetlands floating algae attained high maximum and high minimum contributions 60% and 30% of the time respectively while only attaining low maximum contributions 5% of the time (Table 7.4).

Table 7.4: Ranked importance of primary sources to consumers at typical storage (TS) and natural wetland (NW) sites based on the frequency that each source made high max (> 55%), high min (> 40%) or low max (< 35%) contributions to consumer diets (Leigh, 2008).

TS	Source (frequency high max contributions)	Source (frequency high min contributions)	Source (frequency low max contributions)
1	Algae C (86%)	Biofilm (28%)	Algae C (0%)
2	CPOM (65%)	Algae F (10%)	CPOM (5%)
3	Seston (65%)	Zooplankton (5%)	Seston (8%)
4	Biofilm (54%)	CPOM (3%)	Zooplankton (17%)
5	Zooplankton (47%)	Seston (0%)	Biofilm (21%)
6	Algae F (10%)	Algae C (0%)	Algae F (33%)
NW			
1	Seston (62%)	Algae F (30%)	Algae F (5%)
2	CPOM (60%)	Zooplankton (18%)	CPOM (20%)
3	Algae F (60%)	CPOM (8%)	Seston (25%)
4	Zooplankton (52%)	Biofilm (4%)	Zooplankton (28%)
5	Biofilm (32%)	Seston (2%)	Biofilm (38%)

Based on IsoSource mixing models using δ^{13}C values.

The importance of primary sources to consumer diets varied greatly between consumers and sites (Table 7.5 and Appendix 43). In natural wetlands seston and CPOM appeared consistently important (maximum contributions > 55%) across all sites. In Yarramildi Feedlot Dam (TS15), where the algal clump was present it was important (maximum contributions > 55%) to all consumers with the exception of two macroinvertebrates (Appendix 43). It was also apparent that more than one source could be important (maximum contributions > 55%) to a single consumer suggesting that consumers at all trophic levels could be generalist feeders, for example, in typical storages 75% of primary consumers, 80% of secondary consumers and 70% of fish had high maximum (> 55%) contributions from more than one basal source. In comparison, the occurrence of generalist feeding was lower in natural wetlands with only 55% of primary consumers, 62% of secondary consumers and 69% of fish showing this trait.

Table 7.5: Mean percent contribution of primary sources to consumers at the natural wetland (NW) and typical storage (TS) sites in April 2008. Results were calculated using IsoSource Mixing Model (Phillips and Gregg, 2003) using δ^{13}C values of source and consumer tissues.

	Algal clump				**Biofilm**				**Floating algae**				**Seston**				**Zooplankton**				**CPOM**			
NW4	Mean	SD	Min	Max	Mean	SD	Min	Max	Mean	SD	Min	Max	Mean	SD	Min	Max	Mean	SD	Min	Max	Mean	SD	Min	Max
1^{o} Consumers					26.8	19.3	0.0	81.0					21.6	15.6	0.0	65.0	28.8	3.4	19.0	36.0	22.8	16.5	0.0	69.0
2^{o} Consumers					24.3	16.7	0.0	70.3					34.9	21.0	0.0	88.0	6.2	3.0	0.0	13.7	34.5	23.1	0.0	93.0
Fish					18.8	13.5	0.0	56.8					33.5	14.7	9.3	70.9	25.0	2.4	18.8	30.5	22.6	16.0	0.0	65.0
NW7																								
1^{o} Consumers					12.7	10.2	0.0	51.5	33.8	6.6	22.0	58.0	13.0	10.5	0.0	53.0	26.4	4.3	21.0	42.5	14.2	11.4	0.0	56.8
2^{o} Consumers					12.9	10.4	0.0	52.2	49.8	6.5	39.2	73.8	13.1	10.5	0.0	53.0	10.8	4.2	5.0	26.7	13.4	10.8	0.0	54.2
Fish					16.1	12.9	0.0	65.1	29.6	7.8	18.2	59.0	16.0	12.8	0.0	64.8	22.3	5.2	14.2	41.8	15.9	12.7	0.0	63.9
NW9																								
1^{o} Consumers													21.9	13.0	0.0	45.0	15.4	9.3	0.0	31.8	62.6	3.8	55.0	69.6
2^{o} Consumers													34.0	28.5	3.5	57.5	29.6	20.3	6.9	46.0	36.4	8.9	26.6	43.1
Fish													45.6	26.6	0.0	92.0	32.1	19.0	0.0	65.5	22.4	7.7	8.0	36.0
NW10																								
1^{o} Consumers					22.8	6.4	13.1	40.8					18.4	13.3	0.0	55.5	40.1	7.1	28.5	59.8	18.7	13.6	0.0	56.6
2^{o} Consumers					18.0	7.5	6.8	39.0					21.5	15.5	0.0	64.8	38.5	8.3	25.1	61.6	22.0	15.9	0.0	66.1
Fish					10.5	7.7	0.0	31.9					21.5	15.6	0.0	64.7	45.0	8.6	30.1	68.6	23.0	16.6	0.0	69.4
NW11																								
1^{o} Consumers					27.5	7.7	12.0	48.5					29.4	21.2	0.0	88.0	17.2	12.5	0.0	52.0	25.9	18.7	0.0	77.5
2^{o} Consumers					20.3	5.5	13.6	34.6					23.0	14.2	9.5	61.0	33.3	9.3	22.0	58.0	23.4	14.6	9.2	62.8
Fish					8.4	5.9	0.0	24.5					20.4	14.7	0.0	61.0	47.6	10.4	27.7	75.7	23.6	17.0	0.0	70.8
TS1																								
1^{o} Consumers					16.0	1.4	12.0	19.0					27.5	19.8	0.0	84.0	27.5	19.8	0.0	84.0	29.1	20.9	0.0	88.0
2^{o} Consumers					52.7	1.1	49.8	54.8					15.5	11.3	0.0	47.5	15.5	11.3	0.0	47.5	16.3	11.9	0.0	50.3

Table 7.5: Continued.

	Algal clump				**Biofilm**				**Floating algae**				**Seston**				**Zooplankton**				**CPOM**			
TS1	Mean	SD	Min	Max	Mean	SD	Min	Max	Mean	SD	Min	Max	Mean	SD	Min	Max	Mean	SD	Min	Max	Mean	SD	Min	Max
Fish					44.2	1.2	41.2	46.2					18.3	13.3	0.0	55.8	18.3	13.3	0.0	55.8	19.3	14.0	0.0	58.8
TS6																								
1^{o} Consumers																								
2^{o} Consumers					49.5	6.5	31.0	66.0					22.1	16.0	0.0	69.0	11.9	8.8	0.0	38.0	16.5	12.0	0.0	52.0
Fish					43.8	7.0	25.8	61.2					23.9	17.1	0.0	72.6	13.6	9.8	0.0	41.8	18.7	13.4	0.0	57.6
TS7																								
1^{o} Consumers					24.4	15.7	0.0	61.0					27.1	15.1	0.0	58.7	15.3	10.4	0.0	42.7	33.1	7.0	17.0	47.0
2^{o} Consumers					15.5	10.5	0.0	42.4					15.9	10.1	0.0	41.2	21.7	15.7	0.0	69.0	46.9	10.6	15.0	64.8
Fish					18.1	11.8	0.0	51.5					17.4	11.4	0.0	49.5	34.7	22.9	0.0	86.0	30.0	15.1	0.0	56.5
TS15																								
1^{o} Consumers	16.5	13.2	0.0	66.0					36.5	3.2	31.5	48.5	16.4	13.1	0.0	65.8	14.0	10.1	0.0	51.8	16.7	13.3	0.0	66.5
2^{o} Consumers	18.9	15.0	0.0	75.8					16.6	3.7	11.3	30.5	18.6	14.8	0.0	74.5	26.4	11.6	8.9	69.5	19.4	15.4	0.0	77.9
Fish	18.8	14.9	0.0	75.3					29.8	3.7	23.3	43.7	19.1	15.2	0.0	76.7	14.1	11.3	0.0	56.7	18.2	14.5	0.0	73.2
TS16																								
1^{o} Consumers																								
2^{o} Consumers					21.5	15.5	0.0	65.5					21.8	15.7	0.0	65.0	34.7	0.9	32.8	36.3	22.0	15.8	0.0	67.3
Fish					31.9	22.6	0.0	94.2					31.6	22.3	0.0	93.8	5.7	1.0	3.4	7.4	30.8	22.2	0.0	94.4

Where no data is given IsoSource was unable to compute solutions. Results for individual consumers are given in Appendix 43

The importance of algal carbon, specifically, to consumer diets also varied greatly between sites and between waterbody types (Table 7.6 and Fig. 7.6). Overall, the consumers in one natural wetland site, Tarrawatta Weir Waterhole (NW7), derived more than 50% of their carbon from algal sources, while consumers in the other three natural wetlands, at which algal sources were collected, derived less than 21% of their carbon from algal sources. There appeared to be a more consistent reliance on algae as a carbon source in the typical storages with it contributing over 40% of the carbon to consumer diets in three of the five storage sites; Jericho Storage (TS1), Yattlewondi Storage (TS6) and Yarramildi Feedlot Dam (TS15) and on average 38% across all typical storage sites (Table 7.6 and Fig. 7.6). The importance of algae to consumer diets also varied greatly between taxa. Algae represented a relatively minor source of carbon to both mussels and Nepids while, in contrast, it contributed relatively high proportions of carbon to the diet of Acarina and Trichopteran larvae. No obvious pattern was evident for other consumers with great variability between sites (Table 7.6). However, if we assume that zooplankton provide a proxy for phytoplankton (Hadwen, 2002), then algae, in its various forms, contributed 50% of the organic carbon to consumers in natural wetlands and 54% in typical storages (Fig. 7.6).

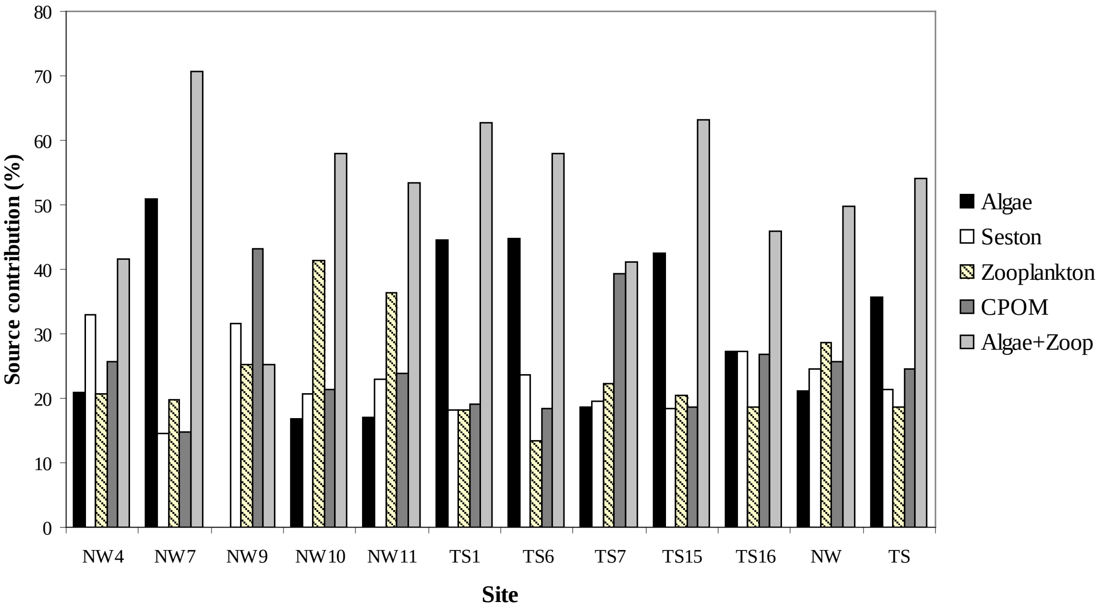


Fig. 7.6: Stable isotope analyses of average percent contribution of dominant carbon sources to consumer diets in individual sites and natural wetlands (NW) and typical storages (TS).

Table 7.6: Mean percent contribution of algal carbon to consumer diets in four natural wetlands (no algal sources were collected from NW9) and five typical storages in April 2008. Percent contributions were calculated using IsoSource Mixing Model (Phillips and Gregg, 2003) using $\delta^{13}C$ values of source and consumer tissues.

	Natural Wetlands				**Typical Storages**					**Means**	
Consumer	**NW4**	**NW7**	**NW10**	**NW11**	**TS1**	**TS6**	**TS7**	**TS15**	**TS16**	**NW**	**TS**
Acarina			59.2					31.9		59.2	31.9
Baetidae			19.2	38.1				62.4		28.7	62.4
Caenidae	26.8		4.5							15.7	
Chironominae			8.6					54.3		8.6	54.3
Corixidae A		75.1	12.6	16.8	16.0		4.7	28.6		34.8	16.4
Corixidae N		42.0	9.1							25.6	
Spencerhydrus sp.		79.6	12.0	15.0				27.6	32.3	35.5	30.0
Eretes sp.				10.3			5.9			10.3	5.9
Hyphydrus sp.							7.9				7.9
Epiproctophora	33.3		6.5	84.7	58.4		9.3	27.9		41.5	31.9
Nepidae				14.6			11.7	15.7	29.9	14.6	19.1
Notonectidae A	18.7	27.0	5.3	4.4				19.4	16.2	13.9	17.8
Notonectidae N			2.6	2.0		49.5	42.6	20.7	7.7	2.3	30.1
Tanypodinae	21.0	40.1	13.7	8.5				40.6		20.8	40.6
Trichoptera		62.7	44.7				42.1	66.7		53.7	54.4
Zygoptera			10.7					43.9		10.7	43.9
Physidae			79.4				26.3			79.4	26.3
Cherax sp.		83.8	34.7	29.0	44.5			59.0		49.2	51.8
Prawn L		78.9	19.8	20.4	54.0			54.2		39.7	54.1
Prawn S		66.8	15.9	13.7	53.8			49.9		32.1	51.9
Mussel		5.9	4.6							5.3	
C. auratus			11.7		55.5			42.6		11.7	49.1
C. carpio B			14.7		4.6					14.7	4.6
C. carpio		80.1	13.9	10.1	49.3			57.7		34.7	53.5
G. holbrooki	2.0	65.9	14.6	9.9	50.1	52.5		48.9	31.5	23.1	45.8
Hypseleotris spp.	12.4	46.6						47.8		29.5	47.8
L. unicolor			17.6	15.7	49.9			48.3	36.1	16.7	44.8
M. ambigua	31.3	36.2	11.9	7.9	65.1	17.3		46.2	27.8	21.8	39.1
M. fluviatilis		85.8								85.8	
N. erebi L	21.7	28.1	3.5			39.9	18.8		31.5	17.8	30.1
N. erebi LB	17.6	28.9	2.3			47.7			32.4	16.3	40.1
N. erebi M	20.0	23.4	7.5	3.4		61.7				13.6	61.7
N. erebi S	24.4	25.5	7.5	3.6	61.0		17.3			15.3	39.2
R. semoni	21.0	37.1								29.1	
Mean	**20.9**	**51.0**	**16.7**	**17.1**	**46.9**	**44.8**	**18.7**	**42.6**	**27.3**	**27.6**	**37.5**

A = adult, N = nymph, S = small, M = medium, L = large, B = bone, Prawn = *Macrobrachium* sp.

It would be expected that the isotope signatures of the major source of organic carbon in these waterbodies would track the signatures of consumers in space and time. In other words, the isotope signatures of consumers should be positively correlated with the isotope signatures of their food sources (Fig. 7.7). However, there was variation between consumers, trophic levels and sites. In both waterbody types there did not appear to be a strong positive correlation between any of the sources sampled and mean δ^{13}C values for primary consumers ($p > 0.05$). The only significant positive relationship was between Baetidae and zooplankton in natural wetlands ($r = 0.99$, $p < 0.05$).

Seston explained 79% of the variation in mean δ^{13}C values for secondary consumers in natural wetlands ($r = 0.89$, $p <0.05$) and this relationship was also observed for *Cherax* sp. ($r = 0.91$, $p < 0.05$) and Tanypodinae ($r = 0.97$, $p < 0.01$) when individual taxa were examined. In typical storages the variation in mean δ^{13}C values for secondary consumers was best explained by CPOM ($r = 0.98$, $p < 0.01$) and this was replicated in small *Macrobrachium* sp. separately ($r = 0.94$, $p < 0.05$). However, δ^{13}C values for notonectid nymphs were positively correlated with CPOM in natural wetlands ($r = 1.00$, $p < 0.01$) and with zooplankton in typical storages ($r = 0.99$, $p < 0.05$).

Algae explained 81% of the variation in mean δ^{13}C values for fish in typical storages ($r = 0.90$, $p < 0.05$). However, when individual species were examined separately, the δ^{13}C values of *G. holbrooki* ($r = 0.92$, $p < 0.05$), *L. unicolor* ($r = 0.99$, $p < 0.05$) and *Macquaria ambigua* ($r = 0.91$, $p < 0.05$) in typical storages all displayed a positive relationship with zooplankton. In comparison, the variation in mean δ^{13}C for fish in natural wetlands was explained most effectively by seston ($r = 0.91$, $p < 0.05$) and this was also observed for small *N. erebi* separately ($r = 0.98$, $p < 0.01$). In contrast, medium *N. erebi* at natural wetlands were positively correlated with algae ($r = 0.99$, $p < 0.01$).

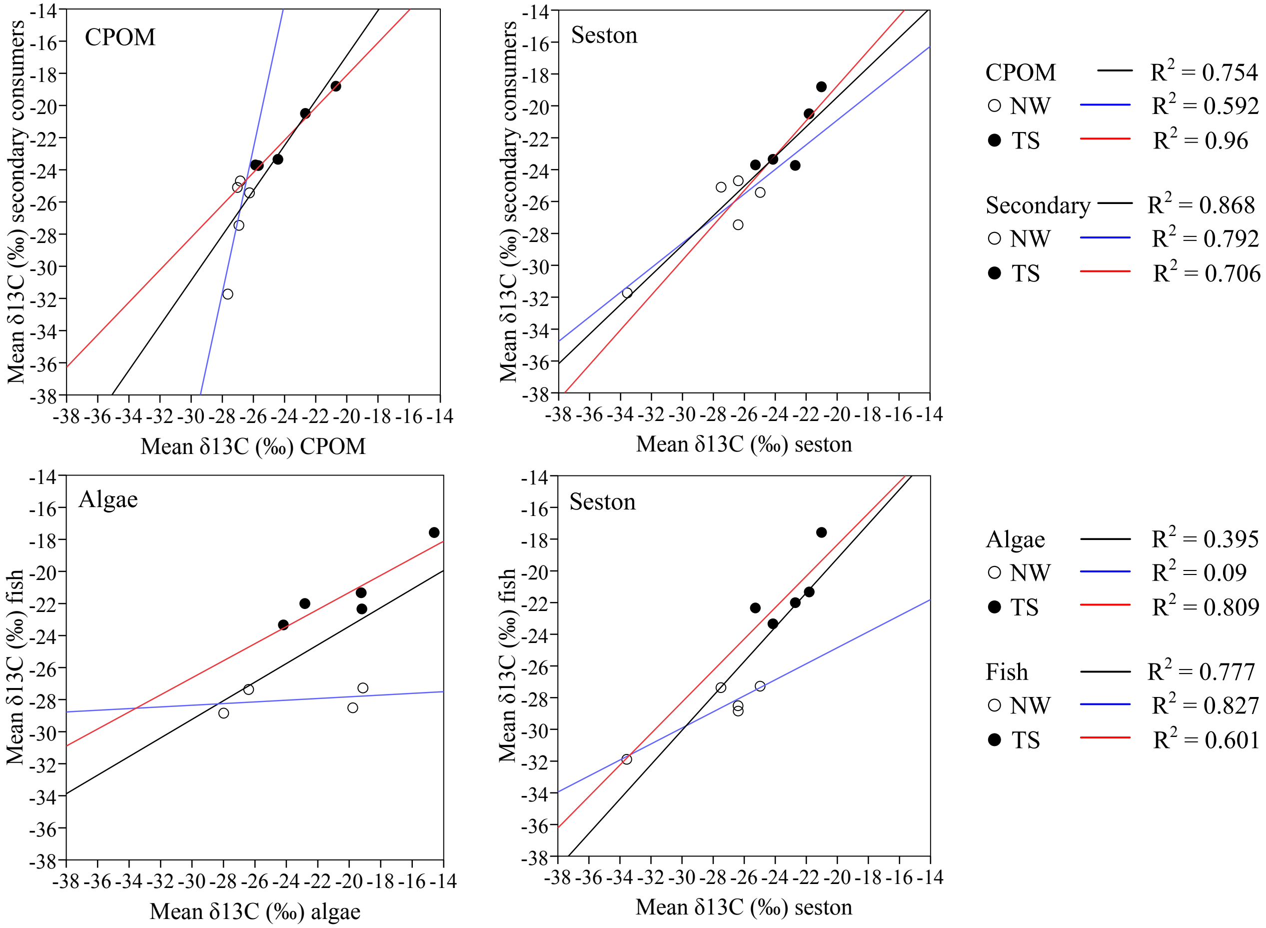


Fig. 7.7: Correlation for significant relationships in site mean δ^{13}C values (‰) between primary sources and consumers in natural wetlands (NW) and typical storages (TS), showing linear correlation trend lines and R^2 values.

7.4.6 Comparison of food web variability between natural wetland and typical storage sites

Variance values were greater in typical storages for $\delta^{15}N$ values across all trophic levels. The variance in $\delta^{13}C$ values was greater in typical storages for all consumers but less for the sources (Table 7.7). Across all trophic levels, there were also significant differences in similarity of $\delta^{15}N$ and $\delta^{13}C$ signatures of individual food web components between the two waterbody types (Table 7.8). In the majority of cases there was decreased similarity in isotope signatures of food web components in typical storages compared to natural wetlands (Appendix 44 and 45).

There was more generalist feeding in storages compared with natural wetlands (see Section 7.4.5) suggesting increased variability of sources at these sites. However, no significant differences between waterbody types were detected in the number of sources making high maximum and minimum contributions to consumers (Table 7.9).

Table 7.7: Descriptive statistics for $\delta^{13}C$ and $\delta^{15}N$ values in natural wetlands (NW) and typical storages (TS).

	$\delta^{13}C$		$\delta^{15}N$	
All Consumers	**NW**	**TS**	**NW**	**TS**
Mean	-27.91	-22.09	10.35	12.47
Median	-27.60	-22.00	10.90	13.50
Variance	11.22	13.83	10.77	16.60
Sources				
Mean	-27.95	-23.16	6.09	7.15
Median	-27.35	-23.35	5.65	7.21
Variance	18.48	17.11	15.61	21.00
Primary Consumers				
Mean	-29.26	-24.04	9.06	11.65
Median	-29.95	-24.55	9.10	11.80
Variance	10.82	17.11	3.36	5.57
Secondary Consumers				
Mean	-26.19	-21.28	10.83	13.11
Median	-25.60	-21.10	10.80	13.60
Variance	9.93	13.47	3.07	6.34
Fish				
Mean	-28.38	-21.59	12.52	14.98
Median	-29.00	-21.40	12.40	15.10
Variance	6.3	9.46	2.68	3.52

Table 7.8: Results of *t*-tests comparing mean Euclidean distance values between $\delta^{13}C$ and $\delta^{15}N$ isotope signatures of primary sources and consumers collected from both natural wetlands (NW) and typical storages (TS). Only food web components with sample size > 4 included.

	Number of distance values		p values for comparisons of Euclidean distance values between $\delta^{13}C$ signatures from NW & TS sites	p values for comparisons of Euclidean distance values between $\delta^{15}N$ signatures from NW & TS sites
	TS	NW		
Seston	105	105	0.000*	0.000*
FPOM	105	105	0.000*	0.000*
CPOM	105	105	0.000*	0.290
Algae (all types)	136	78	0.248	0.912
Zooplankton	91	105	0.000*	0.005
Corixidae A	15	36	0.112	0.684
Notonectidae A	55	91	0.000*	0.000*
Notonectidae N	36	10	0.000*	0.028
Cherax sp.	210	190	0.583	0.000*
Spencerhydrus sp.	45	10	0.023	0.000*
Epiprotophora	36	28	0.236	0.021
Macrobrachium sp. L	190	190	0.000*	0.000*
Macrobrachium sp. S	300	300	0.000*	0.020
Trichoptera	15	15	0.855	0.028
C. carpio	91	105	0.523	0.000*
G. holbrooki	210	190	0.000*	0.000*
L. unicolor	325	45	0.000*	0.000*
M. ambigua	210	210	0.000*	0.206
N. erebi L	91	91	0.000*	0.000*
N. erebi L B	36	66	0.000*	0.062
N. erebi M	66	253	0.000*	0.063
N. erebi S	105	210	0.000*	0.015

*$p < 0.002$ (Significance of *p*-value adjusted according to the Bonferroni method), A=adults, N=nymphs, S=small, M=medium, L=large, B=bone.

Table 7.9: Number of sources making a ≥ 55% maximum contribution or ≥ 40% minimum contribution to a consumer's diet. Maximum of 4 sources (CPOM, seston, algae & zooplankton). Shows *p* values of *t*-tests comparing mean number of sources between natural wetlands (NW) and typical storages (TS).

	NW4	**NW7**	**NW10**	**NW11**	**TS1**	**TS6**	**TS7**	**TS15**	**TS16**	***p* values**
# of sources with ≥ 55% max. cont.										
Corixidae A	NS	1	3	3	3		1	4	NS	0.778
Notonectidae A	2	4	1	1		NS	NS	4	0	1.000
Epiproctophora	3	NS	1	1	1	NS	2	4		0.579
Spencerhydrus sp.		1	3	3	NS	NS		4	3	0.302
Macrobrachium sp. S	NS	2	2	3	1	NS	NS	4	NS	0.898
Macrobrachium sp. L	NS	1	2	3	1	NS		3	NS	1.000
Cherax sp.	NS	1	3	2	3	NS		3	NS	0.272
N. erebi S	3	4	1	1	1	NS	2		NS	0.555
N. erebi M	1	4	1	1		1	NS		NS	0.685
N. erebi L	2	4	1			3	2		3	0.742
C. carpio		1	3	3	0			3	NS	0.596
M. ambigua	3	3	3	3	1	3	NS	4	3	0.705
G. holbrooki	1	3	3	3	0	2	NS	4	3	0.809
# of sources with ≥ 40% min. cont.										
Corixidae A	NS	1	0	0	0		1	0	NS	1.000
Notonectidae A	0	0	1	1		NS	NS	0	1	1.000
Epiproctophora	0	NS	1	1	1	NS	0	0		0.519
Spencerhydrus sp.		1	0	0	NS	NS		0	0	0.495
Macrobrachium sp. S	NS	1	0	0	1	NS	NS	0	NS	0.789
Macrobrachium sp. L	NS	1	0	0	1	NS		0	NS	0.789
Cherax sp.	NS	1	0	0	0	NS		0	NS	0.495
N. erebi S	0	0	1	1	1	NS	0		NS	1.000
N. erebi M	0	1	1	1		1	NS		NS	0.685
N. erebi L	0	0	1			0	0		0	0.423
C. carpio		1	0	0	1			0	NS	0.789
M. ambigua	0	0	0	0	1	0	NS	0	0	0.391
G. holbrooki	1	0	0	0	1	0	NS	0	0	1.000

NS = no solution in IsoSource; where no data is given the taxa was not present at that site

7.4.7 Isotopic signatures of mobile taxa compared with resident taxa

Resident macroinvertebrate taxa showed a distinct separation in isotopic signatures between those found in natural wetlands and those in typical storages (Table 7.10); suggesting that the isotopic values of these taxa are reflecting the food sources available at each waterbody type. With the exception of Nepidae, the moderately mobile taxa displayed a similar pattern (Table 7.10).

Table 7.10: Mean $\delta^{13}C$ and $\delta^{15}N$ values (± s.e.) for sources, resident taxa, moderately mobile taxa, highly mobile taxa and very highly mobile taxa across typical storages and natural wetlands. Only taxa common to both waterbody types are included.

		Typical Storage		**Natural Wetland**	
	Mobility	$\delta^{13}C$	$\delta^{15}N$	$\delta^{13}C$	$\delta^{15}N$
Biofilm	Source	-20.0	7.0	-22.9	7.7
		(±1.4)	(±0.5)	(±1.7)	(±0.8)
CPOM	Source	-23.9	6.2	-26.9	4.4
		(±0.7)	(±0.6)	(±0.2)	(±0.5)
FPOM	Source	-23.2	7.4	-27.0	5.1
		(±0.4)	(±0.3)	(±0.2)	(±0.5)
Seston	Source	-23.0	0.8	-27.8	1.8
		(±0.4)	(±0.8)	(±0.8)	(±0.4)
Zooplankton	Source	-26.7	13.8	-33.9	11.5
		(±1.1)	(±0.3)	(±0.4)	(±0.5)
Physidae	Resident	-23.9	10.5	-26.0	5.9
		(±1.4)	(±0.3)	(±1.9)	(±0.3)
Acarina	Resident	-25.4	14.0	-23.1	9.3
		(±1.9)	(±0.9)	(±0.0)	(±0.0)
Cherax sp.	Resident	-19.1	13.2	-24.4	9.5
		(±0.4)	(±0.3)	(±0.4)	(±0.2)
Macrobrachium sp. L	Resident	-19.5	15.1	-24.6	12.2
		(±0.6)	(±0.3)	(±0.3)	(±0.2)
Macrobrachium sp. S	Resident	-19.6	15.2	-26.3	12.2
		(±0.6)	(±0.2)	(±0.3)	(±0.2)
Baetidae	Moderately	-19.8	12.5	-28.9	9.4
		(±2.4)	(±0.6)	(±1.8)	(±0.9)
Chironominae	Moderately	-21.3	12.0	-29.5	9.0
		(±2.9)	(±0.4)	(±1.1)	(±0.4)
Tanypodinae	Moderately	-23.8	14.0	-28.5	10.2
		(±0.0)	(±0.0)	(±1.2)	(±0.3)
Trichoptera	Moderately	-19.9	10.7	-27.1	8.2
		(±1.2)	(±1.1)	(±1.3)	(±0.6)
Zygoptera	Moderately	-22.1	12.8	-32.1	10.5
		(±1.1)	(±0.5)	(±1.5)	(±0.3)
Epiproctophora	Moderately	-21.7	12.0	-26.5	10.0
		(±1.5)	(±0.6)	(±1.9)	(±0.9)
Nepidae	Moderately	-25.8	9.1	-25.4	10.2
		(±0.7)	(±0.4)	(±0.3)	(±0.6)
Corixidae A	Highly	-22.8	9.2	-26.2	8.4
		(±1.3)	(±0.9)	(±0.8)	(±0.7)
Corixidae N	Highly	-23.5	11.8	-29.8	8.0
		(±0.0)	(±0.0)	(±1.0)	(±0.5)
Notonectidae A	Highly	-27.5	10.6	-29.9	9.8
		(±0.4)	(±0.6)	(±0.6)	(±0.3)
Notonectidae N	Highly	-25.6	14.5	-32.1	10.7
		(±1.3)	(±0.3)	(±0.6)	(±0.9)
Spencerhydrus sp.	Very highly	-26.1	9.8	-25.8	8.6
		(±0.4)	(±0.4)	(±1.1)	(±0.2)
Eretes sp.	Very highly	-25.2	12.0	-26.4	9.8
		(±0.3)	(±0.2)	(±0.1)	(±0.5)
Hyphydrus sp.	Very highly	-25.0	12.4	-32.7	8.9
		(±0.0)	(±0.0)	(±0.9)	(±0.5)

The significance of mobility for determining isotopic signatures can clearly be seen when comparing notonectid and corixid nymphs, which are non-flying residents, with the highly mobile adult stage (Table 7.10). As would be expected, the nymphs of both families had δ^{13}C and δ^{15}N isotopic signatures that reflected the differences in primary sources between natural wetlands and storages. In comparison, adults of both taxa collected from typical storages had similar isotopic values to those individuals collected from natural wetlands (Table 7.10). The beetles, *Spencerhydrus* sp. and *Eretes* sp., collected from typical storages had similar isotopic signatures to the same taxa collected from natural wetlands (Table 7.10). In contrast, *Hyphydrus* sp. had very different isotopic signatures depending on which waterbody they were collected from.

For fish the isotopic signatures of bone samples for *C. carpio* and *N. erebi* were different to those of muscle samples taken from the same individuals for samples from both waterbody types (Appendix 41 and 42). However, the δ^{13}C and δ^{15}N values of bone samples from typical storages were clearly different from the δ^{13}C and δ^{15}N values of both bone and tissue samples from natural wetlands (Appendix 41 and 42). There is no evidence to suggest that those individuals captured in typical storages had previously been feeding in natural wetlands.

7.5 Discussion

7.5.1 Stable isotope signatures

In most cases, the basal sources sampled in this study had similar δ^{13}C and δ^{15}N signatures to those measured in other waterbodies in Australia. The exceptions to this were samples of seston, CPOM and FPOM from typical storages which were ^{13}C-enriched (Bunn *et al.*, 2003; Medeiros, 2004; Hamilton *et al.*, 2005) (Appendix 41 and 42). δ^{13}C signatures of zooplankton were within the range collected at other sites in the catchment (Medeiros, 2004) and had δ^{15}N signatures within the range reported in other Australian studies (Bunn *et al.*, 2003; Medeiros, 2004). For the most part the δ^{13}C signatures of primary and secondary consumers were comparable to those measured at other Australian sites but were ^{15}N-enriched (Bunn and Boon, 1993; Bunn *et al.*, 2003; Medeiros, 2004). In general, δ^{13}C and δ^{15}N signatures for fish in both natural wetlands and typical storages were similar to those reported for other

Australian rivers (Bunn *et al.*, 2003) and other studies in the Border Rivers Catchment (Medeiros, 2004).

Considerable variation was detected in $\delta^{13}C$ and $\delta^{15}N$ values among sites and between waterbody types. Most food web components tended to be more ^{13}C and ^{15}N-enriched in typical storages than natural wetlands. These spatial differences are likely to reflect variations in the sources of nitrogen and carbon available to primary producers at the different waterbody types (Bunn and Boon, 1993; Medeiros, 2004). In other food web studies, CPOM has been reported as terrestrial in origin having similar mean $\delta^{13}C$ values to leaves collected from riparian vegetation, while FPOM is a combination of both terrestrially and aquatically derived material (Bunn *et al.*, 2003; Hamilton *et al.*, 2005; Leigh, 2008). The reduced amount of riparian vegetation at the typical storages could account for variation in stable isotope signatures of sources and the subsequent differences in consumer signatures between waterbody types. Such variation in source signatures has been previously documented; benthic POM was found to be ^{13}C-depleted near beaver lodges, compared to benthic POM from littoral regions without beaver activity, as a result of beaver selection for deciduous wood over coniferous (France, 2000).

7.5.2 Sources of carbon driving the food web in waterbodies of the Border Rivers Catchment

As algal sources have been identified as important sources of organic carbon for consumers in temperate and tropical rivers (e.g. Hamilton *et al.*, 1992; Bunn *et al.*, 2003; Delong and Thorp, 2006; Oliver and Merrick, 2006) and in wetland food webs (Hecky and Hesslein, 1995; France and Steedman, 1996; France, 1997), algae was expected to play a significant role in this study. Many food web studies have proposed that a combination of autochthonous algal carbon sources (e.g. benthic algae, phytoplankton) and to a lesser extent direct inputs from the riparian zone (e.g. particulate organic matter) are the major sources of carbon to the aquatic food webs of large rivers and wetlands (Thorp and Delong, 1994; Hecky and Hesslein, 1995; France and Steedman, 1996; France, 1997). The results of this study support this concept, especially if we assume that the main energy source for zooplankton was phytoplankton. Although there was substantial spatial and taxonomic variation in the importance of primary sources supporting consumers, algae in its various forms appeared to be a major contributor of organic carbon to aquatic food webs. This was true in both waterbody types but in particular storages (Table 7.4 and 7.6 and Fig. 7.6). In natural wetlands and typical storages, biofilm was more ^{13}C-enriched compared to the other primary sources. Many consumers were also enriched in ^{13}C suggesting algae were a major contributor to their diets. CPOM also ranked highly in importance to aquatic food webs in the Border Rivers Catchment which is in accordance with the original RPM (Thorp and Delong, 1994) and numerous other wetland food web studies (e.g. Hecky and Hesslein, 1995; France and Steedman, 1996; France, 1997; Herwig *et al.*, 2004).

Variation in the importance of different primary sources between sites and taxa is to be expected (Zeug and Winemiller, 2008). Studies supporting algae as the major contributor to aquatic food webs in large rivers have concentrated on main channel habitats rather than off channel areas such as floodplain wetlands (Thorp and Delong, 2002; Winemiller, 2005). Food webs vary both spatially and temporally and their structure can change in response to environmental drivers, species interactions, or a combination of these factors (Winemiller, 1996; Zeug and Winemiller, 2008). It is also possible that the absence of a major flood prior to the study had an effect on the sources of organic carbon available to the food web. A recent flood may have increased algal growth allowing this source to contribute more significantly to the food web (Medeiros, 2004). Flooding also increases connectivity between the river channel and floodplain wetlands allowing taxa access to potential source materials on the floodplain (Pringle, 2003; Winemiller, 2005).

It is very likely that not all the sources available to the food webs in natural wetlands and typical storages were collected during sampling (cf. Leigh, 2008). IsoSource was unable to compute solutions for a number of consumer taxa as their δ^{13}C values were either too high or low in relation to the primary sources. Mixing models can only provide solutions based on the input sources (Moore and Semmens, 2008). Zooplankton were included as a primary source in the mixing model as a proxy stable isotope signature for an as yet unmeasured component of the food web, e.g. phytoplankton (Shiel, 1995; Hadwen, 2002). However, there are likely to be other sources available at some or all sites that have not been accounted for in this way.

There may also be a problem with the timing of collection of sources and consumers. It is possible that the primary sources collected were not the same sources available to the consumers prior to sampling that have been assimilated into muscle or bone. The stable isotope signatures of consumers may reflect their diets a considerable amount of time before sampling was carried out for this study. Ben-David and Schell (2001) suggested that the 77 days of their controlled feeding study on captive mink was not long enough to exclude residual signatures from previous diets. However, the isotopic signatures of the sources in this study did explain a lot of the variation in consumer signatures, suggesting that in this case the consumers had assimilated the sources that were collected.

7.5.2.1 Variation in stable isotope signatures between life stages, individual size and different tissues

Not only were differences in stable isotope signatures apparent between taxa but between different life stages and sizes of the same taxa and between different tissues of the same individual. Prawns, *Macrobrachium* sp., were split into two size classes. Although they exhibited similar $\delta^{15}N$ values, larger individuals were enriched in ^{13}C compared to the small prawns. At all sites where both large and small prawns were collected, larger individuals were more reliant on biofilm compared to smaller individuals (see Table 7.6). This size related shift in dependence on algae may reflect the change in behaviour as they mature from pelagic larvae to benthic adults (Cook *et al.*, 2002; Nandlal and Pickering, 2005).

The size related changes in $\delta^{15}N$ values of *N. erebi* apparent in this study have been recorded elsewhere (Bunn *et al.*, 2003). However, the positive relationship between $\delta^{13}C$ values and standard length witnessed by Bunn *et al*. (2003) was not evident here. Bunn *et al*. (2003) attributed these shifts to a decreased dependence on zooplankton and a greater dependence on benthic algae and terrestrial detritus as *N. erebi* grow, which has also been suggested by dietary studies on this species (Medeiros and Arthington, 2008). The isotopic signature of zooplankton was ^{15}N-enriched and ^{13}C-depleted at both waterbody types in the catchment in comparison to biofilm, which would support this theory. However, this change in feeding behaviour was not obvious from the percentage contribution of primary sources to consumers (see Appendix 43 and Table 7.6).

If individuals of fish collected from storages had only recently been pumped into the storage site, slow turnover tissue samples (bone) should reflect the isotopic signatures of the primary sources of where the fish had been pumped from e.g. natural wetlands. Whereas, fast turnover tissue samples (muscle) should have similar isotopic values to the food sources available in the storages (Bunn *et al.*, 2003). However, there was greater variation in isotopic signatures between bone and muscle samples in the natural wetlands than in the typical storages (Appendix 41 and 42). If anything, the opposite would be expected if bone samples were reflecting the individuals previous feeding grounds. The bone samples of fish captured in the storages should also be more ^{13}C and ^{15}N-depleted given the lower δ^{13}C and δ^{15}N values for primary sources in natural wetlands. However, it is unknown for how long fish collected had been living and feeding in the storages. Hesslein *et al*. (1993) also suggested that in predatory fish the differences observed between the stable isotope signatures of slow to fast turnover tissues may reflect differences in tissue composition (e.g. lipid concentrations) rather than changes in diet isotope signatures over time. Although not predatory, the differences between the isotopic signatures of bone and tissue samples of *C. carpio* and *N. erebi* observed in this study may be indicating something other than changes in feeding grounds.

7.5.3 Food web variability: natural wetlands versus typical storages

In contrast to our hypothesis, typical storages appeared to be displaying more variability in δ^{13}C and δ^{15}N values of consumers and sources than found in natural wetlands. This could reflect the need for consumers to rely on whatever carbon sources were available at storages. If each storage site has a limited selection of energy sources and these vary between sites, then consumers may be restricted to the sources that are available, in turn causing the isotope signatures of consumers to vary between sites. In contrast, at the natural wetlands, if there are a range of energy sources available, consumers may always select the food source of higher nutritional value to them and thus their stable isotope signatures will be similar between sites.

7.5.4 Isotopic signatures of mobile taxa

It is possible that mobile macroinvertebrate taxa collected at typical storages may actually have fed at nearby natural wetlands and have isotopic signatures reflecting the primary sources found therein. The results presented in this study would appear to support this theory. The more mobile taxa had similar isotope signatures at each waterbody type compared with resident taxa, and mobile and non-mobile life stages in taxa had substantially different signatures.

Beetles are known to fly large distances, for instance, dytiscids have been reported to disperse over distances of several kilometres (Schafer *et al.*, 2006). In this study, isotopic signatures of *Spencerhydrus* sp. and *Eretes* sp in typical storages suggested that, although collected from typical storages, they may have been feeding in natural wetlands. In comparison, the beetle, *Hyphydrus* sp., appeared to be limited to feeding in the waterbody from where it was collected. Although nepids are generally considered poor fliers (Fernando, 1963; Lansbury, 1972; MDFRC, 2006), the data from the present study would suggest that irrespective of where they were collected, nepids were feeding only in natural wetlands. The same was true for adults of both Corixidae and Notonectidae but not for the nymphs. Variation in isotopic signatures between adults and nymphs is most likely related to the mobility of the two life stages. The adult stages of both taxa are strong fliers (Williams, 1980) allowing them to feed at other locations remote from where they were collected. Meanwhile, nymphs are fully aquatic and would only have access to primary sources within the waterhole where they were found.

The results from this study appear to support the theory that particular mobile macroinvertebrate taxa are feeding in natural wetlands irrespective of where they are collected. However, it could also be the case that these mobile taxa may have only recently moved into the storages and begun feeding, not providing sufficient time for their new ‘storage diet’ to exclude the residual signatures from their old ‘natural wetland diet’.

7.5.5 Conclusions

The results presented in this study are in agreement with the revised RPM (Thorp and Delong, 2002), namely that autochthonous organic carbon, in particular algae, is the major source of energy fuelling food webs of floodplain wetlands in the Border Rivers Catchment. The increased reliance on algae in typical storages in comparison to natural wetlands may reflect the lack of alternative primary sources available at these waterbodies. A strategy to improve aquatic biodiversity on irrigation properties may be to increase the diversity of food sources in storages for example by improving riparian vegetation or adding hard substrates to the storage bank (e.g. logs) to increase algal biofilm growth (Benke *et al.*, 1984; McGregor *et al.*, 2006).

A potential management problem for storages, especially those with tailwater, will be contamination by herbicides. The presence of herbicides in storages will undoubtedly inhibit the growth of algae and in turn reduce their contribution to energy production (Johnson, 1986; Okamura *et al.*, 2002).

Diversion of water, and therefore aquatic taxa, into storage systems and away from natural wetlands is likely to have a more detrimental effect on certain taxa than others. The results of this study suggest that mobile macroinvertebrate taxa may still be able to gain access to primary sources at nearby natural wetlands. If sources of organic carbon are limited at storages, resident macroinvertebrate taxa or fish which are pumped into storages are more likely to be adversely affected than mobile taxa.

It is important to note that, given the assumptions associated with stable isotope analysis and mixing models and the variability of the results obtained during this study, there is a need for further investigation in this area. The structure of food webs can change over space and time for a number of reasons (Zeug and Winemiller, 2008). To capture any temporal variation or changes between wet and dry years, food web sampling over the course of at least a year would be required.

Chapter 8 General discussion

8.1 Background

Water resource development throughout the world is a major threat to natural floodplain wetlands (OECD, 1996; Kingsford, 2000; Tockner and Stanford, 2002; Tockner *et al.*, 2008) and in turn aquatic biodiversity at the landscape scale. The production of cotton and other irrigated crops in the Border Rivers Catchment, Australia has contributed to the alteration of flow regimes (Southwell, 2002; Thoms *et al.*, 2005) and contraction or complete loss of many natural wetlands in the region (DWR, 1995; Kingsford, 1999). In light of this many texts have highlighted the possible potential of on-farm wetlands as alternative aquatic habitat (Bowmer *et al.*, 1994; Brooks, 1995; Linton and Goulder, 2000; Hazell *et al.*, 2001; Knutson *et al.*, 2004). On-farm water storages are common place in irrigation areas of Australia, including the Border Rivers Catchment. These structures can be of enormous size and, along with the associated levee systems, can significantly alter the flow regime downstream by diverting and storing floodplain flows. However, despite their presence on the floodplain for almost 50 years, there has been little investigation into the ecological value of these systems. This study aimed to address this issue by increasing our understanding of the sources, patterns and functions of aquatic assemblages associated with storages and the food web processes within these artificial wetlands.

In this study, several factors were identified as being potentially important in understanding the biodiversity value of storages and in determining:

1) How the structure and function of aquatic assemblages within storages compared with natural wetlands and
2) The ultimate sources of energy supporting the aquatic food web of storages compared to natural wetlands.

Factors thought to be influencing the aquatic assemblages within storages were 1) the morphological characteristics i.e. size, shape and habitat characteristics, 2) the hydrology of storages i.e. presence of tailwater and temporal variation in water levels, especially during the irrigation season 3) the access storages have to colonisation methods of aquatic taxa such as those present in the water column or emerging from the egg bank and 4) the connectivity of storages to the floodplain and river channel. The main factor expected to be affecting the ultimate sources of energy supporting the aquatic food web of storages was the diversity of primary sources available to aquatic consumers within storages and this in turn could be affected by differences in fringing vegetation and water quality (especially nutrients and turbidity).

8.2 Conceptual model

8.2.1 Morphological characteristics

On-farm water storages are generally man made structures, constructed to overcome the problem of an unreliable water supply, allowing water to be stored for use at a later date. Given this, it is fair to assume that the majority of storages will be built in the most cost effective and timely manner as possible, creating similar structures across the floodplain. In comparison natural wetlands have formed over thousands of years in response to changes in hydrology and soil deposition, taking on their own shape and associated biota.

Physical and hydrological characteristics of natural waterholes have been shown to influence the diversity and abundance of aquatic assemblages which they support (Sheldon and Walker, 1998; Davis *et al.*, 2002; Jenkins and Boulton, 2003; Marshall *et al.*, 2006). As shown in Chapter 3, storages and natural wetlands differed considerably with respect to their morphology, such as depth, surface area, volume and perimeter, and these were some of the major factors hypothesised to be affecting aquatic assemblages and processes in floodplain waterbodies (see section 1.4.1). In general storages were large, deep structures and more regular in shape while natural wetlands were shallower and irregular with large perimeters. A large number of storages surveyed for this study could be placed in one group which shared similar morphological and hydrological characteristics. This group was identified as the ‘typical storage’ group. The presence of numerous uniform storages on the floodplain and loss of natural wetlands is likely to lead to a lack of wetland diversity across the landscape along with reduced habitat complexity within wetland sites. Both factors will result in reduced aquatic biodiversity in the catchment (Olden *et al.*, 2001; Bunn and Arthington, 2002; Thoms *et al.*, 2006).

8.2.2 Aquatic assemblages and water chemistry

The biodiversity of aquatic habitats are strongly influenced by shoreline and channel complexity and their morphology (Jackson and Harvey, 1993; Olden *et al.*, 2001; Bunn and Arthington, 2002; Thoms *et al.*, 2006). In general increased complexity, surface area, volume and perimeter increases habitat diversity (Jackson *et al.*, 2001) which in turn increases the abundance and diversity of fish and macroinvertebrate populations (O'Connor, 1991; Bunn and Arthington, 2002). It was proposed that the lack of variability between storage sites and the fact that they were fundamentally different to natural waterbodies would lead to reduced aquatic biodiversity in the region if these structures were to become the dominant form of aquatic habitat on the floodplain. In general this was the case and natural wetlands were more diverse and productive than storages. However, this theory was not supported by the data collected from fish sampling.

Fish assemblages in storages had a similar diversity and were more abundant (CPUE) than those found in natural wetlands. One species, namely *N. erebi*, tended to dominate the abundances compared with other species at storage sites, while the contribution of exotic fish species was greatly reduced in storages in comparison to natural wetlands. In comparison, fish assemblages of natural wetlands had a much more even distribution of species. The domination of one or two fish species at storages may reflect the ability of these species to survive the extraction process or successfully breed once in the irrigation system. Uneven distribution of assemblages is often a feature of disturbed sites and in such cases ‘weedy’ species are typically the ones that dominate (Resh and Jackson, 1993). The high fecundity (Puckridge and Walker, 1990) and positively skewed distribution of *N. erebi* in storages makes it likely that a small number of mature adults were breeding in the storages. The widespread distribution of species like *N. erebi* and *L. unicolor* (Pusey *et al.*, 2004; Bostock *et al.*, 2006) and their limited genetic diversity (Bostock *et al.*, 2006; Hughes and Hillyer, 2006) indicate that they have high dispersal capabilities due to a tolerance of a wide range of environmental conditions (Pusey *et al.*, 2004; Bostock *et al.*, 2006). These traits could also explain the ability of *N. erebi* and, to a lesser extent, *L. unicolor* to dominate fish populations of storages.

Over all sites, a diverse fauna of macroinvertebrates was found representing 109 taxa, dominated by low mobility taxa and Insecta. Zooplankton assemblages consisted of 28 taxa dominated by rotifers. In accordance with the original conceptual model (see Section 1.4), both macroinvertebrate and zooplankton assemblages in typical storages were less diverse than natural wetlands and had lower densities. However, in the case of macroinvertebrates these reduced abundances were only apparent during the spatial study. Unfortunately none of the measured morphology or habitat variables were strongly associated with macroinvertebrate or zooplankton assemblages.

Although during the spatial study there was no significant difference in water quality between the two waterbody types, during the temporal sampling programme pH, salinity and conductivity were all significantly higher in typical storages in comparison with natural wetlands. Increased pH could increase the toxicity of any pollutants present in the storages (DIPNR, 2003), while increased salinity and conductivity indicates an increase in the amount of dissolved salts in the water which can be toxic to aquatic animals (Hart *et al.*, 1991; Nielsen *et al.*, 2003; Velasco *et al.*, 2006; Kefford *et al.*, 2007). These differences in water quality could account for the spatial variation in aquatic assemblages between natural wetlands and typical storages. A number of macroinvertebrate orders were under-represented in storages compared with natural wetlands, in particular Odonata, Coleoptera and Hemiptera which could reflect their relative mobility. If faced with unfavourable conditions in storages, individuals from these orders have the ability to relocate to nearby natural wetlands.

In general, natural wetlands became less diverse and increasingly similar to storages over time as water levels dropped. It is likely that the low water levels created unfavourable conditions in both storages and natural wetlands, causing only tolerant generalists to remain and the assemblages to converge; similar patterns have been observed in other wetland systems (Sheldon *et al.*, 2002). Temporal variations in water levels are often followed by fluctuations in water quality parameters (Osbourne *et al.*, 1987; Medeiros, 2004). In this case, conductivity, salinity and turbidity were highest when water levels were at their lowest. As stated, increased conductivity and salinity levels are likely to have a detrimental effect on aquatic animals (Hart *et al.*, 1991; Nielsen *et al.*, 2003; Velasco *et al.*, 2006; Kefford *et al.*, 2007). The increased suspended solids in the water column associated with higher turbidity can interfere with the feeding, breathing and reproduction of certain organisms and reduce the amount of light reaching aquatic plants and benthic algae (Arthington, 1995; Wylie, 1995; DIPNR, 2003). There was also a temporal shift in macroinvertebrate assemblages from being dominated by low and zero mobility taxa to high mobility taxa, suggesting that these highly mobile taxa were also taxa that could survive the conditions created by falling water levels.

The expected rise in water temperatures at storages due to the absence of riparian vegetation was not apparent from the results of this study. However, the low water levels experienced throughout the study, along with their shallower depths and smaller areas, may have caused natural wetlands to heat up more quickly than storages and overridden any benefits shading from riparian vegetation provides. Sampling during a wet year may yield different results.

8.2.3 Source of water

A second, but much smaller, group of storages was identified that had the same morphological characteristics as typical storages but did not contain recycled tailwater; ‘no tailwater’ storages. It follows that these storages will contain lower concentrations of agricultural chemicals, which are likely to be having a negative effect on the aquatic assemblages therein, than typical storages.

However, contrary to my hypothesis, the presence of tailwater in typical storages did not appear to influence the diversity or abundance of the aquatic assemblages within. Typical storages and ‘no tailwater’ storages had similar fish and macroinvertebrate diversity. However, due to low water levels, only four ‘no tailwater’ sites were used as a comparison and fish were only sampled from three of these sites. Before dismissing the benefits of restricting tailwater to limited numbers of storages (Lutton, 2005; Kennedy and Jarman, 2006) a thorough investigation into ‘no tailwater’ storages should be undertaken.

8.2.4 Connectivity

During a flood aquatic biota can move freely between the river and natural wetlands via the floodplain (CEC, 2000; Arthington *et al.*, 2005; Balcombe *et al.*, 2007). In contrast, not only are storages separated from the floodplain by an embankment and levees but in general the only passage for water into the storage is through one or more pumps. As expected, when sampling extracted water a number of fish were injured or killed during the pumping process. Those fish that do survive the pumping process will also eventually die as the only outlets are onto irrigated fields (NMFS, 1994; Moffatt and Voller, 2002).

An alternative to improving the fish assemblages of storages is to prevent the extraction of fish from the river channel via irrigation pumps. Rather than trying to increase the diversity and abundance of fish in storages should we instead be improving these barriers to connectivity and also trying to prevent the passage of fish through the pumps. Alternatively it would be extremely beneficial if the pump stations could somehow be adapted so that exotic species were more susceptible to extraction rather than native species as found during this study.

8.2.5 Sources of colonisation in typical storages

In dryland areas of Australia, the survival of aquatic taxa depends on their ability to rapidly colonise newly inundated wetlands (Tronstad *et al.*, 2007). It follows that the aquatic diversity of temporary wetlands will depend somewhat on the access these waterbodies have to sources of colonisation.

The fish fauna of extracted water was similar to assemblages of storages, with a catch dominated by *N. erebi* and low contribution of exotic species. This was as expected as fish have only one method of colonising new habitats; by following the path of floodwaters or in this case extracted water. These results also suggested that the uneven fish assemblages present in storages may be a result of what was pumped into them rather than the increased ability of certain species over others, to survive or breed in storages.

In comparison less than a third of the number of macroinvertebrate taxa encountered in storages throughout the study were present in the pumped water. It is therefore likely that a high proportion of macroinvertebrate taxa found in storages, especially the vagile groups, such as Coleoptera, Hemiptera, Diptera, Ephemeroptera, and Odonata (Nicolet *et al.*, 2004; Schafer *et al.*, 2006; Tronstad *et al.*, 2007) are colonising these systems by means of aerial dispersal. Diverse assemblages of aquatic invertebrates are known to establish on inundated floodplains purely as a result of aerial colonisation and rapid growth rates (Tronstad *et al.*, 2007).

Although zooplankton assemblages of extracted water were similar to that found in storages, it is likely that zooplankton are colonising storages by means of both immigration via extracted water and emergence from the egg bank. The egg bank is unlikely to be the only source of zooplankton taxa present in storages, with only 16 taxa emerging from the sediment of typical storages compared to the 22 taxa collected from typical storages. As expected the egg banks of storages were less diverse than those of the nearby floodplain. However, in contrast to our hypothesis, the egg bank of floodplain sediment was less abundant than that from storages. If storages are permanently inundated this could lead to a build up in the number of resting stages in the egg bank as emergence is suppressed (Delorme, 1991; Nielsen *et al.*, 2000) while causing a reduction in biodiversity (Nielsen *et al.*, 2000). As historical data for water levels in storages are not available it is impossible to come to firm conclusions as to the causes of differences in egg banks of floodplain and storage sediment.

8.2.6 Aquatic food webs

Despite the large amounts of terrestrial carbon available in dryland floodplain systems (Medeiros, 2004), there has been growing support for the riverine productivity model (RPM) (Thorp and Delong, 2002), namely that algal production is the primary source of organic carbon fuelling aquatic food webs throughout the world (Hamilton *et al.*, 1992; Lewis Jr. *et al.*, 2001; Bunn *et al.*, 2003; Thorp *et al.*, 2006; Leigh, 2008). Although results varied greatly between consumers, sites and waterbody types, this study also suggested that algal carbon was the main source of energy to aquatic food webs in the Border Rivers Catchment, particularly if it is assumed that zooplankton provided a proxy stable isotope signature for another algal source, such as phytoplankton. In agreement with the original RPM (Thorp and Delong, 1994), CPOM was also another important source of organic carbon. The absence of a major flood event during the study period may have affected results by limiting the growth of algae leading us to underestimate its contribution to the aquatic food web and by limiting access to potential sources of organic carbon on the floodplain. It was also apparent that more than one source could be important to a single consumer, with results suggesting generalist feeding at all trophic levels. However, this phenomenon was more apparent at typical storages than natural wetlands, perhaps suggesting that the preferred food type was more likely to be available at natural wetlands.

Despite the spatial differences in isotopic signatures of food web components, the hypothesis that food webs of natural wetlands would be more variable was not supported by the data presented in Chapter 7. In fact the opposite appeared to be true, with typical storages displaying more variability in $\delta^{13}C$ and $\delta^{15}N$ values of consumers and sources than natural wetlands. This could reflect a limited number of primary sources within individual storage sites but variation in the types of primary sources available between different storage sites.

As predicted, the isotopic signatures of mobile taxa did suggest that despite being present in storages these individuals were reliant on natural wetlands as a source of organic carbon. It is therefore fair to say that primary sources in typical storages may not be meeting the energy requirements of aquatic taxa which will have a detrimental effect on organisms that can not feed elsewhere. It is also likely that if natural wetlands continue to decline in numbers and size, alternative feeding grounds will not be as accessible for those mobile taxa pumped into storages.

Food webs of typical storages appeared to have an increased reliance on algal carbon in comparison to natural wetlands which may reflect a lack of alternative primary sources in these waterbodies. This may have been a result of the lack of riparian vegetation on storages. The original RPM proposed that both algal carbon sources and direct inputs from the riparian zone (e.g. CPOM) are the major sources of organic carbon to aquatic food webs (Thorp and Delong, 1994). If CPOM is not readily available at storage sites an increased reliance on algal carbon is to be expected.

There was no evidence from this study to suggest that lack of hard substrate, such as logs and rocks, steeper banks or highly turbid water of storages limited the availability of benthic algae to the food web. This has been reported previously; in the Cooper Creek, a narrow band of algae along the littoral zone of highly turbid waterholes was found to be the major source of energy for aquatic consumers (Bunn *et al.*, 2003).

8.2.7 Revised conceptual model

In light of the work presented in this study and combining the information presented in the previous sections (8.2.1 to 8.2.5) it is possible to refine the original conceptual model (Fig. 8.1).

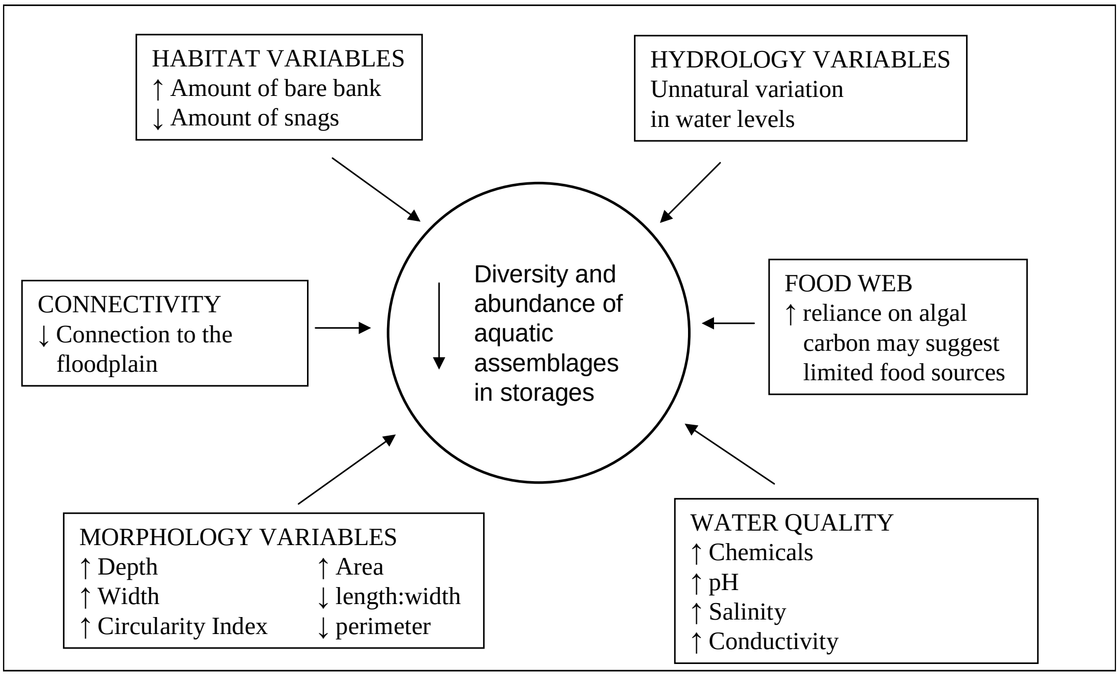


Fig. 8.1: Revised conceptual model comparing potential drivers affecting biodiversity in storages relative to natural wetlands.

8.3 Management implications and recommendations

The findings of the present study have important implications for the way that storages and floodplain wetlands are managed. Australian floodplain wetlands are sites of extraordinary biodiversity (Kingsford, 2000) and several studies have recognised the adverse effects water resource development can have on aquatic biota in these wetlands (Boulton and Lloyd, 1992; Sheldon and Walker, 1993; Gehrke *et al.*, 1995; Kingsford, 2000). Results presented in this thesis underline the importance of preserving natural wetlands and also suggest some options for improving the biodiversity value of storages.

Although storages did support a range of aquatic taxa, in general the aquatic assemblages of storages had reduced diversity and abundance compared to natural wetlands. It is hypothesised that the morphological homogeneity of storages and lack of habitat has contributed to preventing high aquatic biodiversity within these systems. However, the results of this study only detected one case of association between the environmental data and aquatic assemblage structure; with respect to morphology; depth, width and length:width explained 42% of the variation in fish species presence/absence patterns in storages. A more detailed examination of morphology and habitat characteristics will be essential to identify any stronger associations between waterhole characteristics and aquatic assemblages. Section 3.4.3 outlines options for increasing the diversity of habitat within storages and decreasing the uniformity of storages across the landscape by altering existing storages and improving the designs of new storages. It is believed that by providing additional habitat within storages we will increase the biodiversity of these systems. These range from major structural changes e.g. addition of sloping sides, shallow areas and central islands to relatively minor alterations such as planting aquatic vegetation and adding woody debris. Biological monitoring, before and after any alterations, will be vital if we are to realise the benefits of any such changes.

With respect to the impact of tailwater on biodiversity, the results of this study are somewhat inconclusive. Fish diversity was similar between all waterbody types and although ‘no tailwater’ storages had higher abundances of macroinvertebrates compared with typical storages they had similar diversity. However, it is still proposed that the presence of tailwater and associated chemicals will have an adverse effect on the aquatic communities of storages, decreasing the biodiversity and abundance of assemblages. Limiting the number of storages that receive recycled tailwater will help to overcome this problem. This, along with the commitment of the cotton industry to reduce the amount of chemicals used during a cotton season (APVMA, 1999; Deutscher *et al.*, 2005; Fitt, 2008), should reduce the impact chemicals are having on taxa, both within storages and nearby natural wetlands. Monitoring of storages, with and without tailwater, as well as long term chemical monitoring is needed to explore the effects of these changes and to improve upon the available data of chemicals in storage systems. This will also help increase our understanding about the effect the changing complex of pesticides associated with GM systems is having on on-farm wetlands.

A serious decline in fish numbers has been recorded in many irrigation districts around the world (Blackley, 2003). This knowledge along with the presence of fish and invertebrates in pumped water during this study suggest that the extraction of water for irrigation purposes will have a detrimental effect on the aquatic populations of river systems. Eight species of fish along with 22 and 25 species of macroinvertebrates and zooplankton respectively were collected from pumped water. Even if fish survive passage through the pumps, mortality is guaranteed as fish are trapped in storages or released to irrigation fields (NMFS, 1994; Moffatt and Voller, 2002). Fish screens are available to prevent entrainment into irrigation channels and should be installed by property holders. No such preventative measure has been developed for macroinvertebrates or zooplankton and neither, the full extent of the impact extraction has on these populations nor the knock on effect their removal has on riverine food webs, is known. This is an area for further research. Sampling of extracted water, along with investigation of the egg bank of storages, also gave us an insight into the sources of colonisation for storages. It is believed that aerial colonisation will also be a method used to colonise storages.

Altering the natural regime of flooding frequency and duration are known to affect the biodiversity and abundance of the egg bank (Delorme, 1991; Boulton and Lloyd, 1992; Nielsen *et al.*, 2000). The egg banks of floodplain sites were more diverse but less productive than those of typical storages. It is likely that these differences were a result of unnatural variation in water levels in the storages. The egg bank of storages that remain permanently inundated are likely to be less diverse but more abundant. However, further research, including long term monitoring of storage water levels, would be needed to confirm these links.

Stable isotope analysis of aquatic food webs in storages and natural wetlands suggested that, despite being present at storages, mobile taxa were feeding at nearby natural wetlands. If mobile taxa are seeking access to organic carbon in alternative wetlands, it is likely that primary sources available in storages are not meeting the energy requirements of aquatic taxa. It is proposed that planting aquatic vegetation and adding coarse woody debris to the banks of storages will not only increase habitat diversity but will provide additional food sources for aquatic assemblages and may improve the quality of primary sources at storage sites.

Given all of the evidence presented in this study and despite their widespread presence on the floodplain in irrigation areas, I believe that storages can not be thought of as replacements for natural wetlands. Increased water resource development and flow regulation leading to further losses of natural wetlands on the floodplain of the Border Rivers Catchment will only succeed in reducing the aquatic biodiversity of the region. It will therefore be essential to prevent any further decline in the current flooding regime across the catchment.

8.4 Future research

It should be noted that the sampling regime used in this study and the restrictions imposed by the drought provided a limited data set from which to draw conclusions about how storages compare with natural wetlands. Continued study of storages, both in the Border Rivers Catchment and other irrigation regions, is recommended to test the strength of these conclusions and applicability to other areas of Australia and worldwide. This will improve our knowledge about the assemblages of on-farm wetlands and assist in improving the biodiversity value of storages on irrigation properties.

Specifically future research should include:

1) A thorough examination of the morphology, habitat and hydrology of storages should be undertaken to include other variables not measured in this study such as relative values, area/depth ratios, cross sectional shape, bank slope and hydro-period. This would allow a more detailed comparison of storages and natural wetlands and may also identify a relationship between waterhole characteristics and aquatic assemblages.

2) A large scale project to monitor the effects of suggested changes for improving habitat and biodiversity (see Section 8.3). It would perhaps be best carried out in irrigation areas where storages are still being built and design changes can be implemented relatively easily. Long term biological monitoring of storages will be essential to help us understand any effects these changes have on aquatic assemblages. This would also provide an opportunity to investigate any effects the addition of alternative food sources has on aquatic food webs of storages.

3) A further comparison of aquatic assemblages between storages holding tailwater and those without. In addition, chemical levels in both types of storages should be tested to identify any relationships between chemicals and biodiversity.

4) A comparison of the aquatic populations of river channels between areas of high levels of water extraction and those with low levels of extraction.

5) A study of aerial colonisation of storages to provide the complete picture of how storages are colonised.

6) Long term monitoring of storage water levels and investigation into how different these are from natural water level fluctuations. This information will also provide an understanding of the impact of hydrology on assemblages and egg banks.

Appendices

Appendix 1: Example photos of storage sites. Photos: Susan Lutton.

Photo 1: Whynot Storage 1

Photo 2: South Callandoon Storage 2

Photo 3: Teriadi Storage 2

Appendix 2: Example photos of natural wetland sites. Photos: Susan Lutton.

Photo 4: Yambocully Lagoon

Photo 5: Barra Lagoon

Photo 6: Punbougal Lagoon

Appendix 3: Sample of questionnaire sent to property holders in the Border Rivers Catchment.

Questionnaire for Property Holders with on-farm water storages.

Could you please fill out a questionnaire for each of the storages on your property.

Name ____________________________ Property ____________________

Storage __________________________ Date _______________________

Q1. What is the area covered by the storage in hectares? ______________________

Q2. What is the depth of the storage or height of the embankment? ____________

Q3. What is the perimeter of the storage? __________________________________

Q4. Was the storage built on a natural wetland or a new site?

____

Q5. What is the source of water for the storage? Please tick.

a) Allocated (regulated) flows from releases from dams _______
b) Off allocation (unregulated flows) from natural drainage _______
c) Overland flow (floodplain harvesting) _______
d) Groundwater _______

Q6. If water is sourced from river flows, please name the river(s) the water is coming from and the distance of the storage from the river.

____

____

Q7. What is the maximum volume of water held in the storage? ______________

Q8. Are there any islands in the storage?___________________________________

Q9. In what year was the storage completed?______________________________

Q10. When was the storage last completely dry? How long was it dry for?

____

Q11. Are stock allowed access to the storage?

Q12. Has the storage ever been used for cropping when dry?

Q13. Have fish been artificially introduced into the storage? If so what type and how often is this done?

Q14. Is it possible that the storage may contain tailwater at some time during the cotton season?

Q15. Are herbicides used to prevent plant growth along the edges of the storage and on top of the embankment?

Q16. Has vegetation been planted along the edges of the storage? If so, what type?

Q17. Are there any natural lagoons or wetlands on your property that could be used for comparison? Name?

Thank you for taking the time to fill out this questionnaire. Please fax or post back your completed forms to the above address. If you have any queries about any of the questions please phone Susan Lutton on 07 4671 4650

Appendix 4: Hydrology data for all 99 storage sites used for classification of storages.

Site	Allocated Flow	Unregulated Flow	Overland Flow	Groundwater	Tailwater
1	Y	Y	Y		Y
2	Y	Y	Y		
3	Y	Y	Y		Y
4	Y	Y	Y		Y
5		Y	Y		Y
6		Y	Y		
7		Y	Y		Y
8		Y	Y		Y
9		Y	Y		Y
10	Y	Y	Y		Y
11	Y	Y	Y		Y
12	Y	Y	Y		Y
18		Y	Y		Y
19		Y	Y		
20	Y	Y			
21	Y	Y	Y		Y
22	Y	Y	Y		Y
23			Y	Y	Y
24	Y	Y	Y		Y
25	Y	Y	Y		Y
26	Y	Y	Y		Y
27	Y	Y	Y		Y
28	Y	Y	Y		Y
29	Y	Y			Y
30	Y	Y	Y		Y
31	Y	Y			Y
32		Y	Y		Y
33		Y	Y		Y
34	Y	Y	Y		Y
35	Y	Y	Y		Y
36	Y	Y	Y		Y
37	Y	Y			
38	Y	Y			Y
39	Y	Y			Y
40	Y	Y	Y		Y
41	Y	Y			Y
42	Y	Y			Y
43	Y	Y			Y
44	Y	Y	Y		Y
45	Y	Y	Y		Y
46	Y	Y	Y		Y
47	Y	Y	Y		Y
48	Y	Y	Y		Y
49	Y	Y	Y		Y
50	Y	Y	Y		Y
51	Y	Y	Y		Y
52	Y	Y	Y		Y
53	Y	Y	Y		
54		Y			Y
55	Y	Y			

Site	Allocated Flow	Unregulated Flow	Overland Flow	Groundwater	Tailwater
56	Y	Y			
57	Y	Y			Y
58	Y	Y			
59	Y	Y			Y
60	Y	Y	Y		Y
61	Y	Y	Y		Y
62	Y	Y	Y		Y
63	Y	Y	Y		Y
64	Y	Y	Y		Y
65	Y	Y	Y		Y
66			Y		Y
69	Y	Y	Y		Y
70	Y	Y	Y		Y
71	Y	Y	Y		Y
72	Y	Y	Y		Y
73	Y	Y	Y		Y
74	Y	Y	Y		Y
75	Y	Y			Y
76	Y	Y			Y
77		Y	Y		Y
78	Y	Y	Y		Y
79	Y	Y	Y		Y
80	Y	Y	Y		Y
81		Y	Y		Y
82		Y	Y		Y
83		Y	Y		
84	Y	Y	Y		Y
85	Y	Y	Y		Y
86	Y	Y	Y		Y
87	Y	Y	Y		Y
88	Y	Y	Y		Y
90	Y	Y	Y		Y
91	Y	Y	Y		Y
92	Y	Y	Y	Y	Y
93	Y	Y	Y	Y	Y
105	Y	Y	Y		Y
108	Y	Y	Y		Y
109	Y	Y	Y		
110	Y	Y	Y		Y
117	Y	Y			Y
118	Y	Y			Y
119		Y			
120		Y	Y		Y
121	Y	Y	Y		Y
122	Y	Y	Y		Y
123	Y	Y	Y		Y
124	Y	Y	Y		Y
125	Y	Y	Y		Y
126	Y	Y	Y		Y

Appendix 5: Morphology data for all 99 storage sites used for classification of storages and comparison with natural wetlands. *Two sites removed prior to analyses to compare with natural wetlands.

Storage Site	**Age**	**Height (m)**	**Area (m^2)**	**Perimeter (m)**	**Length (m)**	**Width (m)**	**Length:Width**	**Capacity (m^3)**	**Distance from source river (m)**	**Distance from Barwon River (m)**	**Circularity Index**	**Horton's Form Factor**	**Elongation Ratio**
1	19	5.0	658000	3300	1186	462	2.5671	3300000	2000	2000	0.75929035	554.8060708	0.7717619
2	17	4.3	295000	2200	752	419	1.794749	900000	3000	3000	0.76592548	392.287234	0.8149824
3	2	4.5	400000	2700	829	455	1.821978	1600000	1800	1800	0.68951279	482.509047	0.860856
4	5	4.0	50000	1100	349	209	1.669856	275000	2000	2000	0.51927151	143.2664756	0.7229606
5	4	5.0	449000	2700	775	585	1.324786	1500000	500	5571	0.77397811	579.3548387	0.9756106
6	4	5.0	535000	3200	1068	743	1.437416	2000000	1000	1529	0.65654378	500.9363296	0.7727886
7	28	4.0	110000	1460	521	207	2.516908	300000	3000	3000	0.64848038	211.1324376	0.7183129
8	17	4.0	334000	2330	720	452	1.59292	2000000	3000	3000	0.77311569	463.8888889	0.9057238
9	20	4.0	201000	1950	717	263	2.726236	2000000	3000	3000	0.66425785	280.334728	0.70556
10	5	4.0	450000	2500	807	515	1.56699	1600000	7000	7000	0.90477868	557.6208178	0.9379675
11	5	4.0	800000	3850	1217	727	1.674003	4500000	3000	3000	0.67823218	657.3541495	0.8292958
12	5	4.0	800000	3860	1431	602	2.377076	4500000	500	500	0.67472258	559.0496157	0.7052781
*18	22	3.5	150000	5000				300000	100	100			
19	12	6.5	50000	1010	333	255	1.305882	500000	1000	1000	0.61593817	150.1501502	0.7576975
20	23	4.0	150000	1260	320	316	1.012658	400000	6000	6000	1.18729881	468.75	1.3656855
21	8	5.0	220000	2000	762	281	2.711744	900000	8000	8000	0.69115038	288.7139108	0.6945627
22	24	6.0	200000	1840	469	457	1.026258	1200000	5000	5000	0.74234231	426.4392324	1.0759627
23	5	6.1	135000	2100	526	286	1.839161	650000	0	0	0.38468481	256.6539924	0.7881996
24	7	5.0	650000	3100	1051	516	2.036822	2555000	5000	9345	0.84996263	618.4586108	0.8655836
25	5	5.0	620000	2930	892	623	1.431782	2400000	2500	5680	0.90754112	695.0672646	0.9960613
26	3	5.0	510000	2760	807	582	1.386598	2200000	2500	5745	0.84132128	631.9702602	0.9985426
27	12	4.7	270000	2700	594	514	1.155642	1150000	250	7675	0.46542113	454.5454545	0.9870758
28	5	5.3	660000	3500	938	716	1.310056	3200000	250	4880	0.67704527	703.6247335	0.9772917
29	21	5.0	130000	1340	360	334	1.077844	500000	10000	10000	0.90979515	361.1111111	1.1301192
30	19	5.0	180000	1640	414	407	1.017199	550000	8000	8000	0.84099744	434.7826087	1.1563544

Storage Site	Age	Height (m)	Area (m^2)	Perimeter (m)	Length (m)	Width (m)	Length:Width	Capacity (m^3)	Distance from source river (m)	Distance from Barwon River (m)	Circularity Index	Horton's Form Factor	Elongation Ratio
31	2	5.5	200000	1920	648	313	2.070288	1000000	8000	8000	0.68176924	308.6419753	0.7787446
32	9	5.0	640000	3150	866	720	1.202778	1800000	7500	9954	0.81052932	739.0300231	1.0423826
33	8	5.0	1780000	6270	2030	900	2.255556	6800000	10000	10876	0.5689771	876.8472906	0.7415988
34	24	5.0	400000	2630	803	520	1.544231	1400000	1000	1000	0.72670535	498.132005	0.8887293
35	21	5.0	400000	2700	883	476	1.855042	1600000	1200	1200	0.68951279	453.0011325	0.8082102
36	8	4.5	400000	2700	829	499	1.661323	1900000	3700	3700	0.68951279	482.509047	0.860856
37	5	5.0	186600	1630	435	378	1.150794	1200000	7000	7000	0.88256418	428.9655172	1.1205252
38	12	5.0	768800	3300	878	822	1.068127	3000000	7000	7000	0.88714653	875.6264237	1.1268529
39	26	5.0	363700	2970	1255	282	4.450355	1500000	7000	7000	0.51813182	289.8007968	0.5422293
40	11	4.5	300000	1990	804	269	2.988848	1000000	9000	9000	0.95197373	373.1343284	0.7687049
41	18	5.0	280000	2120	721	427	1.688525	1000000	4000	4000	0.78288176	388.3495146	0.8281306
42	14	5.0	280000	1980	597	398	1.5	1000000	4000	4000	0.89750632	469.0117253	1.0001376
*43	25	2.0	300000	6000				600000	3000	3000			
44	20	5.0	480000	2960	883	587	1.504259	1500000	3000	3000	0.68844251	543.601359	0.88535
45	13	4.0	400000	2320	709	496	1.429435	1200000	6000	6000	0.93388604	564.1748942	1.006558
46	11	4.0	450000	2830	866	531	1.630885	1500000	6000	6000	0.70607284	519.630485	0.8740644
47	1	6.0	200000	2070	576	460	1.252174	1550000	6000	6000	0.58654207	347.2222222	0.8760877
48	7	5.0	240000	1670	610	259	2.355212	500000	3000	3000	1.08140448	393.442623	0.9062142
49	5	5.5	991000	4400	1875	524	3.578244	4500000	8000	8000	0.64324759	528.5333333	0.599088
50	4	6.0	401500	2500	693	545	1.27156	2500000	4000	4000	0.80726365	579.3650794	1.0317265
51	4	5.5	834500	3800	1169	735	1.590476	4500000	1000	1000	0.72622135	713.8579983	0.8817668
52	7	5.0	770000	3570	1264	686	1.842566	3400000	100	2176	0.75921391	609.1772152	0.7833455
53	2	3.0	400000	2400	916	324	2.82716	1000000	100	100	0.87266463	436.6812227	0.7790935
54	11	5.0	288000	2500	900	323	2.786378	1100000	40	6685	0.57905836	320	0.6728353
55	9	4.0	280000	2930	1096	259	4.23166	1200000	4500	4500	0.40985728	255.4744526	0.544783
56	20	4.0	240000	1920	489	478	1.023013	900000	4500	4500	0.81812309	490.797546	1.1304512
57	15	4.0	220000	2200	859	271	3.169742	800000	4500	4500	0.57119866	256.1117579	0.6161312
58	12	1.5	100000	1270	465	164	2.835366	150000	800	800	0.77911654	215.0537634	0.7673652

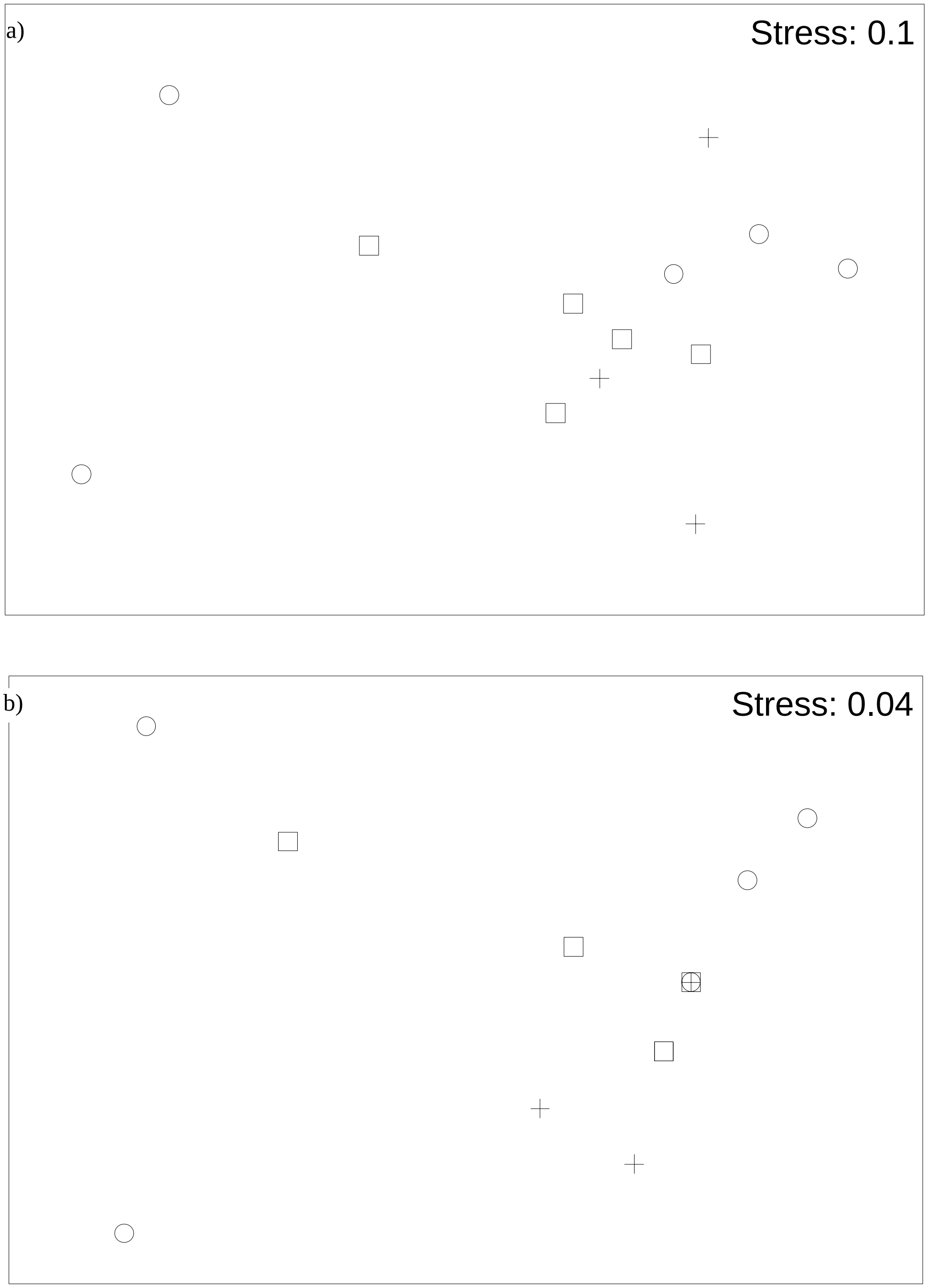
Storage Site	Age	Height (m)	Area (m2)	Perimeter (m)	Length (m)	Width (m)	Length:Width	Capacity (m3)	Distance from source river (m)	Distance from Barwon River (m)	Circularity Index	Horton's Form Factor	Elongation Ratio
59	15	3.0	150000	1750	738	159	4.641509	500000	0	0	0.6154957	203.2520325	0.5921672
60	24	3.0	200000	1460	955	791	1.207332	400000	100	100	1.17905523	209.4240838	0.5284047
61	18	5.0	800000	3440	489	257	1.902724	3110000	3000	3000	0.84953831	1635.99182	2.0639121
62	12	6.0	1200000	4370	1315	938	1.401919	4500000	2000	2000	0.78963836	912.5475285	0.9399828
63	14	5.0	650000	3250	1061	553	1.918626	2500000	3500	3500	0.77331511	612.6295947	0.8574254
64	11	5.5	350000	2400	760	445	1.707865	1250000	3600	3600	0.76358155	460.5263158	0.8783659
65	7	3.0	305000	2260	866	276	3.137681	1000000	2500	2500	0.7504	352.1939954	0.7195933
66	6	3.5	620000	4500	863	618	1.39644	2000000	0	0	0.38474814	718.424102	1.0295326
69	20	4.0	550000	3110	872	689	1.265602	1900000	9000	37103	0.71458151	630.733945	0.9596656
70	13	5.0	750000	3530	1028	784	1.311224	3300000	9000	35286	0.75634809	729.5719844	0.9505885
71	4	7.0	550000	3040	882	638	1.382445	3800000	9000	37932	0.74786875	623.5827664	0.948785
72	16	5.0	1350000	4650	1591	783	2.031928	5000000	100	100	0.78458089	848.5229415	0.8240466
73	13	5.0	600000	3090	1004	600	1.673333	2400000	1500	1500	0.7896673	597.6095618	0.8705565
74	6	7.0	600000	3020	1071	448	2.390625	3600000	300	300	0.82669865	560.2240896	0.8160959
75	20	5.0	170000	1680	563	286	1.968531	750000	10000	10000	0.75690299	301.9538188	0.8263635
76	20	5.0	200000	1810	563	352	1.599432	750000	10000	10000	0.76715428	355.2397869	0.8963171
77	4	8.0	1210000	4400	1367	819	1.669109	10000000	200	23500	0.78539816	885.1499634	0.9079862
78	14	5.0	270000	2060	1055	252	4.186508	1500000	0	0	0.79953814	255.9241706	0.5557564
79	7	5.0	510000	2910	837	643	1.301711	2500000	4500	2895	0.75682255	609.3189964	0.9627526
80	4	5.0	1010000	3590	1172	938	1.249467	4500000	8000	5751	0.98478708	861.774744	0.9675828
81	2	6.0	390000	2740	967	416	2.324519	1800000	2000	7000	0.65278978	403.3092037	0.7287203
82	12	6.0	980000	4630	1528	676	2.260355	3500000	1000	5731	0.57447874	641.3612565	0.7310461
83	2	5.0	520000	3300	1286	379	3.39314	2000000	0	4816	0.60004708	404.3545879	0.6327261
84	1	6.0	810000	4640	1504	536	2.80597	4000000	2000	2000	0.47277981	538.5638298	0.6752269
85	2	6.0	940000	4530	1759	569	3.091388	4800000	2000	2000	0.57562721	534.3945424	0.6219467
86	12	5.3	300000	2450	882	333	2.648649	1200000	200	18261	0.62805684	340.1360544	0.7007242
87	8	4.0	200000	1510	435	355	1.225352	600000	2000	17446	1.10226487	459.7701149	1.1600609
88	3	5.5	300000	2700	965	400	2.4125	2000000	3000	17262	0.51713459	310.880829	0.6404546

Storage Site	Age	Height (m)	Area (m^2)	Perimeter (m)	Length (m)	Width (m)	Length:Width	Capacity (m^3)	Distance from source river (m)	Distance from Barwon River (m)	Circularity Index	Horton's Form Factor	Elongation Ratio
90	17	5.0	954600	3680	1382	597	2.314908	3300000	500	14499	0.88579996	690.7380608	0.7977333
91	7	5.0	515700	2930	1021	489	2.087935	1850000	500	16859	0.75486929	505.093046	0.7936479
92	11	5.0	170000	1600	549	297	1.848485	700000	20	20	0.83448555	309.6539162	0.8474365
93	6	5.0	420000	2300	699	477	1.465409	1400000	1000	1000	0.99770806	600.8583691	1.0461706
105	10	5.0	500000	2860	1285	268	4.794776	1540000	3000	3000	0.76815313	389.1050584	0.6209218
108	13	5.0	330000	2550	675	586	1.151877	1600000	5600	9648	0.63773969	488.8888889	0.9603029
109	8	5.0	310000	2360	668	521	1.28215	1470000	200	9648	0.69943531	464.0718563	0.9405014
110	7	8.0	520000	2900	923	925	0.997838	3512000	200	1432	0.77699319	563.3802817	0.8815664
117	15	5.5	1020000	4100	1212	821	1.476248	3500000	500	11814	0.76250434	841.5841584	0.9402699
118	13	5.5	400000	2400	801	477	1.679245	2200000	400	11723	0.87266463	499.3757803	0.8909484
119	13	3.8	400000	3100	890	579	1.537133	1000000	500	1321	0.52305393	449.4382022	0.8018535
120	8	3.0	350000	2990	1033	568	1.818662	700000	3000	2788	0.4919665	338.8189739	0.6462324
121	14	4.0	340000	2400	816	560	1.457143	1500000	200	4005	0.74176493	416.6666667	0.8063143
122	14	4.0	210000	2360	1016	207	4.908213	800000	200	3343	0.47381101	206.6929134	0.5089452
123	12	4.0	410000	2600	969	406	2.3867	1100000	2200	5453	0.76216153	423.1166151	0.7456297
124	8	4.0	280000	2120	746	361	2.066482	1600000	1500	6939	0.78288176	375.3351206	0.8003782
125	11	4.4	530000	3020	905	582	1.554983	2500000	200	3858	0.73025047	585.6353591	0.9077043
126	2	4.0	230000	1960	737	297	2.481481	1200000	2400	2400	0.75235976	312.0759837	0.7342627

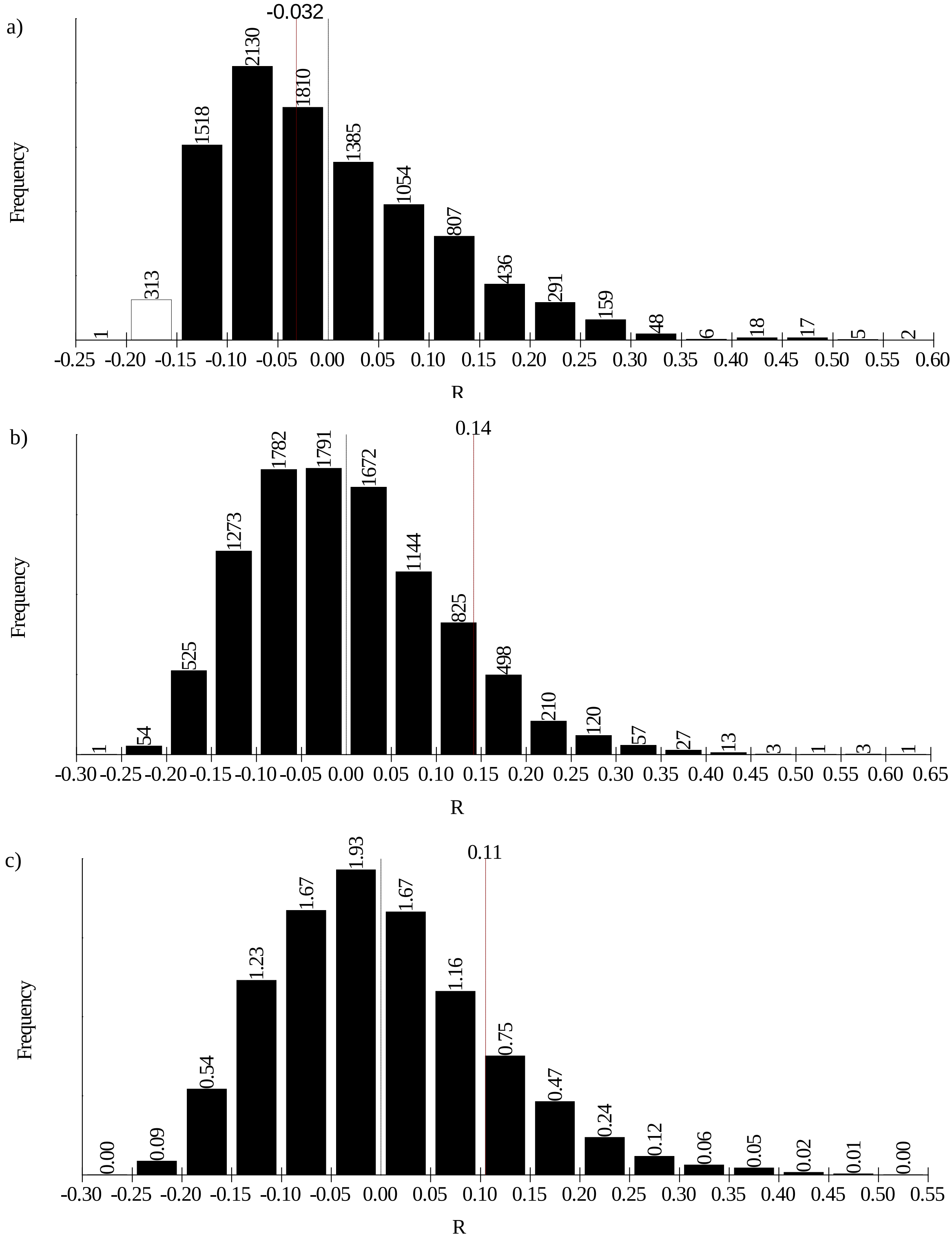
Appendix 6: Actual numbers of fish caught from pilot sampling of seven fyke nets at a typical storage site. Nets set on 14th May 2005 and retrieved on 15th May 2005.

	Fyke 1	Fyke 2	Fyke 3	Fyke 4	Fyke 5	Fyke 6	Fyke 7
C. carpio	1	1	8	2	3	0	9
N. erebi	54	255	74	319	118	107	24
L. unicolor	20	22	19	10	4	50	39
M. ambigua	18	21	18	12	6	8	22

Appendix 7: Non-metric multi dimensional scaling plot for fish assemblage structure based on a) non-standardised $Log_{10}(CPUE+1)$ data (absolute abundance) and b) species presence/absence data.



Appendix 8: Histograms for ANOSIM results of fish data for a) presence/absence (R = -0.032), b) absolute abundance $Log_{10}(CPUE+1)$ (R = 0.142) and c) relative abundance $Log_{10}(CPUE+1)$ (R =-0.105).



Appendix 9: Standard lengths (mm) of different fish species across the three different waterbody types.

	Fish Species								
Natural	*N. erebi*	*Hypseleotris spp*	*M. fluviatilis*	*R. semoni*	*L. unicolor*	*C. carpio*	*M. ambigua*	*C. auratus*	*T. tandanus*
No. of individuals	23	27	1	1	10	41	13	12	1
Min	28	13	48	55	9	109	68	88	168
Max	295	31	48	55	128	388	315	260	168
Mean	134.91	19.37	48.00	55.00	83.80	182.20	152.62	121.42	168.00
s.e.	16.85	1.01			12.26	9.93	21.88	13.30	
No tailwater									
No. of individuals	161				6	15	28	1	
Min	27				108	151	54	129	
Max	161				143	290	179	129	
Mean	64.71				125.17	180.93	110.46	129.00	
s.e.	2.17				4.94	11.17	6.88		
Typical									
No. of individuals	1179	1	2	6	80	17	68	7	
Min	6	25	29	49	39	64	30	48	
Max	156	25	32	57	183	374	287	125	
Mean	52.26	25.00	30.50	52.67	92.10	203.53	67.74	96.29	
s.e.	0.54		1.50	1.12	3.45	17.18	4.99	10.72	

Appendix 10: Water quality data (±SE) for the natural wetland (NW), storages without tailwater (TW) and typical storage (TS) sites in the Border Rivers Catchment. Data correspond to the mean of the three replicates taken at each individual site on one sampling occasion. Dissolved oxygen data were not available for a number of sites (na= not available). 1=August-October 2005, 2 = November 2005, 3 = February 2006, 4 = March 2006, 5 = December 2006, 6 = January 2007, 7 = February 2007.

Time	Site	pH	Conductivity (mS cm^{-1})	Turbidity (NTU)	Dissolved oxygen (mg L^{-1})	Water temperature (^{o}C)	Salinity (%)
1	NW1	7.73	0.115	439	11.17	13.8	0.00
		(±0.05)	(±0.007)	(±18)	(±0.74)	(±0.3)	(0)
1	NW2	8.08	0.528	59	na	23.8	0.02
		(±0.04)	(±0.001)	(±1)		(±0.2)	(0)
1	NW3	7.45	0.404	237	na	22.6	0.01
		(±0.04)	(±0.001)	(±50)		(±0.1)	(0)
1	NW4	7.60	0.116	221	na	20.2	0.00
		(±0.08)	(±0.006)	(±2)		(±0.1)	(0)
1	NW5	8.49	0.422	16	13.15	16.8	0.01
		(±0.14)	(±0.000)	(±6)	(±0.17)	(±0.4)	(0)
1	NW6	8.25	0.259	107	10.79	20.0	0.01
		(±0.07)	(±0.000)	(±16)	(±0.24)	(±0.3)	(0)
1	NW7	7.67	0.105	261	na	20.3	0.00
		(±0.05)	(±0.002)	(±6)		(±0.5)	(0)
1	NW8	8.22	0.193	441	10.99	17.5	0.00
		(±0.02)	(±0.003)	(±8)	(±0.03)	(±0.5)	(0)
1	TS1	8.12	0.194	692	11.61	14.8	0.00
		(±0.10)	(±0.001)	(±13)	(±0.16)	(±0.0)	(0)
1	TS2	7.42	0.349	393	12.75	15.3	0.01
		(±0.08)	(±0.012)	(±24)	(±0.08)	(±0.5)	(0)
1	TS3	8.16	0.233	567	11.57	17.3	0.00
		(±0.02)	(±0.002)	(±193)	(±0.01)	(±0.7)	(0)
1	TS4	7.79	0.249	271	na	26.3	0.00
		(±0.14)	(±0.004)	(±4)		(±0.9)	(0)
1	TS5	7.96	0.276	224	10.63	17.7	0.01
		(±0.08)	(±0.001)	(±4)	(±0.03)	(±0.6)	(0)
1	TW1	9.15	0.547	108	na	22.3	0.02
		(±0.05)	(±0.002)	(±17)		(±0.3)	(0)
1	TW2	8.29	0.300	713	na	20.8	0.01
		(±0.01)	(±0.003)	(±144)		(±0.3)	(0)
1	TW3	7.91	0.199	180	na	24.1	0.00
		(±0.09)	(±0.009)	(±23)		(±0.3)	(0)
1	TW4	8.10	0.155	339	na	22.3	0.00
		(±0.03)	(±0.004)	(±4)		(±0.6)	(0)
2	NW7	7.28	0.137	451	na	23.7	0.00
		(±0.08)	(±0.001)	(±23)		(±0.1)	(0)
2	TS7	7.93	0.205	397	na	25.7	0.00
		(±0.11)	(±0.001)	(±4)		(±0.1)	(0)
2	TS8	8.10	0.310	222	na	24.3	0.01
		(±0.07)	(±0.046)	(±3)		(±0.1)	(0)
3	NW1	7.87	0.291	876	na	30.7	0.01
		(±0.02)	(±0.001)	(±63)		(±0.7)	(0)
3	NW4	7.60	0.133	296	na	32.0	0.00
		(±0.03)	(±0.010)	(±9)		(±0.4)	(0)

Time	Site	pH	Conductivity (mS cm^{-1})	Turbidity (NTU)	Dissolved oxygen (mg L^{-1})	Water temperature (^{o}C)	Salinity (%)
3	NW7	7.66	0.158	580	na	29.2	0.00
		(±0.01)	(±0.001)	(±1)		(±0.0)	(0)
3	NW9	8.84	0.352	165	na	32.2	0.01
		(±0.02)	(±0.000)	(±1)		(±0.1)	(0)
3	NW10	7.75	0.241	999	na	31.3	0.00
		(±0.10)	(±0.004)	(±0)		(±0.5)	(0)
3	TS1	8.34	0.267	857	na	28.1	0.01
		(±0.04)	(±0.002)	(±95)		(±0.2)	(0)
3	TS6	8.33	0.323	222	na	30.2	0.01
		(±0.09)	(±0.001)	(±3)		(±0.4)	(0)
3	TS7	8.60	0.286	356	na	27.4	0.01
		(±0.25)	(±0.018)	(±4)		(±0.3)	(0)
3	TS8	8.70	0.336	269	na	27.5	0.01
		(±0.01)	(±0.003)	(±116)		(±0.3)	(0)
4	NW1	7.55	0.190	490	na	24.1	0.00
		(±0.04)	(±0.007)	(±29)		(±0.2)	(0)
4	NW4	7.59	0.131	540	na	25.1	0.00
		(±0.03)	(±0.003)	(±10)		(±0.1)	(0)
4	NW7	7.65	0.215	336	na	26.8	0.00
		(±0.01)	(±0.001)	(±16)		(±0.1)	(0)
4	NW9	7.81	0.234	389	na	25.0	0.00
		(±0.04)	(±0.000)	(±6)		(±0.3)	(0)
4	NW10	7.26	0.102	738	na	25.3	0.00
		(±0.05)	(±0.001)	(±56)		(±0.3)	(0)
4	TS1	8.10	0.291	848	na	28.3	0.01
		(±0.09)	(±0.001)	(±21)		(±1.3)	(0)
4	TS6	7.97	0.319	240	na	24.2	0.01
		(±0.02)	(±0.006)	(±19)		(±0.1)	(0)
4	TS7	8.13	0.256	396	na	27.3	0.00
		(±0.01)	(±0.001)	(±36)		(±0.4)	(0)
4	TS8	8.54	0.370	497	na	23.6	0.01
		(±0.04)	(±0.001)	(±17)		(±0.0)	(0)
4	TS9	8.56	0.291	435	na	26.3	0.01
		(±0.04)	(±0.000)	(±13)		(±0.2)	(0)
6	NW1	7.87	0.368	1590	5.90	31.4	0.00
		(±0.11)	(±0.007)	(±98)	(±0.17)	(±2.2)	(0)
6	NW4	8.17	0.335	1567	6.62	29.9	0.01
		(±0.14)	(±0.002)	(±284)	(±0.24)	(±1.5)	(0)
6	TS1	8.80	0.714	490	8.73	29.0	0.03
		(±0.02)	(±0.003)	(±168)	(±0.28)	(±0.1)	(0)
6	TS6	9.53	0.554	834	7.32	24.9	0.02
		(±0.01)	(±0.000)	(±12)	(±0.18)	(±0.4)	(0)
6	TS7	9.88	1.187	184	11.78	30.7	0.05
		(±0.04)	(±0.003)	(±7)	(±0.55)	(±0.6)	(0)
6	TS9	8.81	0.353	227	9.77	30.6	0.01
		(±0.05)	(±0.002)	(±24)	(±0.04)	(±0.1)	(0)
7	NW1	8.27	0.493	1556	6.41	34.2	0.02
		(±0.02)	(±0.003)	(±79)	(±0.38)	(±1.0)	(0)

Time	**Site**	**pH**	**Conductivity (mS cm^{-1})**	**Turbidity (NTU)**	**Dissolved oxygen (mg L^{-1})**	**Water temperature (^{o}C)**	**Salinity (%)**
7	NW4	8.73	0.472	266	8.88	33.8	0.01
		(±0.05)	(±0.002)	(±34)	(±0.42)	(±0.5)	(0)
7	TS6	8.82	1.453	698	8.63	28.4	0.06
		(±0.02)	(±0.009)	(±41)	(±0.19)	(±0.4)	(0)
7	TS9	8.29	0.365	218	6.58	25.3	0.01
		(±0.07)	(±0.006)	(±8)	(±0.09)	(±0.1)	(0)

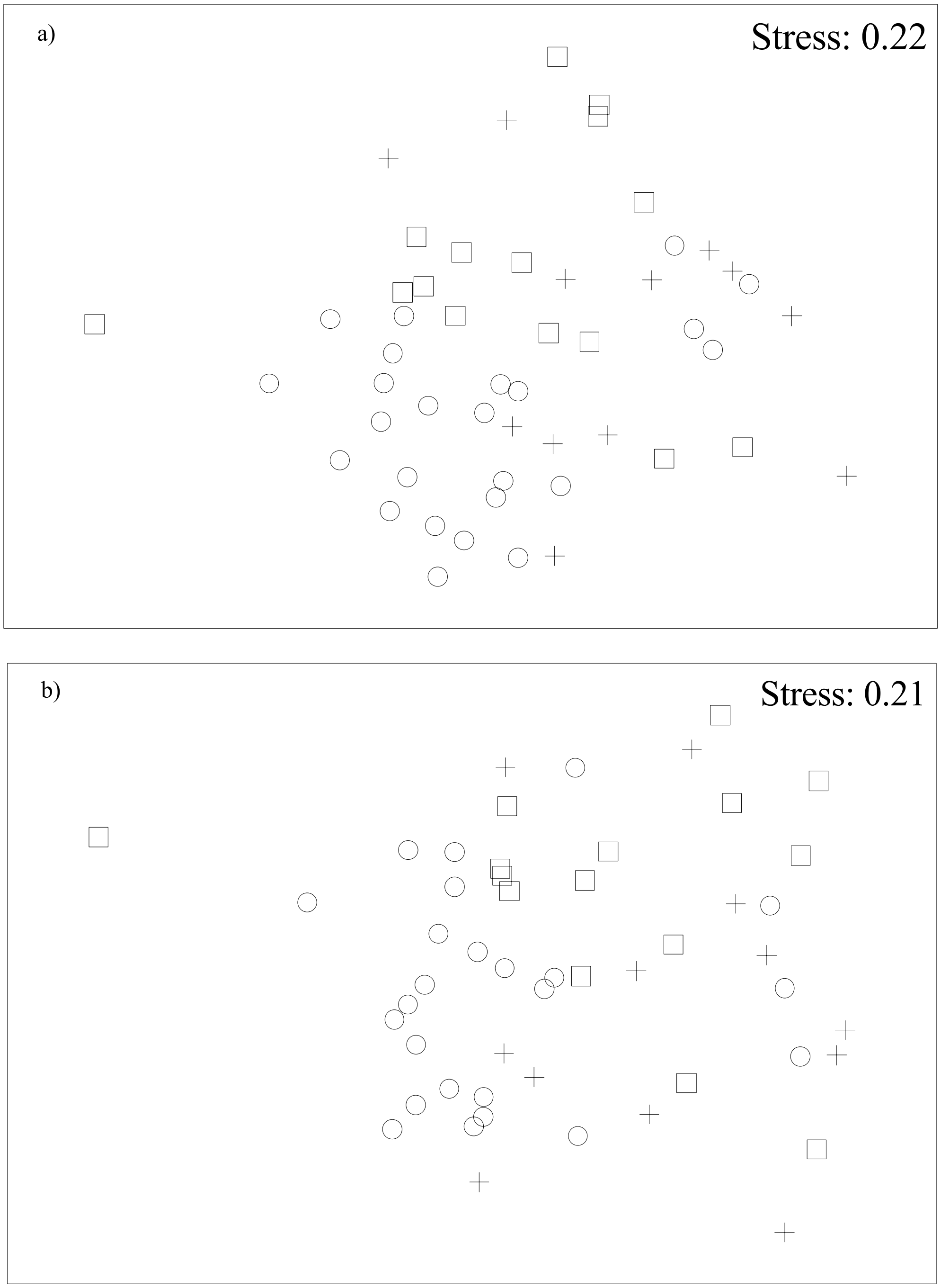
Appendix 11: Macroinvertebrate species and total abundance in natural wetlands (NW), ‘no tailwater’ storages (TW) and typical storages (TS) in the Border Rivers Catchment during sampling from August 2005 to February, 2007.

		NW	TW	TS
CRUSTACEA				
BRANCHIOPODA				
Anostraca	*Branchinella* sp.	173		
Notostraca	Indeterminate sp.	1		
Conchostraca	Indeterminate sp.	23		15
DECAPODA				
Aytidae	*Parataya australiensis*	51		12
Palaemonidae	*Machrobrachium australiense*	138	1	41
INSECTA				
LEPIDOPTERA				
Pyralidae	*Nymphulinae* sp.	8		3
COLEOPTERA				
Adults				
Dysticidae	*Allodesus* sp.	1		1
	Hyphydrus sp.	1		
	Hydroglyphus sp.			1
	Hydraena sp.	1		3
	Megaporus sp.			1
	Necterosoma sp.	1		
	Onychohydrus scutellaris			1
Haliplidae	*Haliplus* sp.	1		
Hydraenidae	Octhebiinae subtribe	43		1
Hydrophilidae	*Berosus* sp.	1		11
	Coelostoma sp.	2	1	
	Paracymus sp.	1		
	Enochrus mastersi	1		
Elmidae	Indeterminate sp.	20		12
Curculionidae	Indeterminate sp.	8		
Hydrochidae	*Hydrochus* sp.	6		1
Larvae				
Hydrophilidae	*Berosus* sp.	3	2	3
Dytisicidae	Tribe Bidessini			1
	Eretes australis	1		42
	Helochares sp.	1		
	Spercheus sp.	1		
DIPTERA				
Chaoboridae	Indeterminate sp.	355		35
Culicidae	Indeterminate sp.	1311		
Tabanidae	Indeterminate sp.	33		
Chironomidae				
Chironominae	*Cladopema* sp.	300		520
	Cryptochironomus sp.	21	4	
	Dicrotendipes sp.	234	63	1015
	Harnischia sp.	36	6	1
	Parachironomus sp.	423	19	65
	Paracladopelma sp.	11	20	180
	Polypedilum sp.	33	22	1
	Stenochironomus sp.	9		
	Stictochironomus sp.	5		
	Tanytarsus sp.	65	1	59
	Cladotanytarsus sp.	556		120
	Paratanytarsus sp.	29		
Chironomidae				

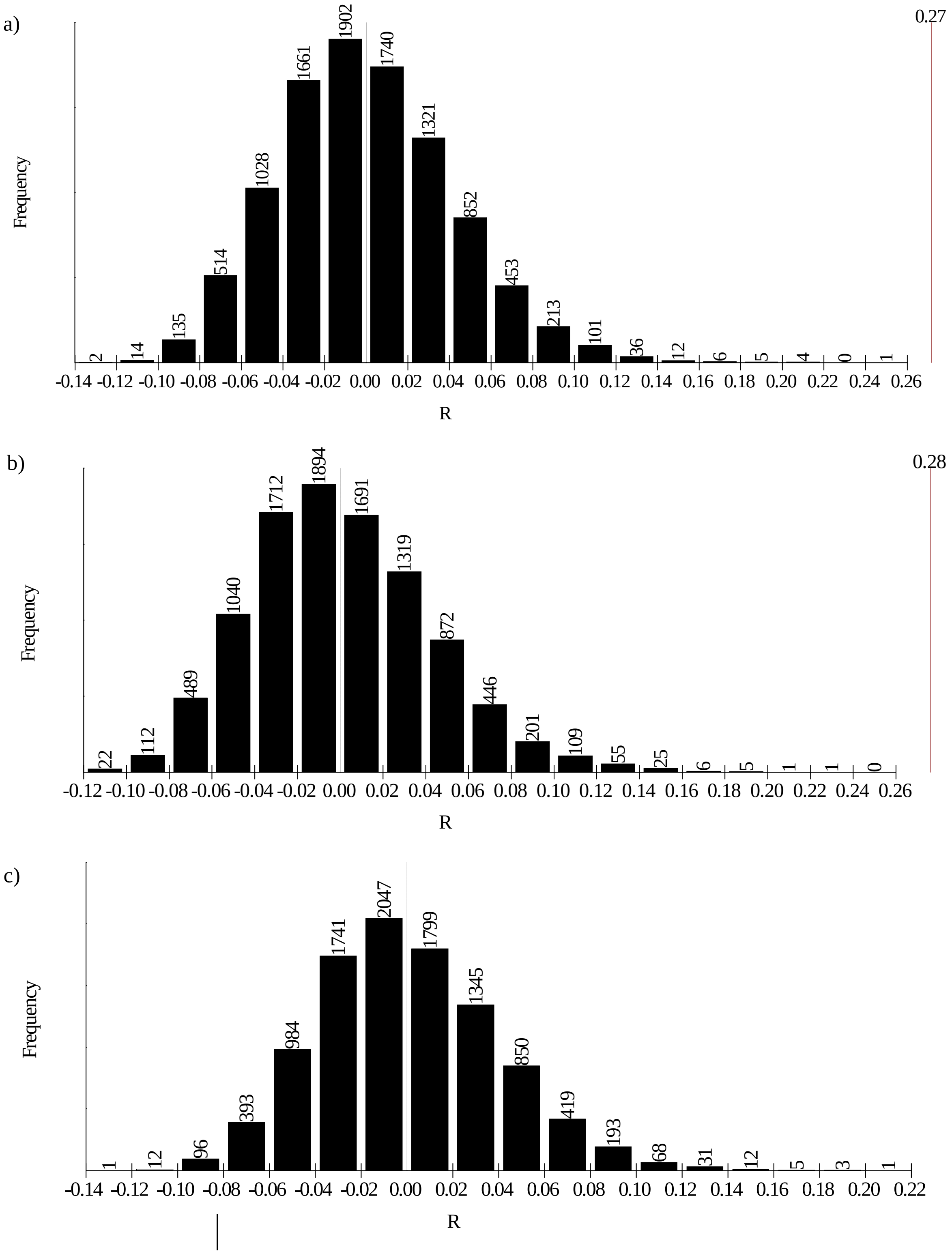
		NW	TW	TS
Tanypodinae	*Clinotanypus* sp.	2		
	Coelopynia sp.	49	40	130
	Larsia sp.	80		1
	Paramerina sp.	3		
	Pentaneura sp.	1		
	Procladius sp.	744	97	286
Chironomidae				
Orthocladinae	*Ablabesmyia* sp.	19	239	1
	Cricotopus sp.	434	107	867
	Nanocladius sp.	152		262
	Paratrichocladius sp.		1	
Chironomidae	Mutant sp.		1	
	Tiny chironomidae	2719	565	1789
	Pupae	62	6	21
Ceratopogonidae				
Ceraotpogoninae	Indeterminate sp.	1114	15	96
Dasyheleinae	Indeterminate sp.	4		1
Ceratopogonidae	Pupae	119		
	Tiny Ceratopogonidae	725	40	255
EPHEMEROPTERA				
Baetidae	*Cloeon* sp.	634	6	57
Caenidae	*Wundacaenis* sp.	19	2	5
	Tasmanocoenis sp.	15	7	
	Tiny Ephemeroptera	423	30	
HEMIPTERA				
Notonectidae	*Anisops* sp.	492	263	69
Corixidae	*Micronecta* sp.	1813	5582	5324
	Agraptocorixa sp.	21	215	27
Gelastocoridae	*Nerthra* sp.	1		
Gerridae	*Rheumatometra philarete*	2		1
Mesoveliidae	*Mesovelia* sp.	92		1
Naucoridae	*Naucoris* sp.	1		
Hebridae	Indeterminate sp.	3		
Veliidae	Indeterminate sp.	2		
Pleidae	*Neoplea* sp.	2		
Saldidae	Indeterminate sp.	3		2
ODONATA				
Aeshnidae	*Hemiarex papuensis*	16		
Coenagrionidae	*Austrocnemis splendida*	371		3
	Ischnura aurora	31		1
	Ischnura heterosticta	22		
	Pseudagrion ignifer	7		
	Pseudagrion microcephalum	2		
	Xanthagrion erythroneurum	5		
Hemicoduliidae	*Hemicordulia tau*	1		
Isosticidae	*Rhadinosticta simplex*	1		
Libellulidae	*Orthetrum caledonicum*	1		
	Diplacodes bipunctata	2		
	Tiny Odonata	696		3
	Indeterminate sp.	10		2
PLECOPTERA	Tiny Plecoptera	3		4
TRICHOPTERA				
Leptoceridae	*Triplectides australis*	42	3	10
	Triplectides proximus	2		1
	Oecitis sp.	9	61	
	Notalina spira	1	1	
Hydroptilidae	*Hellyethira maleoforma*	34	133	
Ecnomidae	*Ecnomus* sp.	4		
	Indetermniate sp.		20	

		NW	TW	TS
	Tiny Trichoptera	150	21	3
GASTROPODA				
BASOMMATOPHORA				
Planorbidae	*Glytophysa* sp.	56		2
Ancylidae	*Ferrissia* sp.	23		5
Lymnaeidae	*Austropeplea*	2		
CAENOGASTROPODA				
Physidae	*Physa acuta*	90		1
OLIGOCHAETA	Indeterminate sp.	85		16
NEMERTEA	Indeterminate sp.	83		103
ARACHNIDA				
Acarnina	Indeterminate sp.	48		1
	Tiny acarina	10		
NEMATODA	Indeterminate sp.	5044	17	430
CNIDARIAN				
HYDROZOA				
Hydridae	Indeterminate sp.	450		35
BRYOZOA	Indeterminate sp.	840		15

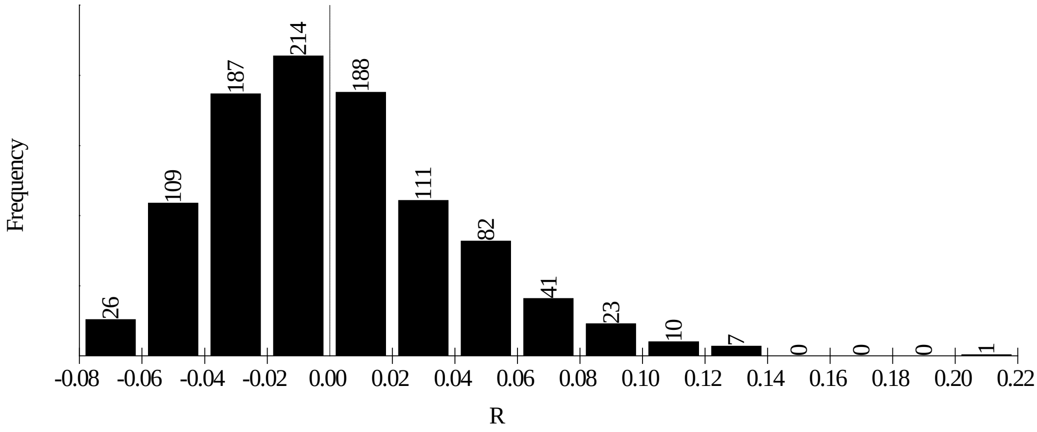
Appendix 12: Non-metric multi-dimensional scaling plots for macroinvertebrate assemblage structure based on a) presence/absence data, and b) non-standardised Log$_{10}$(x+1). O = Natural wetlands, + = Storages without tailwater, □ = Typical storages.



Appendix 13: Histograms for ANOSIM results of macroinvertebrate data for a) presence/absence data (R = 0.27), b) absolute abundance Log$_{10}$(x+1) (R = 0.28) and c) relative abundance Log$_{10}$(x+1) (R = 0.37).



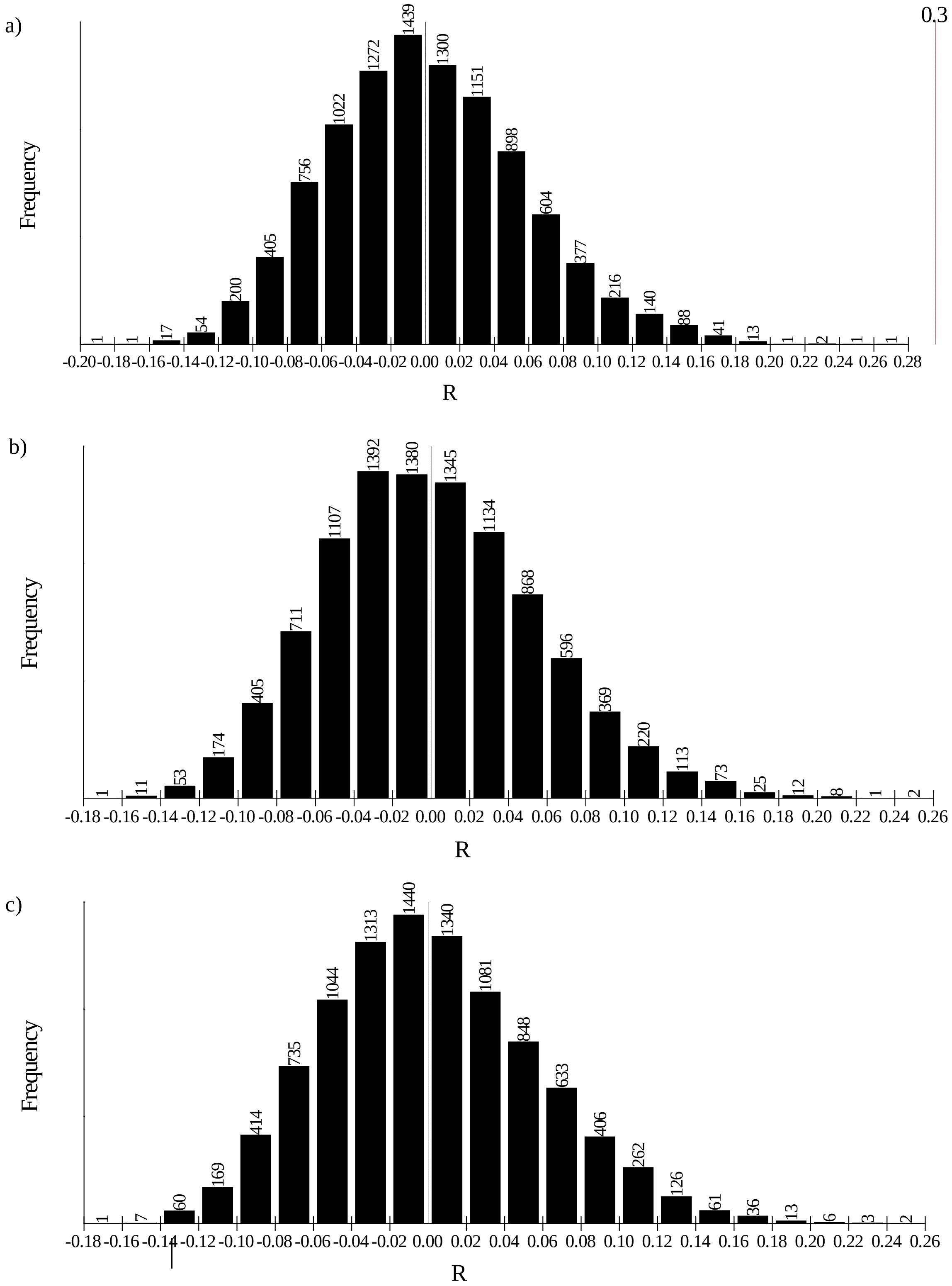
Appendix 14: Histogram for ANOSIM results of macroinvertebrate assemblage function data based on relative abundance $Log_{10}(x+1)$ (R = 0.46).



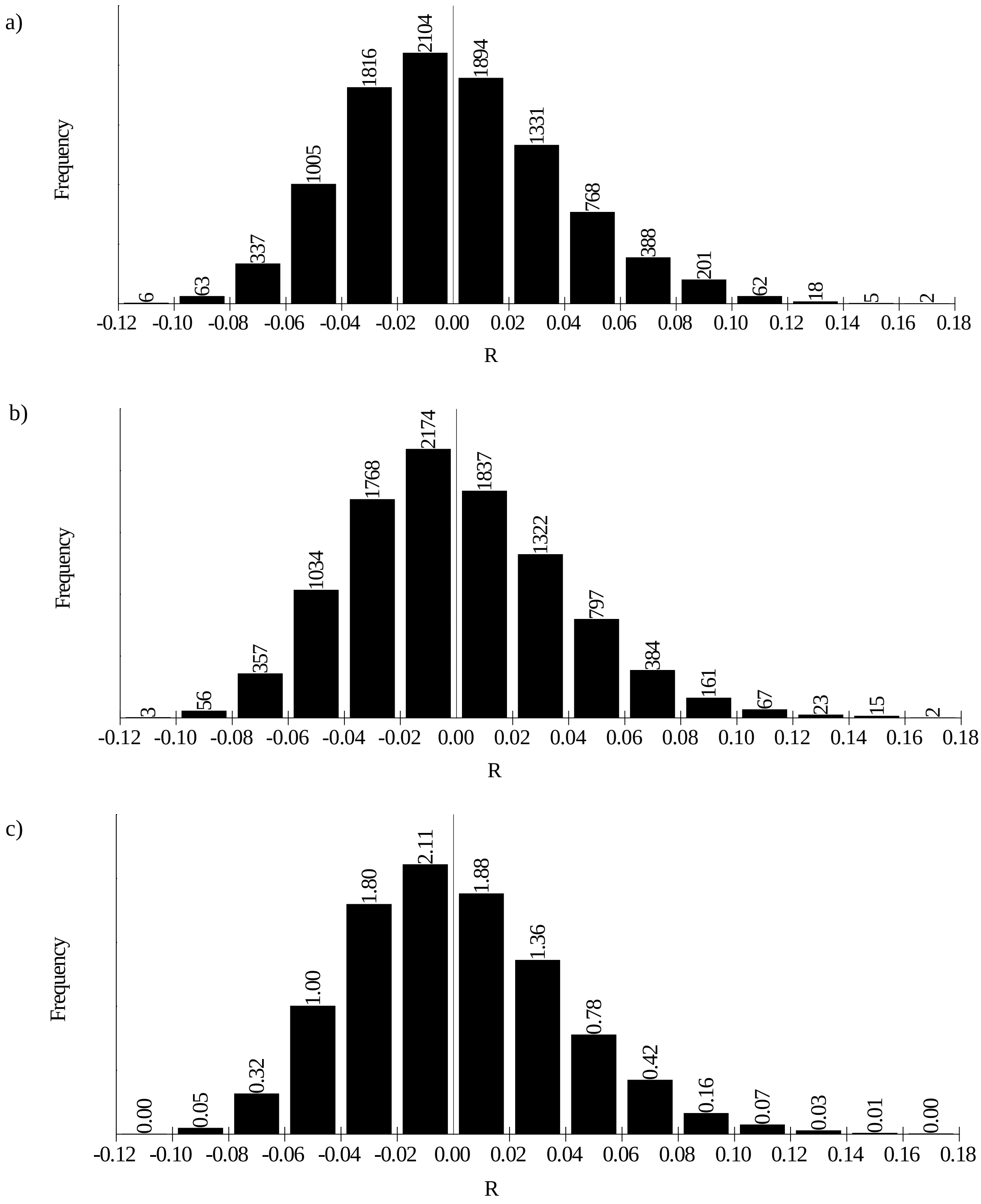
Appendix 15: Summary of ANOSIM results comparing macroinvertebrate assemblage structure (based on $\log_{10}(x+1)$, $\log_{10}(x+1)$ standardised for relative abundance and presence absence data) among waterbody types (NW = natural wetlands, TS = typical storages) and sampling times occasions (2 = November 2005, 3 = February 2006, 4 = March 2006, 5 = December 2006, 6 = January 2007, 7 = February 2007). * *P* = 0.001, ** *P* < 0.01, ****P* < 0.05.

Waterbody	Transformation	Factor	Global R	*P*	Significant pairwise tests
NW	$\text{Log}_{10}(x+1)$Std	Time	0.373	<0.001	3-5, 3-7, 4-7 <0.001 2-3, 2-4 * 3-4 , 4-5, 6-7 ** 2-5, 2-6, 2-7, 5-6, 5-7 ***
NW	$\text{Log}_{10}(x+1)$	Time	0.369	<0.001	3-5, 3-7, 4-7 <0.001 2-3, 2-4 * 4-5, 6-7 ** 2-5, 2-6, 2-7, 3-4, 5-6 5-7 ***
NW	Presence/Absence	Time	0.269	<0.001	2-3, 2-4 * 3-5, 6-7 ** 2-5, 2-6, 2-7, 3-4, 3-7, 4-5, 4-7 ***
TS	$\text{Log}_{10}(x+1)$Std	Time	0.305	<0.001	2-5, 2-6, 3-5, 3-6, 4-6 <0.001 2-7, 4-5, 4-7, 5-6, 5-7 ** 6-7 ***
TS	$\text{Log}_{10}(x+1)$	Time	0.3	<0.001	2-5, 2-6, 3-5, 3-6, 4-5, 4-6 <0.001 2-7, 4-7 ** 5-7, 6-7 ***
TS	Presence/Absence	Time	0.282	<0.001	2-5, 2-6, 3-5, 3-6, 4-6 <0.001 4-5 * 5-7, 6-7 ** 2-7, 4-7, 5-6 ***
NW & TS	$\text{Log}_{10}(x+1)$Std	Waterbody	0.406	<0.001	
NW & TS	$\text{Log}_{10}(x+1)$	Waterbody	0.416	<0.001	
NW & TS	Presence/Absence	Waterbody	0.359	<0.001	
NW & TS	$\text{Log}_{10}(x+1)$Std	Time	0.340	<0.001	2-3, 2-4, 2-5, 2-6, 2-7, 3-5, 3-7 4-5, 4-7 <0.001 3-6, 4-6 * 3-4, 5-6, 5-7 6-7 **
NW & TS	$\text{Log}_{10}(x+1)$	Time	0.336	<0.001	2-3, 2-4, 2-5, 2-6, 2-7, 3-5, 3-6, 3-7, 4-5, 4-6, 4-7 <0.001 3-4, 5-7 & 6-7 ** 5-6 ***
NW & TS	Presence/Absence	Time	0.289	<0.001	2-4, 2-5, 2-6, 2-7, 3-5, 4-5 6-7 <0.001 2-3, 3-6 * 3-7, 4-6, 4-7, 5-7 ** 3-4, 5-6 ***

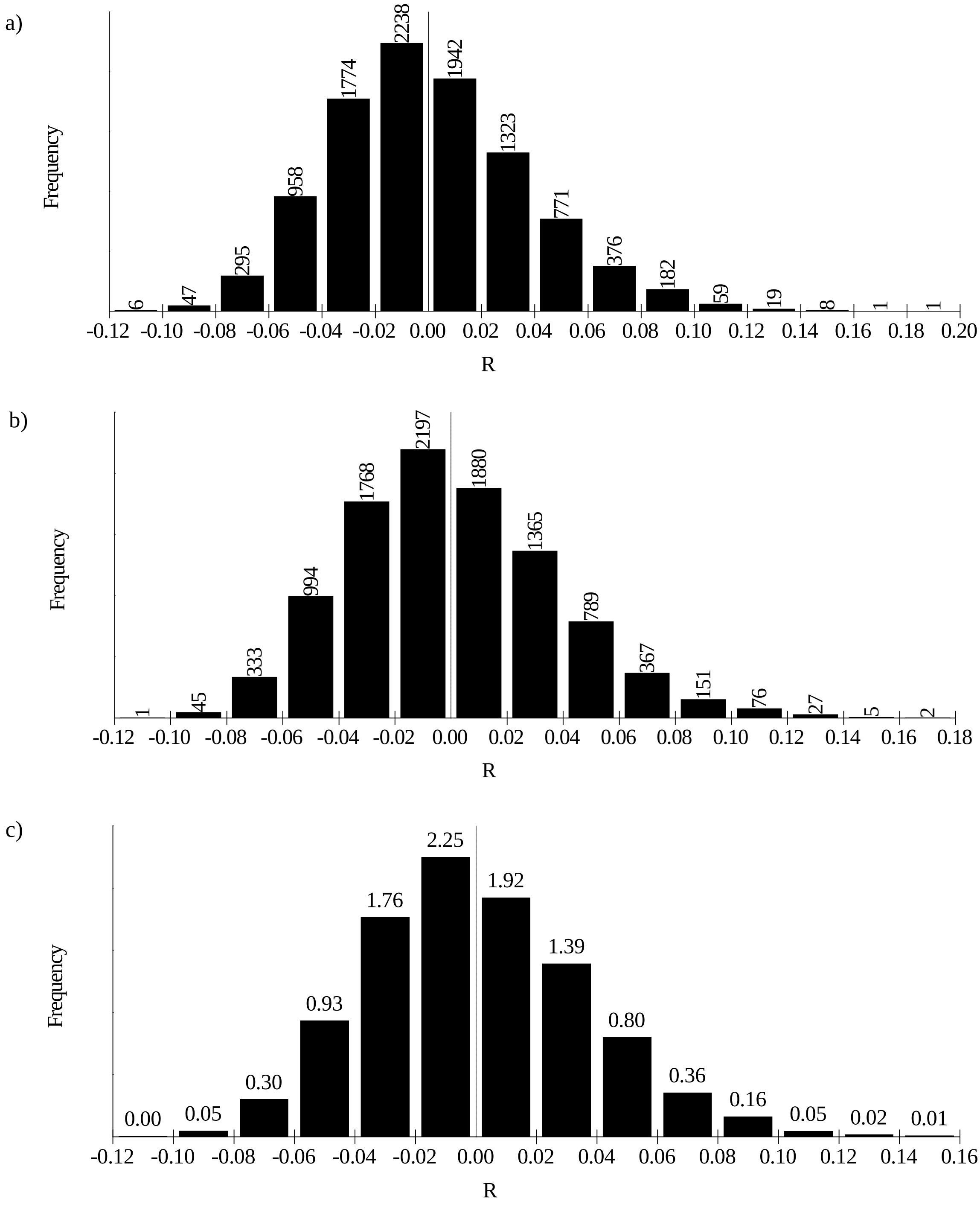
Appendix 16: Histograms for ANOSIM results of macroinvertebrate assemblage structure in natural wetlands based on a) presence/absence data (R = 0.27), b) absolute abundance $\text{Log}_{10}(x+1)$ (R = 0.37) and c) relative abundance $\text{Log}_{10}(x+1)$ (R = 0.37) across temporal sampling occasions November 2005 to February 2007.



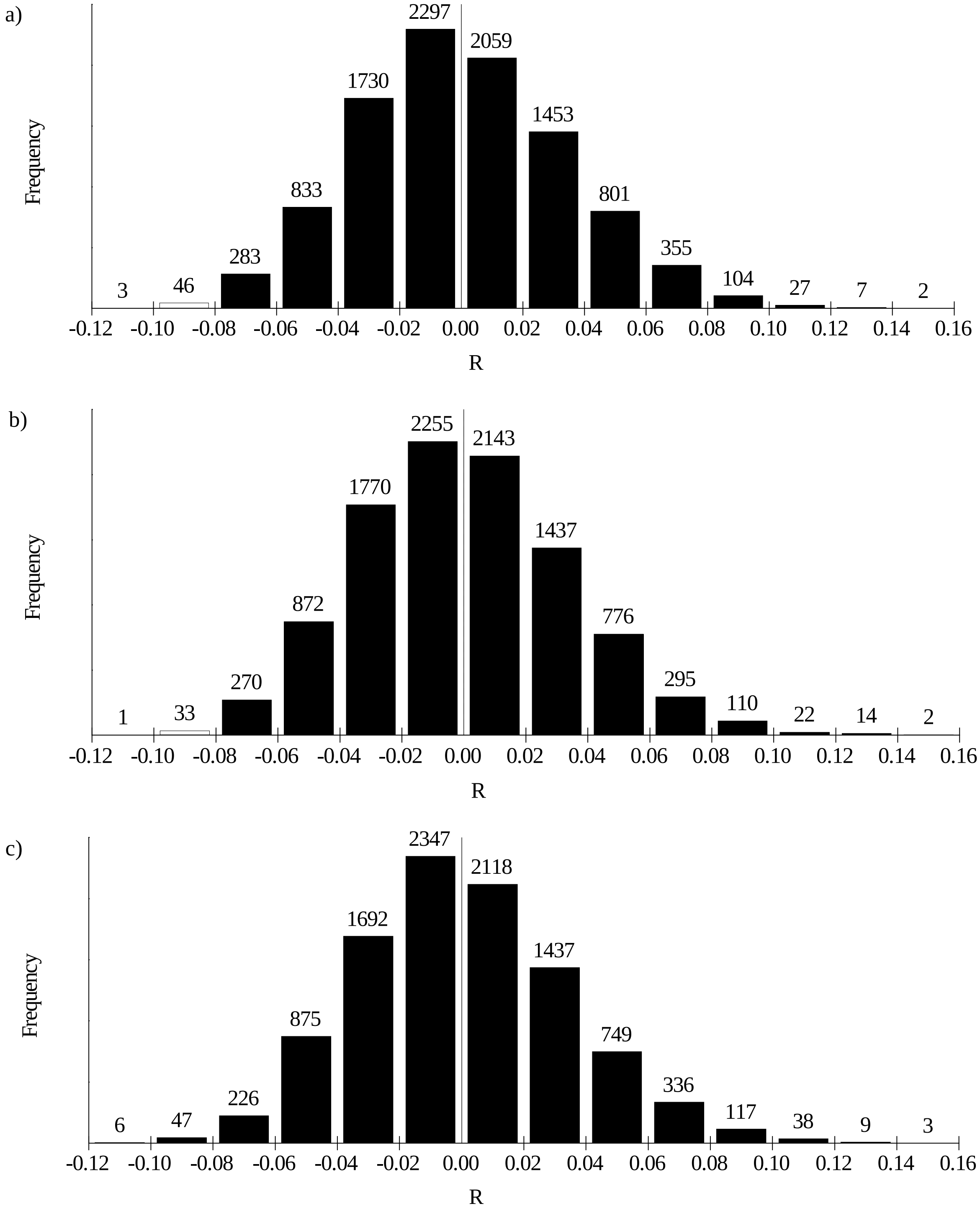
Appendix 17: Histograms for ANOSIM results of macroinvertebrate assemblage structure in typical storages based on a) presence/absence data (R = 0.28), b) absolute abundance $\text{Log}_{10}(x+1)$ (R = 0.30) and c) relative abundance $\text{Log}_{10}(x+1)$ (R = 0.31) across temporal sampling occasions November 2005 to February 2007.



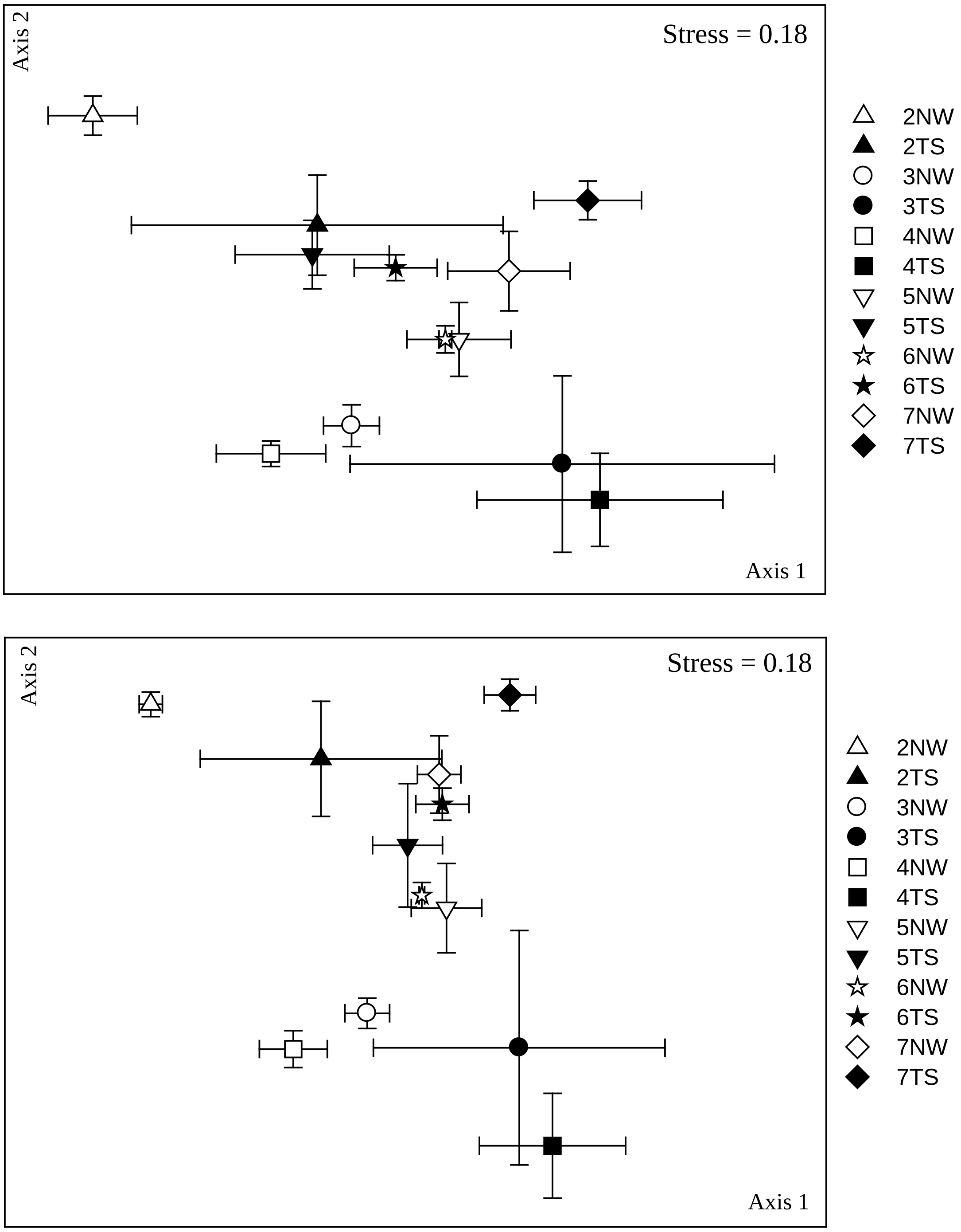
Appendix 18: Histograms for ANOSIM results comparing macroinvertebrate assemblage structure between typical storages and natural wetlands based on a) presence/absence data (R = 0.36), b) absolute abundance $\text{Log}_{10}(x+1)$ (R = 0.42) and c) relative abundance $\text{Log}_{10}(x+1)$ (R = 0.41) across temporal sampling occasions November 2005 to February 2007.



Appendix 19: Histograms for ANOSIM results comparing macroinvertebrate assemblage structure between temporal sampling occasions November 2005 to February 2007 in typical storages and natural wetlands based on a) presence/absence data (R = 0.29), b) absolute abundance $Log_{10}(x+1)$ (R = 0.34) and c) relative abundance $Log_{10}(x+1)$ (R = 0.34).

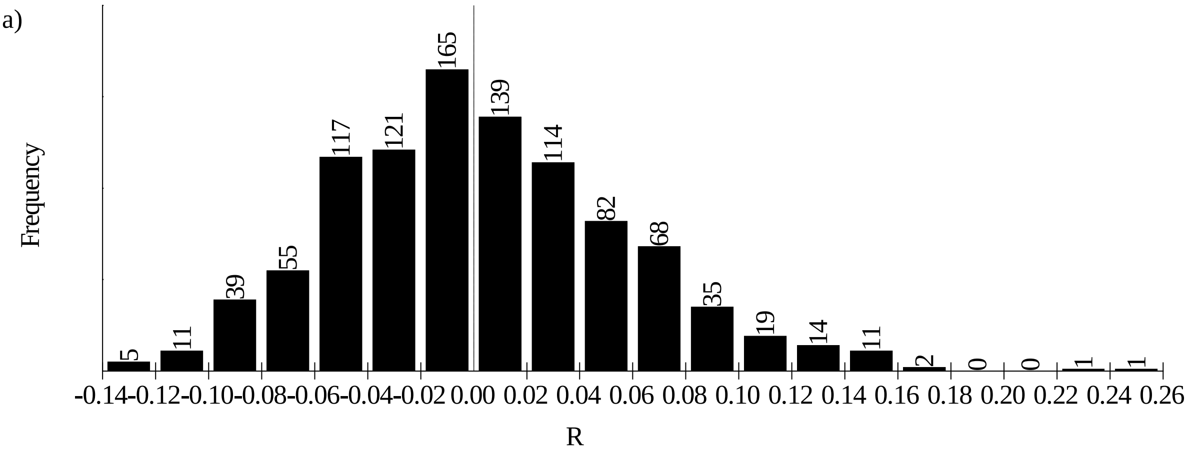


Appendix 20: Non-metric multi-dimensional scaling plots for macroinvertebrate assemblage structure based on a) presence absence data and b) $\text{Log}_{10}(x+1)$ data, across all temporal sampling times for natural wetlands (open) and typical storages (closed). Samples collected from each waterbody type on each sampling occasion are represented as centroids (mean x-y coordinates) ± 1 s.e. bars. 2 = November 2005, 3 = February 2006, 4 = March 2006, 5 = December 2006, 6 = January 2007, 7 = February 2007. NW = natural wetland and TS = typical storage.

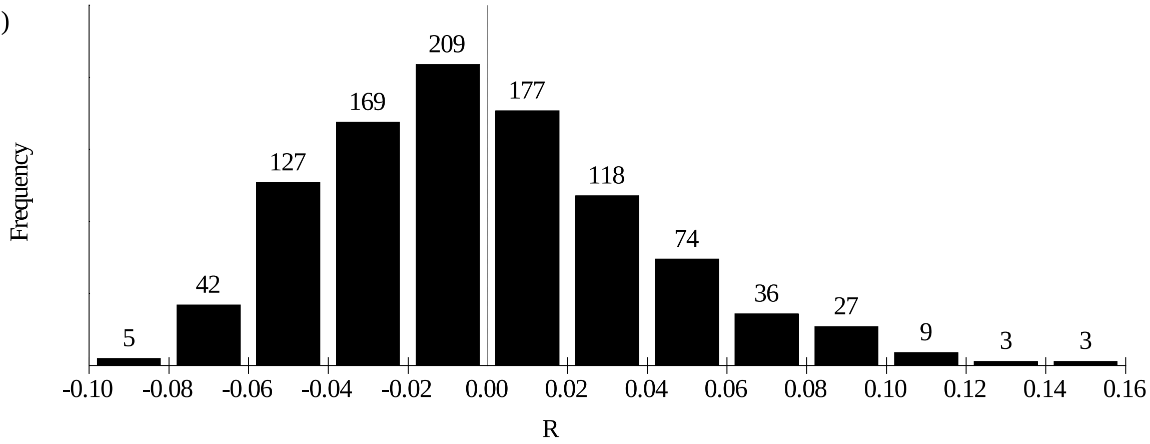


Appendix 21: Histograms for one-way ANOSIM results comparing macroinvertebrate assemblage structure in a) natural wetlands (R = 0.35) and b) typical storages (R = 0.35) based on relative abundance $\text{Log}_{10}(x+1)$ data between temporal sampling occasions (November 2005 to February 2007).

a)

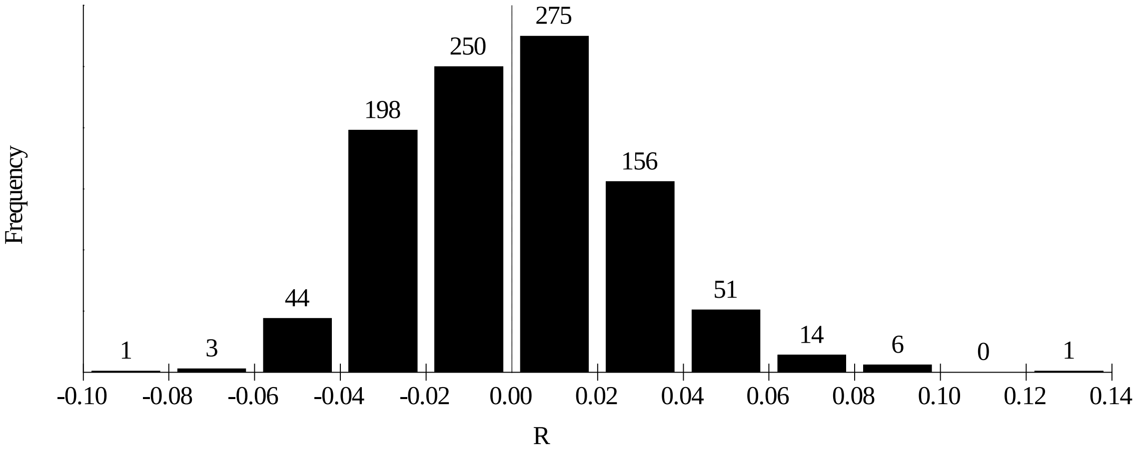


b)

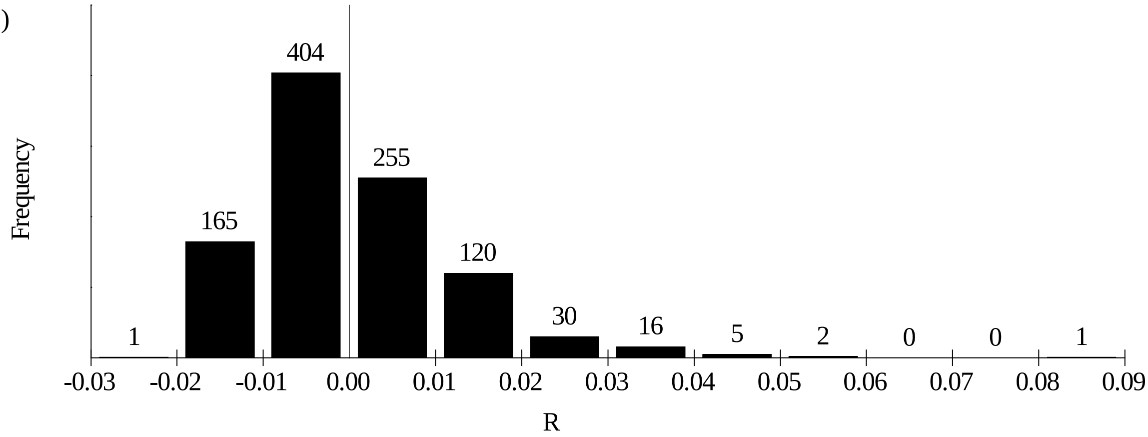


Appendix 22: Histograms for two-way ANOSIM results comparing macroinvertebrate assemblage structure between a) sampling times across both waterbody types (R = 0.35) and b) waterbody type across all times (R = 0.31) based on relative abundance $\mathrm{Log}_{10}(x+1)$ data for natural wetlands and typical storages across all temporal sampling occasions (November 2005 to February 2007).

a)

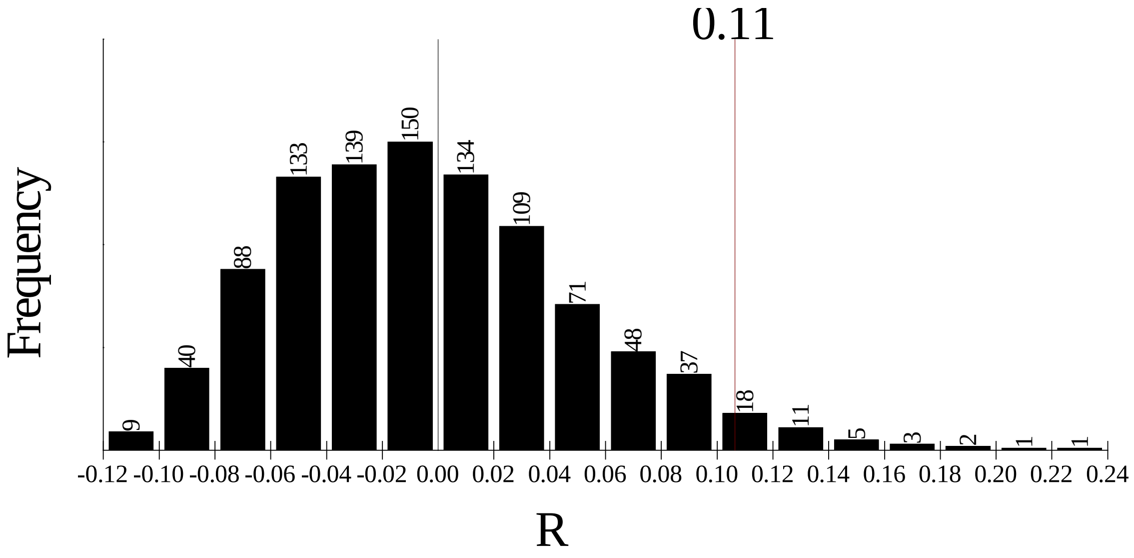


b)

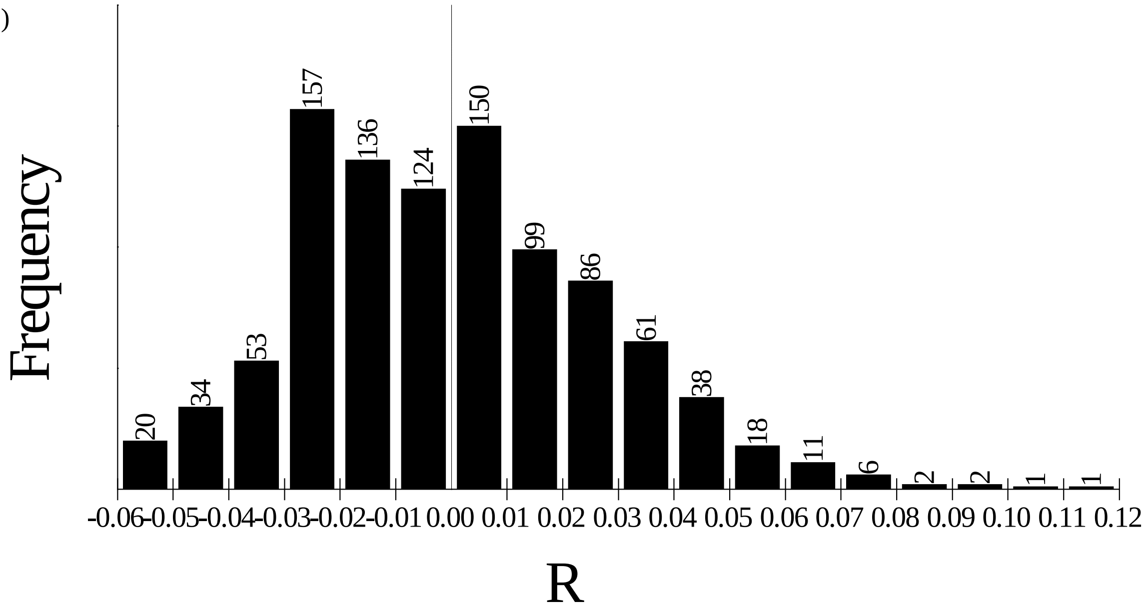


Appendix 23: Histograms for one-way ANOSIM results comparing EPT scores in a) natural wetlands (R = 0.11) and b) typical storages (R = 0.15) between temporal sampling occasions (November 2005 to February 2007).

a)

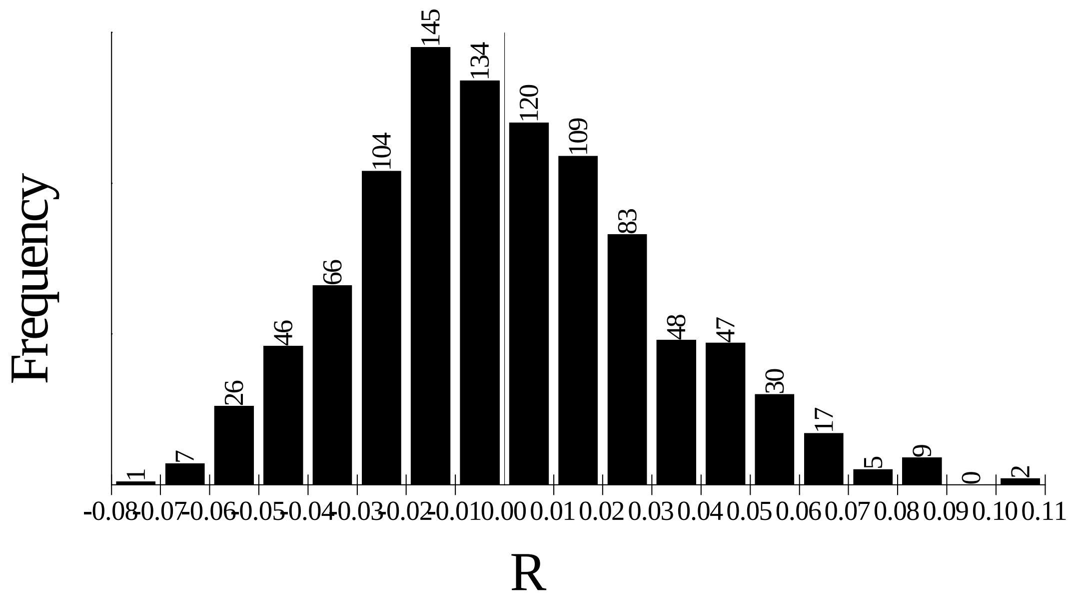


b)

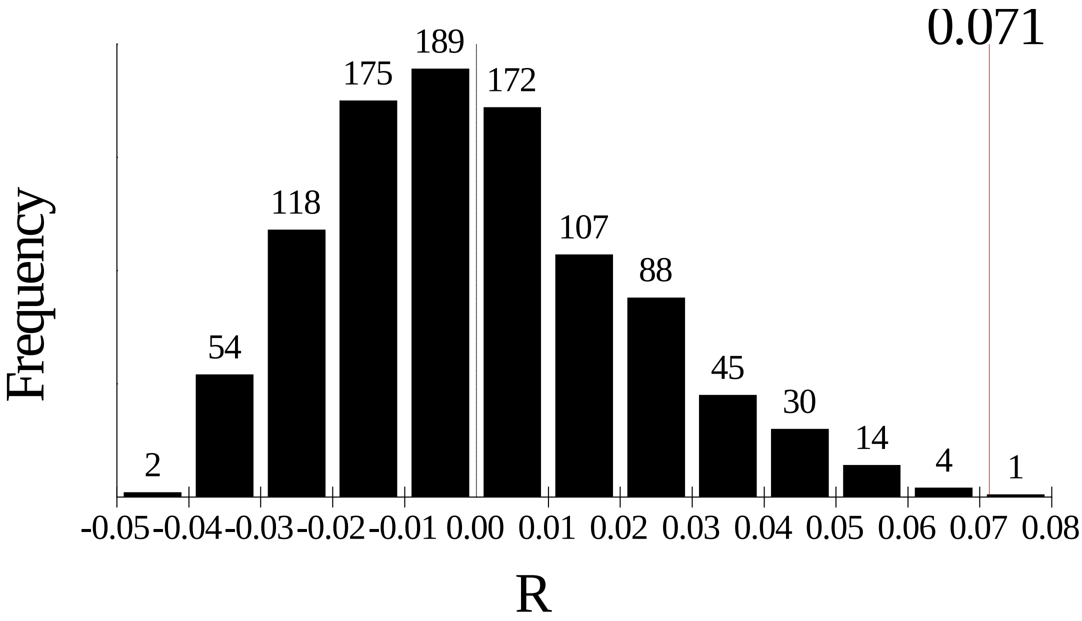


Appendix 24: Histograms for two-way ANOSIM results comparing EPT scores between a) sampling times across both waterbody types (R = 0.13) and b) waterbody type across all times (R = 0.07) based on data for natural wetlands and typical storages across all temporal sampling occasions (November 2005 to February 2007).

a)

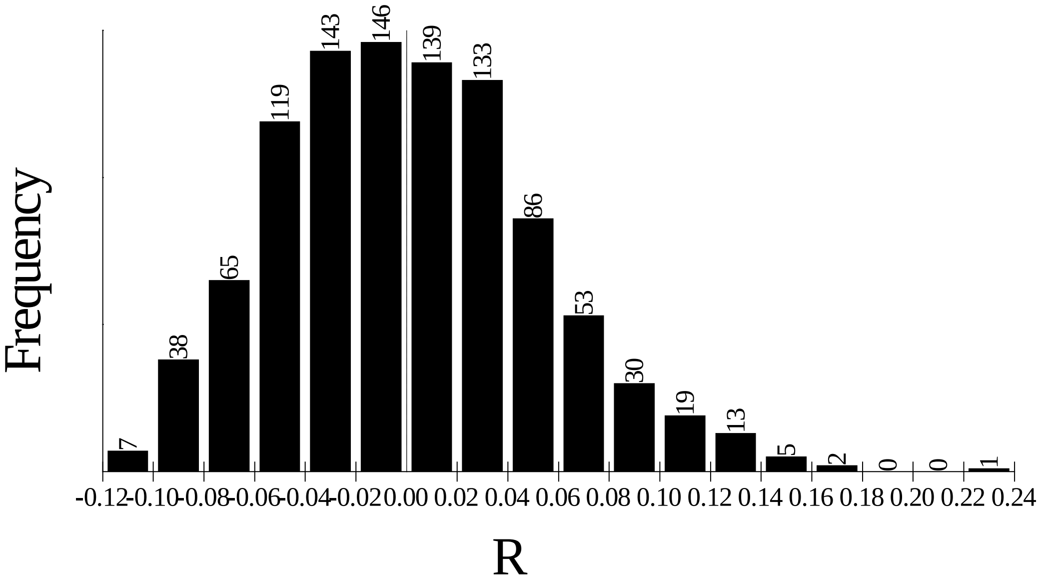


b)

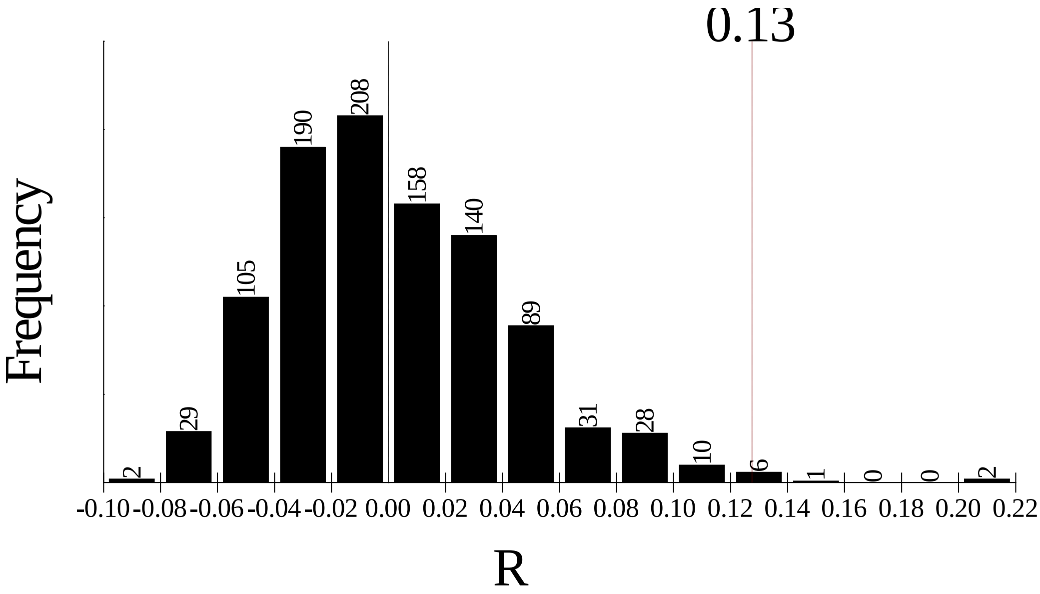


Appendix 25: Histograms for one-way ANOSIM results comparing SIGNAL scores in a) natural wetlands (R = 0.34) and b) typical storages (R = 0.13) between temporal sampling occasions (November 2005 to February 2007).

a)

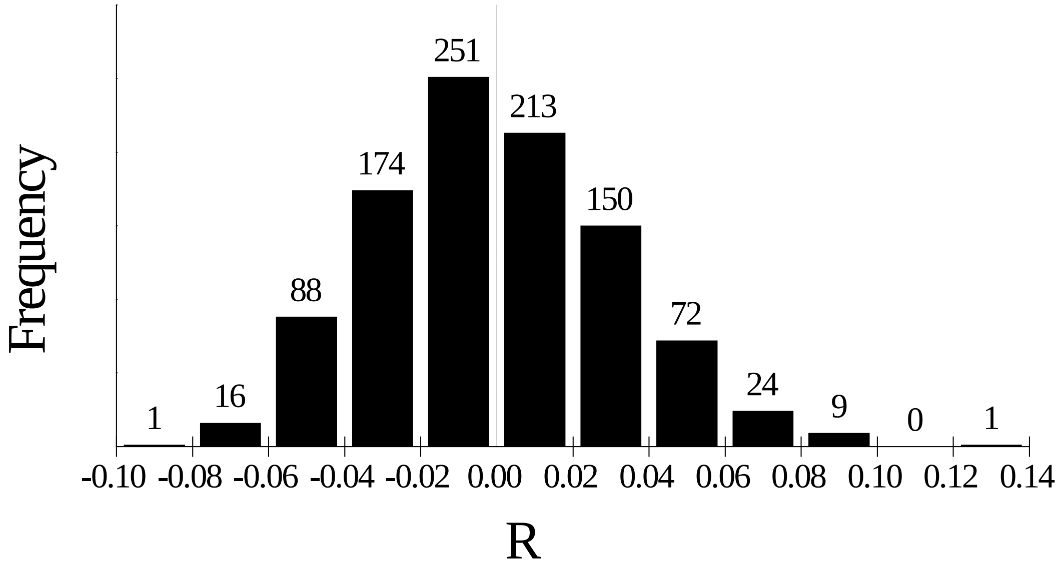


b)

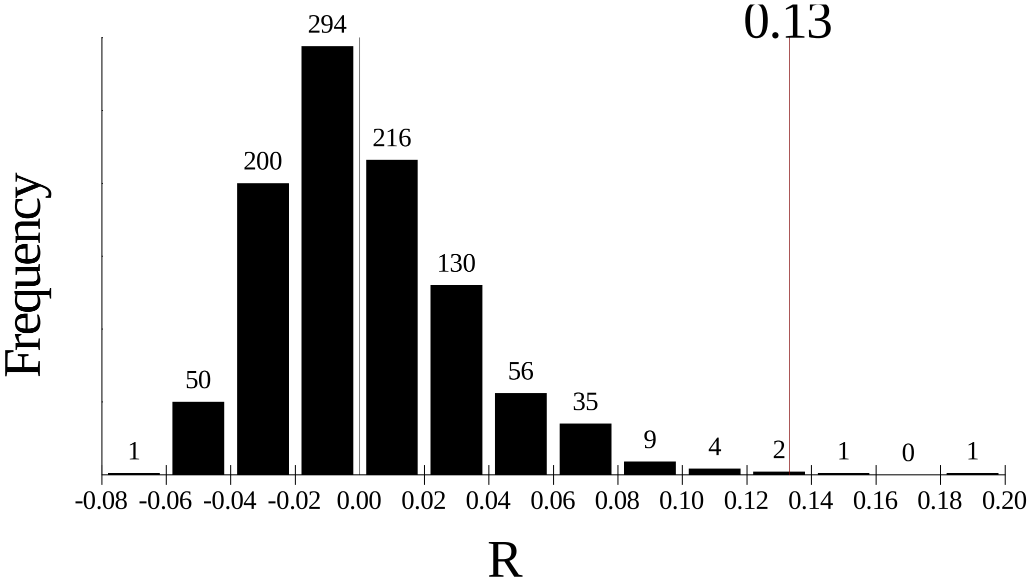


Appendix 26: Histograms for two-way ANOSIM results comparing EPT scores between a) sampling times across both waterbody types (R = 0.23) and b) waterbody type across all times (R = 0.13) based on data for natural wetlands and typical storages across all temporal sampling occasions (November 2005 to February 2007).

a)



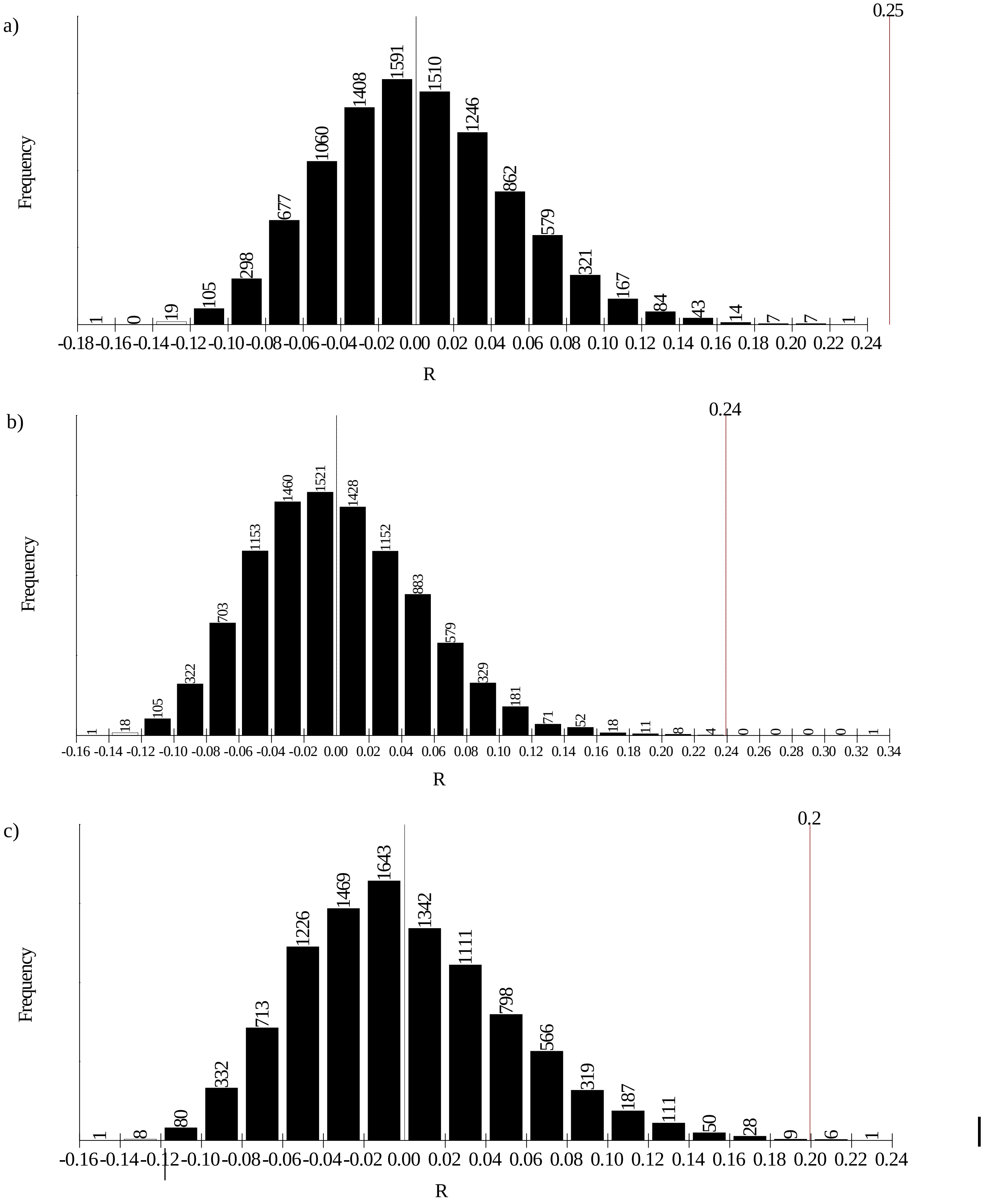
b)



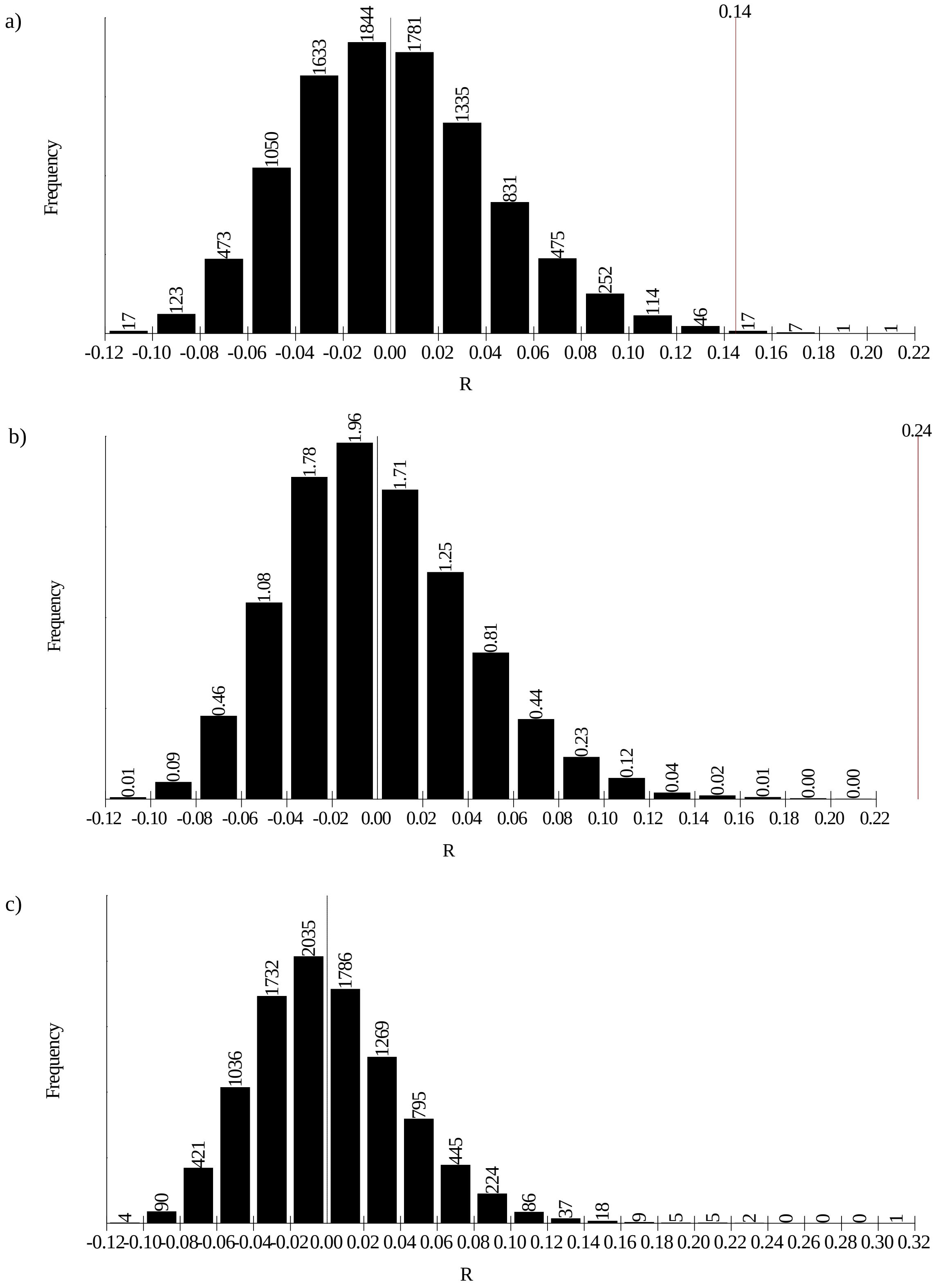
Appendix 27: Summary of ANOSIM results comparing zooplankton assemblage structure (based on $\log_{10}(x+1)$, $\log_{10}(x+1)$ standardised for relative abundance and presence absence data) among waterbody types (NW = natural wetlands, TS = typical storages) and sampling times occasions (3 = February 2006, 4 = March 2006, 5 = December 2006, 6 = January 2007, 7 = February 2007). * *P* = 0.001, ** *P*<0.01, ****P*<0.05.

Waterbody	Transformation	Factor	Global R	*P*	Significant pairwise tests
NW	$\text{Log}_{10}(x+1)$Std	Time	0.199	0.001	4-5 <0.001 3-4, 4-6, 4-7 **
NW	$\text{Log}_{10}(x+1)$	Time	0.239	<0.001	4-6, 4-7 * 3-7, 4-5 ** 3-4, 3-6 ***
NW	Presence/Absence	Time	0.252	<0.001	3-7, 4-6, 4-7 <0.001 4-5 ** 3-6 ***
TS	$\text{Log}_{10}(x+1)$Std	Time	0.365	<0.001	3-5, 3-6, 3-7, 4-5, 4-6, 4-7 <0.001 5-7 ***
TS	$\text{Log}_{10}(x+1)$	Time	0.238	<0.001	3-6, 4-6 <0.001 3-7 * 4-7 ** 3-5, 4-5 ***
TS	Presence/Absence	Time	0.145	0.002	3-6, 4-6 * 3-7 ** 3-5, 4-7 ***
NW & TS	$\text{Log}_{10}(x+1)$Std	Waterbody	0.238	<0.001	
NW & TS	$\text{Log}_{10}(x+1)$	Waterbody	0.219	<0.001	
NW & TS	Presence/Absence	Waterbody	0.139	0.001	
NW & TS	$\text{Log}_{10}(x+1)$Std	Time	0.285	<0.001	3-7, 4-5, 4-6, 4-7 <0.001 3-4, 3-5, 3-6 ** 5-7 ***
NW & TS	$\text{Log}_{10}(x+1)$	Time	0.239	<0.001	3-6, 3-7, 4-5, 4-6, 4-7 <0.001 3-4, 3-5 ***
NW & TS	Presence/Absence	Time	0.197	<0.001	3-6, 3-7, 4-6, 4-7 <0.001 3-5, 4-5 ***

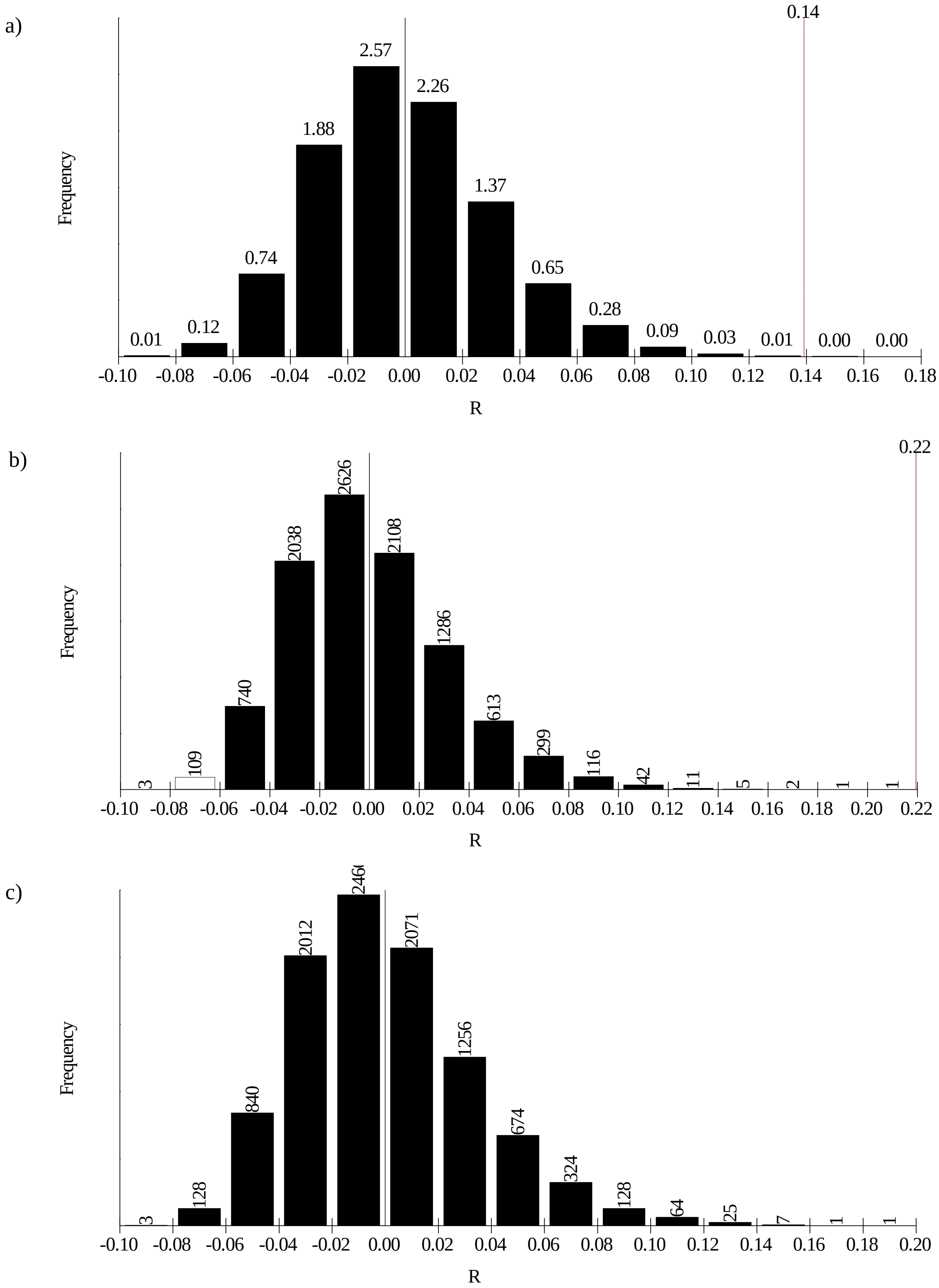
Appendix 28: Histograms for ANOSIM results of zooplankton assemblage structure in natural wetlands based on a) presence/absence data (R = 0.25), b) absolute abundance $\text{Log}_{10}(x+1)$ (R = 0.24) and c) relative abundance $\text{Log}_{10}(x+1)$ (R = 0.20) across temporal sampling occasions February 2006 to February 2007.



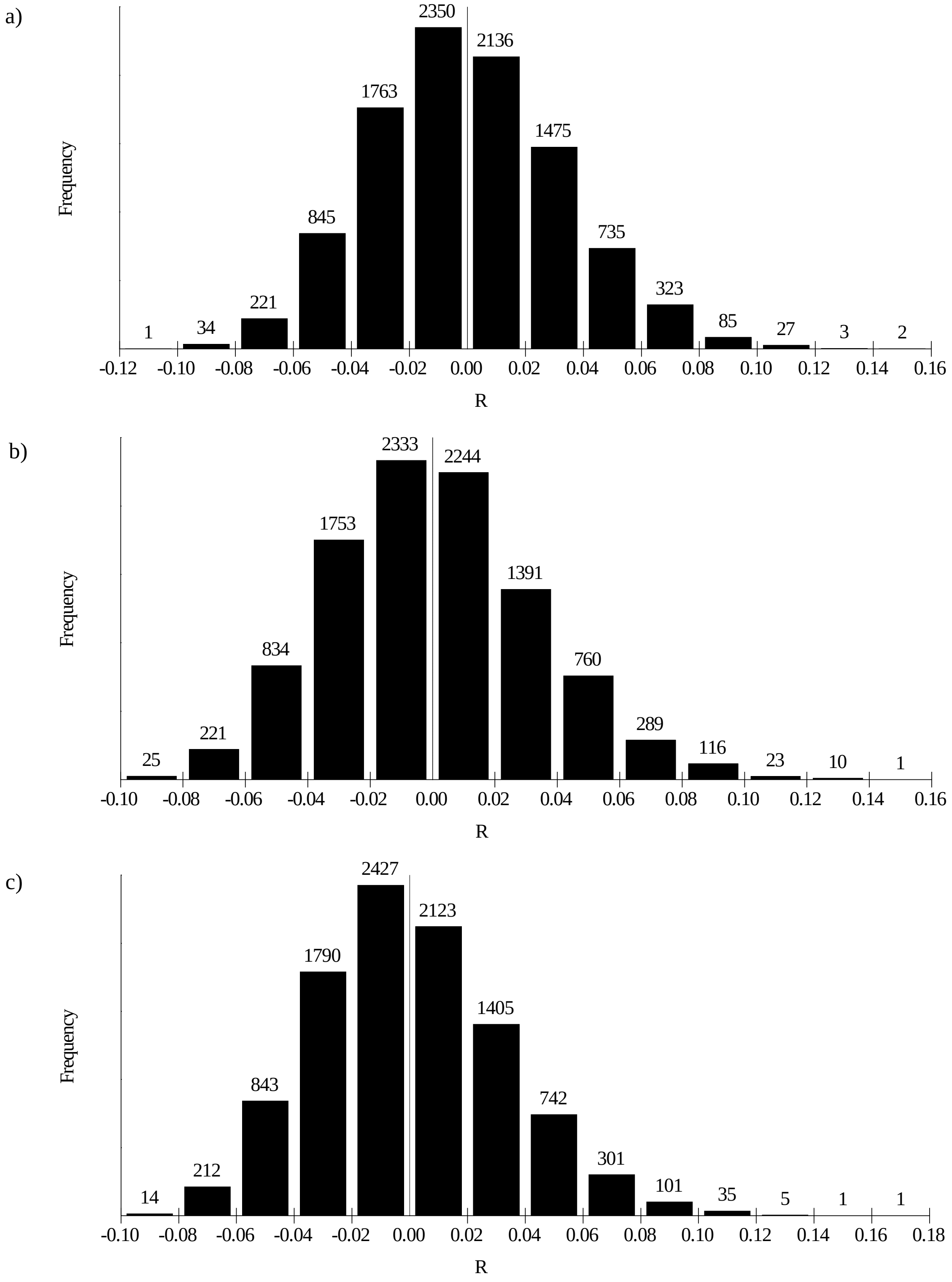
Appendix 29: Histograms for ANOSIM results of zooplankton assemblage structure in typical storages based on a) presence/absence data (R = 0.15), b) absolute abundance $Log_{10}(x+1)$ (R = 0.24) and c) relative abundance $Log_{10}(x+1)$ (R = 0.37) across temporal sampling occasions February 2006 to February 2007.



Appendix 30: Histograms for ANOSIM results comparing zooplankton assemblage structure between typical storages and natural wetlands based on a) presence/absence data (R = 0.14), b) absolute abundance $\text{Log}_{10}(x+1)$ (R = 0.22) and c) relative abundance $\text{Log}_{10}(x+1)$ (R = 0.24) across temporal sampling occasions February 2006 to February 2007.



Appendix 31: Histograms for ANOSIM results comparing zooplankton assemblage structure between temporal sampling occasions February 2006 to February 2007 in typical storages and natural wetlands based on a) presence/absence data (R = 0.20), b) absolute abundance $\mathrm{Log}_{10}(x+1)$ (R = 0.24) and c) relative abundance $\mathrm{Log}_{10}(x+1)$ (R = 0.29).



Appendix 32: Standard lengths (mm) of different fish species collected across all pumping samples.

	No. of individuals	**Min (mm)**	**Max (mm)**	**Mean (mm)**	*s.e.*
R.. semoni	13	19	37	25.85	1.68
N. erebi	304	25	207	79.89	2.87
M. ambigua	231	5	215	39.29	1.61
L. unicolor	97	29	116	48.54	2.27
T. tandanus	1			68	
G. holbrooki	11	17	31	21.27	1.45
C. carpio	3	53	120	76.67	21.70
C. auratus	2	60	93	76.50	16.50

Appendix 33: Macroinvertebrate species and total abundance from extracted water between February and March 2006.

		Newinga	**Jericho**
CRUSTACEA			
DECAPODA			
Aytidae	*Parataya australiensis*		2
Palaemonidae	*Machrobrachium australiense*	1	5
ISOPODA	Indeterminate sp.		4
INSECTA			
DIPTERA			
Chaoboridae	Indeterminate sp.	4	26
Chironomidae			
Chironominae	*Cladopema* sp.	1	6
	Cryptochironomus sp.		1
	Dicrotendipes sp.	1	5
	Parachironomus sp.		2
	Polypedilum sp.		2
Tanypodinae	*Coelopynia* sp.	1	1
	Procladius sp.		56
Chironomidae	Tiny chironomidae		8
	Pupae	1	2
Ceratopogonidae	Tiny Ceratopogonidae	1	
Simulidae	Indeterminate sp.	4	
EPHEMEROPTERA			
Baetidae	*Cloeon* sp.	1	
HEMIPTERA			
Notonectidae	*Anisops* sp.	1	
Corixidae	*Micronecta* sp.		1
ODONATA	Tiny Odonata	2	
GASTROPODA			
BASOMMATOPHORA			
Ancylidae	*Ferrissia* sp.	1	
NEMATODA	Indeterminate sp.	184	4
BRYOZOA	Indeterminate sp.	4	

Appendix 34: Minimum and maximum daily temperatures recorded from two gauges in the greenhouse between Day 0 (24th March 2007) and Day 72 (4th June 2007).

	G1		**G2**	
	Minimum daily temperature	**Maximum daily temperature**	**Minimum daily temperature**	**Maximum daily temperature**
Day 0	18	>50	18	44
Day 1	22	48	21	44
Day 2	14	48	16	40
Day 3	15	>50	16	44
Day 4	16	50	18	46
Day 5	19	>50	19	48
Day 6	10	>50	12	44
Day 7	12	>50	14	44
Day 8	14	>50	15	46
Day 9	17	>50	18	47
Day 10	16	>50	17	45
Day 11	16	>50	18	46
Day 12	15	>50	17	44
Day 13	15	>50	17	44
Day 14	12	>50	14	42
Day 15	12	47	14	44
Day 16	12	47	14	43
Day 17	11	46	13	43
Day 18	12	48	13	43
Day 19	12	47	14	43
Day 20	14	>50	15	43
Day 21	14	>50	15	44
Day 22	14	>50	15	42
Day 23	14	>50	16	46
Day 24	12	>50	14	45
Day 25	12	>50	13	45
Day 26	12	49	14	46
Day 27	14	50	15	41
Day 28	12	>50	14	46
Day 72	2	35	3	32

Appendix 35: Water quality data for controls (C), floodplain samples (F) and typical storage samples (TS) from the mesocosm sampled between 24th March 2007 and 5th June 2007 (1 = Day 2, 2 = Day 7, 3 = Day 10, 4 = Day 14, 5 = Day 17, 6 = Day 21, 7 = Day 24, 8 = Day 28 and 9 = Day 73)

Time	Replicate	pH	Conductivity (mS cm^{-1})	Dissolved oxygen (ppm)	Water temperature (^{o}C)	Salinity (%)
1	C1	8.96	0.240	7.17	25.0	0.00
1	C2	9.02	0.252	7.11	25.8	0.01
1	C3	9.16	0.261	5.84	36.4	0.01
1	F1 1	8.60	0.258	5.45	40.9	0.01
1	F1 2	8.79	0.238	6.67	36.9	0.00
1	F1 3	8.30	0.227	6.36	35.3	0.00
1	F2 1	9.06	0.238	7.21	29.9	0.00
1	F2 2	8.95	0.270	6.08	38.5	0.01
1	F2 3	8.77	0.241	7.10	28.5	0.00
1	F3 1	8.24	0.279	4.60	41.5	0.01
1	F3 2	8.25	0.244	7.47	28.8	0.00
1	F3 3	8.54	0.232	7.12	28.0	0.00
1	F4 1	9.08	0.280	7.44	24.6	0.01
1	F4 2	8.82	0.313	6.71	27.7	0.01
1	F4 3	8.87	0.286	6.51	37.5	0.01
1	F5 1	9.32	0.338	6.28	34.8	0.01
1	F5 2	9.01	0.297	5.41	38.9	0.01
1	F5 3	9.08	0.270	7.20	30.8	0.01
1	TS10 1	9.09	0.289	6.48	26.4	0.01
1	TS10 2	9.12	0.288	6.81	29.5	0.01
1	TS10 3	9.25	0.293	5.55	39.5	0.01
1	TS11 1	8.63	0.315	6.96	28.4	0.01
1	TS11 2	8.81	0.324	6.57	33.4	0.01
1	TS11 3	9.15	0.338	6.92	38.5	0.01
1	TS12 1	9.36	0.306	6.41	34.5	0.01
1	TS12 2	9.11	0.286	6.99	27.7	0.01
1	TS12 3	8.84	0.325	5.24	41.3	0.01
1	TS13 1	9.13	0.253	6.77	31.6	0.00
1	TS13 2	9.06	0.270	6.92	24.7	0.01
1	TS13 3	8.62	0.250	5.40	37.4	0.01
1	TS14 1	8.84	0.312	6.63	28.1	0.01
1	TS14 2	9.23	0.266	7.20	26.2	0.01
1	TS14 3	9.21	0.299	6.74	30.0	0.01
2	C1	8.43	0.261	8.70	18.4	0.01
2	C2	8.41	0.281	8.18	22.0	0.01
2	C3	8.57	0.309	7.19	29.8	0.01
2	F1 1	8.75	0.271	7.27	34.7	0.01
2	F1 2	8.10	0.283	7.87	27.9	0.01
2	F1 3	8.38	0.262	7.76	29.6	0.01
2	F2 1	8.15	0.267	9.09	18.4	0.01
2	F2 2	8.37	0.339	7.59	28.0	0.01
2	F2 3	7.73	0.297	9.08	18.6	0.01
2	F3 1	8.54	0.414	5.65	35.4	0.01
2	F3 2	7.48	0.321	9.28	19.9	0.01
2	F3 3	8.54	0.288	9.07	21.7	0.01
2	F4 1	8.44	0.452	9.30	14.9	0.01
2	F4 2	8.27	0.381	8.06	27.4	0.01
2	F4 3	8.60	0.371	7.90	32.8	0.01
2	F5 1	8.59	0.454	7.61	27.8	0.01

Time	Replicate	pH	Conductivity (mS cm^{-1})	Dissolved oxygen (ppm)	Water temperature (^{o}C)	Salinity (%)
2	F5 2	8.63	0.368	7.20	32.4	0.01
2	F5 3	8.88	0.318	8.93	24.1	0.01
2	TS10 1	7.57	0.396	9.30	15.7	0.01
2	TS10 2	8.30	0.371	8.68	18.1	0.01
2	TS10 3	8.52	0.408	7.10	31.2	0.01
2	TS11 1	8.13	0.397	9.68	21.2	0.01
2	TS11 2	8.80	0.380	9.49	26.3	0.01
2	TS11 3	8.50	0.400	8.33	30.0	0.01
2	TS12 1	8.45	0.434	7.02	27.7	0.01
2	TS12 2	7.95	0.388	9.08	16.9	0.01
2	TS12 3	8.74	0.431	7.32	34.5	0.01
2	TS13 1	8.23	0.303	8.21	21.4	0.01
2	TS13 2	8.60	0.378	9.12	18.8	0.01
2	TS13 3	8.13	0.306	7.27	27.2	0.01
2	TS14 1	8.38	0.475	7.99	22.9	0.01
2	TS14 2	8.52	0.337	8.84	21.3	0.01
2	TS14 3	8.60	0.386	8.46	21.7	0.01
3	C1	7.83	0.267	8.05	24.8	0.01
3	C2	8.55	0.298	7.03	32.1	0.01
3	C3	8.47	0.323	8.03	24.1	0.01
3	F1 1	8.28	0.254	8.49	25.7	0.00
3	F1 2	8.61	0.283	7.44	35.9	0.01
3	F1 3	8.26	0.269	9.64	25.8	0.01
3	F2 1	8.08	0.275	8.85	18.9	0.01
3	F2 2	8.87	0.326	8.16	36.1	0.01
3	F2 3	8.50	0.305	9.61	28.3	0.01
3	F3 1	8.39	0.417	9.04	26.0	0.01
3	F3 2	8.61	0.329	11.10	31.5	0.01
3	F3 3	8.41	0.315	10.49	22.2	0.01
3	F4 1	8.42	0.404	8.58	22.7	0.01
3	F4 2	8.64	0.385	7.52	36.9	0.01
3	F4 3	8.32	0.389	8.83	25.9	0.01
3	F5 1	8.79	0.427	7.24	35.2	0.01
3	F5 2	8.01	0.367	8.46	23.5	0.01
3	F5 3	8.69	0.319	10.71	29.4	0.01
3	TS10 1	8.64	0.379	10.87	23.7	0.01
3	TS10 2	8.52	0.389	8.56	18.8	0.01
3	TS10 3	7.95	0.406	8.41	22.4	0.01
3	TS11 1	8.87	0.409	10.09	32.6	0.01
3	TS11 2	8.95	0.403	8.29	30.8	0.01
3	TS11 3	7.72	0.448	8.38	21.4	0.01
3	TS12 1	8.02	0.432	8.94	20.0	0.01
3	TS12 2	8.70	0.406	9.65	24.8	0.01
3	TS12 3	8.44	0.426	9.37	25.7	0.01
3	TS13 1	7.89	0.309	8.66	19.2	0.01
3	TS13 2	8.16	0.397	8.79	25.7	0.01
3	TS13 3	8.44	0.316	6.82	35.1	0.01
3	TS14 1	8.67	0.462	8.34	33.5	0.01
3	TS14 2	8.66	0.354	10.06	22.1	0.01
3	TS14 3	8.75	0.403	8.24	26.9	0.01
4	C1	8.49	0.283	8.67	20.7	0.01

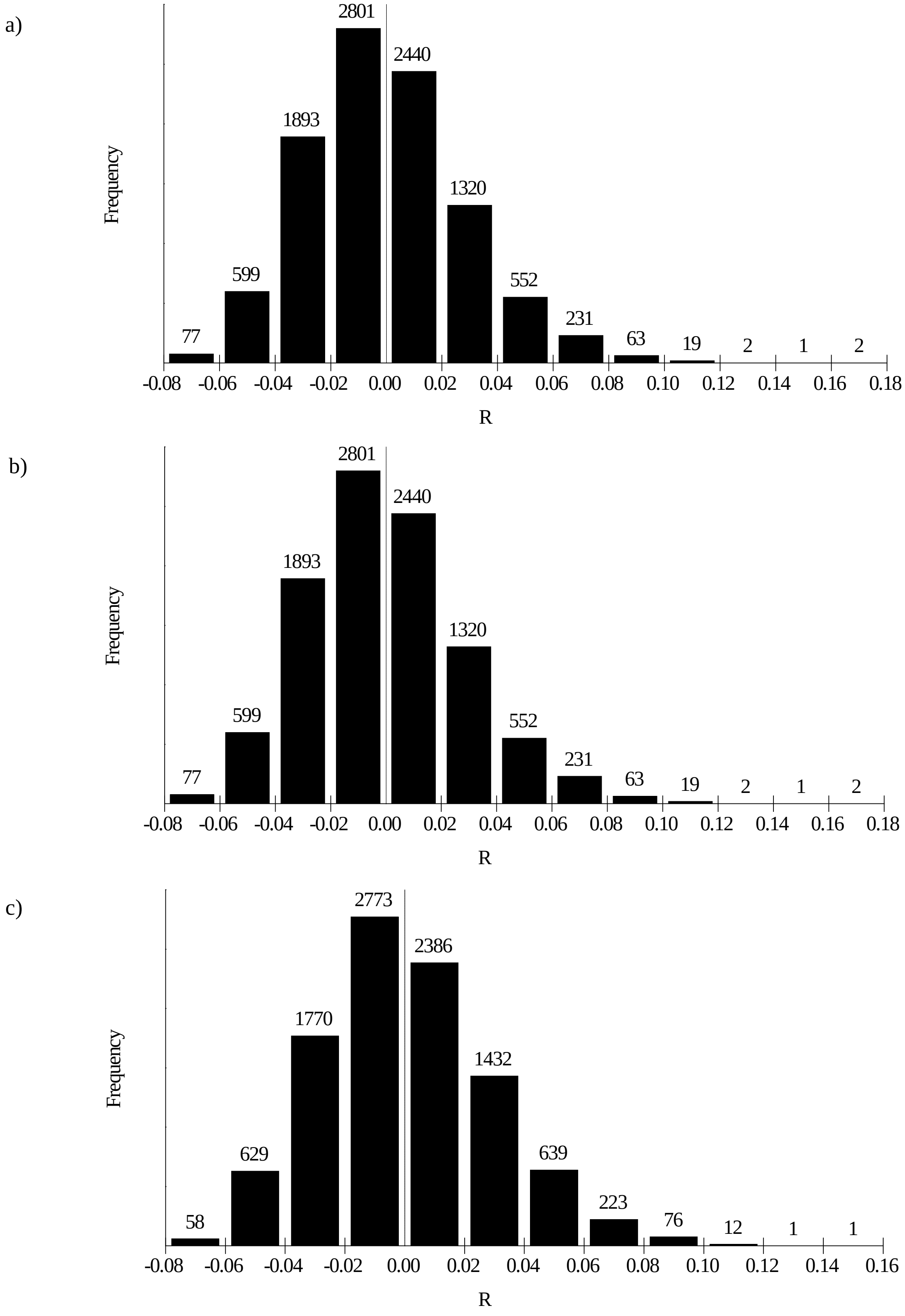
Time	Replicate	pH	Conductivity (mS cm^{-1})	Dissolved oxygen (ppm)	Water temperature (^{o}C)	Salinity (%)
4	C2	8.50	0.326	8.12	23.8	0.01
4	C3	8.53	0.361	7.46	28.4	0.01
4	F1 1	9.28	0.236	7.86	34.6	0.00
4	F1 2	8.47	0.283	8.70	29.1	0.01
4	F1 3	8.52	0.264	9.67	25.2	0.01
4	F2 1	8.96	0.270	10.57	23.3	0.01
4	F2 2	8.62	0.346	8.77	29.1	0.01
4	F2 3	8.42	0.326	9.80	21.4	0.01
4	F3 1	8.89	0.425	11.35	33.5	0.01
4	F3 2	8.40	0.360	10.50	23.2	0.01
4	F3 3	8.80	0.335	9.57	21.8	0.01
4	F4 1	7.93	0.487	8.86	19.0	0.01
4	F4 2	8.53	0.412	8.37	28.3	0.01
4	F4 3	8.56	0.404	8.47	29.5	0.01
4	F5 1	8.72	0.423	8.86	28.7	0.01
4	F5 2	8.97	0.375	8.26	33.0	0.01
4	F5 3	8.75	0.338	9.29	22.6	0.01
4	TS10 1	8.35	0.356	9.36	19.5	0.01
4	TS10 2	8.61	0.400	9.20	23.3	0.01
4	TS10 3	8.69	0.408	8.31	32.5	0.01
4	TS11 1	8.78	0.439	12.03	24.4	0.01
4	TS11 2	8.92	0.430	10.66	25.0	0.01
4	TS11 3	9.19	0.475	9.88	31.0	0.01
4	TS12 1	9.25	0.477	9.54	26.1	0.01
4	TS12 2	8.77	0.457	9.44	20.0	0.01
4	TS12 3	9.05	0.394	11.83	34.5	0.01
4	TS13 1	8.51	0.333	8.50	23.7	0.01
4	TS13 2	8.74	0.420	9.05	21.4	0.01
4	TS13 3	8.32	0.337	7.96	28.0	0.01
4	TS14 1	8.69	0.478	10.02	25.2	0.01
4	TS14 2	8.83	0.382	9.29	21.4	0.01
4	TS14 3	8.89	0.398	10.46	22.8	0.01
5	C1	8.51	0.288	8.80	19.8	0.01
5	C2	8.48	0.337	8.69	20.7	0.01
5	C3	8.59	0.365	7.24	29.2	0.01
5	F1 1	8.91	0.250	7.37	35.9	0.01
5	F1 2	8.41	0.277	9.21	26.0	0.01
5	F1 3	8.72	0.264	9.76	24.0	0.01
5	F2 1	9.08	0.266	10.80	22.8	0.01
5	F2 2	8.48	0.335	9.26	25.4	0.01
5	F2 3	8.49	0.323	9.88	17.2	0.01
5	F3 1	9.19	0.382	11.63	35.1	0.01
5	F3 2	8.50	0.352	10.79	19.0	0.01
5	F3 3	8.86	0.335	9.99	20.8	0.01
5	F4 1	8.59	0.424	9.58	15.6	0.01
5	F4 2	8.43	0.412	8.87	24.9	0.01
5	F4 3	8.75	0.409	8.38	28.3	0.01
5	F5 1	8.68	0.402	9.83	25.0	0.01
5	F5 2	9.01	0.370	8.30	34.3	0.01
5	F5 3	8.91	0.337	9.74	22.0	0.01
5	TS10 1	8.44	0.367	9.49	16.1	0.01

Time	Replicate	pH	Conductivity (mS cm^{-1})	Dissolved oxygen (ppm)	Water temperature (^{o}C)	Salinity (%)
5	TS10 2	8.92	0.369	9.68	23.5	0.01
5	TS10 3	8.76	0.397	8.46	32.7	0.01
5	TS11 1	8.74	0.460	10.31	20.6	0.01
5	TS11 2	8.86	0.450	10.49	23.7	0.01
5	TS11 3	9.40	0.459	10.33	29.4	0.01
5	TS12 1	9.05	0.495	9.04	25.1	0.02
5	TS12 2	8.75	0.456	10.18	16.6	0.01
5	TS12 3	9.16	0.365	12.66	35.6	0.01
5	TS13 1	8.63	0.333	8.64	23.3	0.01
5	TS13 2	8.78	0.420	9.01	20.6	0.01
5	TS13 3	8.28	0.344	8.48	24.9	0.01
5	TS14 1	8.68	0.460	10.20	21.6	0.01
5	TS14 2	8.95	0.365	9.80	20.4	0.01
5	TS14 3	8.95	0.384	10.49	21.9	0.01
6	C1	8.25	0.301	8.61	18.7	0.01
6	C2	8.24	0.360	8.55	19.1	0.01
6	C3	8.44	0.397	7.64	25.2	0.01
6	F1 1	8.58	0.270	7.29	31.4	0.01
6	F1 2	8.25	0.269	9.13	23.5	0.01
6	F1 3	8.56	0.259	10.25	21.3	0.01
6	F2 1	8.79	0.269	10.32	21.7	0.01
6	F2 2	8.34	0.343	9.19	23.2	0.01
6	F2 3	8.33	0.316	9.41	17.8	0.01
6	F3 1	9.02	0.342	10.92	29.5	0.01
6	F3 2	8.18	0.328	10.07	18.5	0.01
6	F3 3	8.59	0.315	10.18	19.9	0.01
6	F4 1	8.34	0.404	9.13	16.9	0.01
6	F4 2	8.43	0.398	9.24	21.9	0.01
6	F4 3	8.43	0.403	9.05	24.1	0.01
6	F5 1	8.46	0.396	9.88	21.9	0.01
6	F5 2	8.71	0.379	8.09	30.4	0.01
6	F5 3	8.59	0.341	9.50	20.3	0.01
6	TS10 1	8.27	0.366	9.04	17.2	0.01
6	TS10 2	8.64	0.337	9.55	22.0	0.01
6	TS10 3	8.57	0.375	8.74	29.1	0.01
6	TS11 1	8.44	0.480	9.88	19.2	0.01
6	TS11 2	8.59	0.468	10.01	20.6	0.01
6	TS11 3	9.11	0.456	11.08	25.7	0.01
6	TS12 1	8.86	0.523	8.96	22.5	0.02
6	TS12 2	8.60	0.475	9.36	17.4	0.01
6	TS12 3	8.88	0.362	10.70	30.1	0.01
6	TS13 1	8.35	0.342	8.52	21.5	0.01
6	TS13 2	8.55	0.420	9.25	19.4	0.01
6	TS13 3	8.10	0.362	8.31	23.0	0.01
6	TS14 1	8.38	0.456	9.45	19.7	0.01
6	TS14 2	8.71	0.351	9.97	19.5	0.01
6	TS14 3	8.60	0.395	9.79	20.0	0.01
7	C1	8.32	0.306	8.43	19.8	0.01
7	C2	8.25	0.367	8.14	21.1	0.01
7	C3	8.50	0.426	7.61	26.1	0.01
7	F1 1	8.74	0.271	7.10	33.2	0.01

Time	Replicate	pH	Conductivity (mS cm^{-1})	Dissolved oxygen (ppm)	Water temperature (^{o}C)	Salinity (%)
7	F1 2	8.38	0.261	9.03	24.8	0.01
7	F1 3	8.80	0.253	9.99	22.5	0.00
7	F2 1	8.80	0.273	9.44	23.6	0.01
7	F2 2	8.34	0.337	9.05	25.2	0.01
7	F2 3	8.30	0.312	9.12	19.0	0.01
7	F3 1	9.24	0.347	11.53	31.2	0.01
7	F3 2	8.34	0.321	10.13	19.8	0.01
7	F3 3	8.82	0.299	10.02	21.2	0.01
7	F4 1	8.36	0.384	9.06	18.1	0.01
7	F4 2	8.40	0.382	9.18	24.5	0.01
7	F4 3	8.51	0.408	9.38	24.3	0.01
7	F5 1	8.56	0.388	9.79	22.4	0.01
7	F5 2	8.97	0.390	7.97	32.2	0.01
7	F5 3	8.73	0.337	9.53	21.7	0.01
7	TS10 1	8.42	0.357	8.94	18.4	0.01
7	TS10 2	8.84	0.332	9.16	24.5	0.01
7	TS10 3	8.70	0.366	8.15	29.7	0.01
7	TS11 1	8.51	0.456	10.28	21.4	0.01
7	TS11 2	8.80	0.466	9.73	21.4	0.01
7	TS11 3	9.06	0.479	10.08	25.7	0.01
7	TS12 1	8.94	0.541	9.37	24.1	0.02
7	TS12 2	8.55	0.492	9.15	18.7	0.02
7	TS12 3	8.90	0.356	11.18	32.5	0.01
7	TS13 1	8.47	0.342	8.20	23.3	0.01
7	TS13 2	8.67	0.410	9.20	21.0	0.01
7	TS13 3	8.16	0.364	8.08	24.8	0.01
7	TS14 1	8.50	0.437	9.71	22.1	0.01
7	TS14 2	8.86	0.338	9.46	21.2	0.01
7	TS14 3	8.62	0.410	9.40	20.9	0.01
8	C1	8.53	0.321	8.68	17.1	0.01
8	C2	8.49	0.392	8.47	19.1	0.01
8	C3	8.58	0.421	7.98	22.3	0.01
8	F1 1	8.80	0.298	7.23	31.0	0.01
8	F1 2	8.62	0.260	9.48	20.5	0.01
8	F1 3	8.86	0.250	10.03	19.6	0.00
8	F2 1	8.72	0.282	10.09	21.7	0.01
8	F2 2	8.49	0.361	8.46	22.9	0.01
8	F2 3	8.33	0.313	9.12	16.6	0.01
8	F3 1	9.23	0.332	12.07	28.4	0.01
8	F3 2	8.44	0.309	9.99	17.2	0.01
8	F3 3	9.03	0.283	12.12	18.7	0.01
8	F4 1	8.02	0.392	9.02	15.8	0.01
8	F4 2	8.54	0.390	8.72	22.5	0.01
8	F4 3	8.93	0.389	10.64	20.7	0.01
8	F5 1	8.73	0.388	9.74	19.3	0.01
8	F5 2	8.92	0.374	7.97	28.9	0.01
8	F5 3	8.92	0.338	9.69	18.9	0.01
8	TS10 1	7.94	0.353	9.05	16.1	0.01
8	TS10 2	8.59	0.334	9.87	22.0	0.01
8	TS10 3	8.84	0.346	8.77	25.2	0.01
8	TS11 1	8.59	0.442	10.63	18.2	0.01
8	TS11 2	8.82	0.458	10.27	18.7	0.01

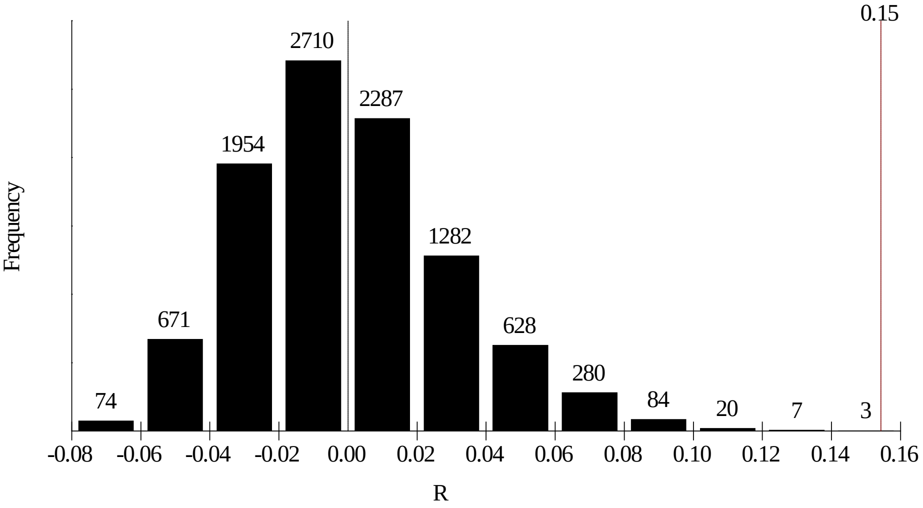
Time	Replicate	pH	Conductivity (mS cm^{-1})	Dissolved oxygen (ppm)	Water temperature (^{o}C)	Salinity (%)
8	TS11 3	9.03	0.479	10.54	22.0	0.01
8	TS12 1	9.00	0.528	9.92	21.0	0.02
8	TS12 2	8.41	0.503	9.15	16.3	0.02
8	TS12 3	8.80	0.401	7.52	29.3	0.01
8	TS13 1	8.50	0.355	8.55	20.6	0.01
8	TS13 2	8.88	0.394	9.45	18.3	0.01
8	TS13 3	8.31	0.379	8.51	22.7	0.01
8	TS14 1	8.55	0.423	9.93	19.7	0.01
8	TS14 2	8.77	0.352	10.12	18.5	0.01
8	TS14 3	8.76	0.423	9.41	18.3	0.01
9	C1	8.55	0.343	10.53	13.1	0.01
9	C2	8.44	0.409	11.07	13.4	0.01
9	C3	8.74	0.435	10.15	15.6	0.01
9	F1 1	8.82	0.287	11.89	17.9	0.01
9	F1 2	8.99	0.216	11.72	13.6	0.00
9	F1 3	8.62	0.245	11.44	14.8	0.00
9	F2 1	8.66	0.291	10.41	16.5	0.01
9	F2 2	8.63	0.352	11.17	14.9	0.01
9	F2 3	8.27	0.298	10.85	11.6	0.01
9	F3 1	8.84	0.358	11.75	17.3	0.01
9	F3 2	8.33	0.306	10.60	11.8	0.01
9	F3 3	8.74	0.311	10.76	13.8	0.01
9	F4 1	8.31	0.431	10.62	10.7	0.01
9	F4 2	8.51	0.422	10.15	15.3	0.01
9	F4 3	8.72	0.372	11.10	15.6	0.01
9	F5 1	8.72	0.402	11.63	13.4	0.01
9	F5 2	8.87	0.393	10.72	16.7	0.01
9	F5 3	8.95	0.341	11.86	14.3	0.01
9	TS10 1	8.27	0.370	10.65	11.0	0.01
9	TS10 2	8.60	0.428	9.71	16.8	0.01
9	TS10 3	8.60	0.414	10.60	16.6	0.01
9	TS11 1	8.53	0.461	11.98	12.5	0.01
9	TS11 2	9.40	0.382	13.08	13.4	0.01
9	TS11 3	9.04	0.467	12.98	17.1	0.01
9	TS12 1	8.83	0.550	9.75	16.3	0.02
9	TS12 2	8.73	0.482	11.81	11.3	0.01
9	TS12 3	8.68	0.457	9.38	18.4	0.01
9	TS13 1	8.62	0.358	10.24	16.2	0.01
9	TS13 2	8.63	0.402	10.33	13.7	0.01
9	TS13 3	8.44	0.363	10.78	13.7	0.01
9	TS14 1	8.56	0.458	11.26	13.1	0.01
9	TS14 2	8.68	0.393	10.77	13.7	0.01
9	TS14 3	8.84	0.410	11.75	13.3	0.01

Appendix 36: Histograms for ANOSIM results of zooplankton egg bank assemblage structure in floodplain pots based on a) presence/absence data (R = 0.28), b) absolute abundance $Log_{10}(x+1)$ (R = 0.29) and c) relative abundance $Log_{10}(x+1)$ (R = 0.29) across all sampling occasions.

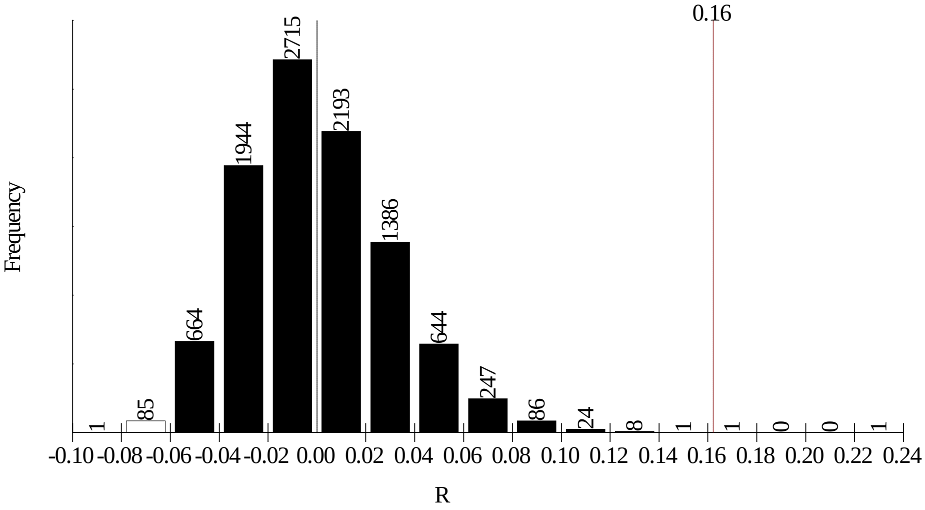


Appendix 37: Histograms for ANOSIM results of zooplankton egg bank assemblage structure in typical storage pots based on a) presence/absence data (R = 0.15), b) absolute abundance $\mathrm{Log}_{10}(x+1)$ (R = 0.16) and c) relative abundance $\mathrm{Log}_{10}(x+1)$ (R = 0.15) across all sampling occasions.

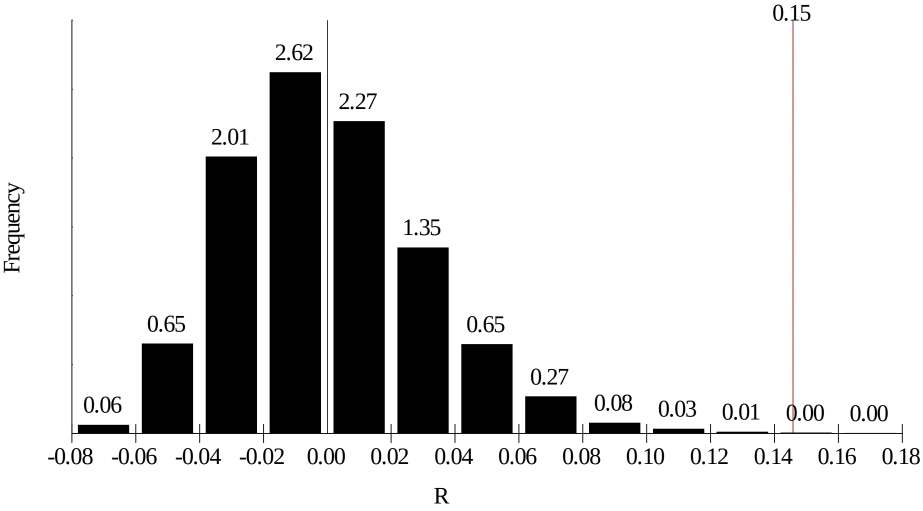
a)



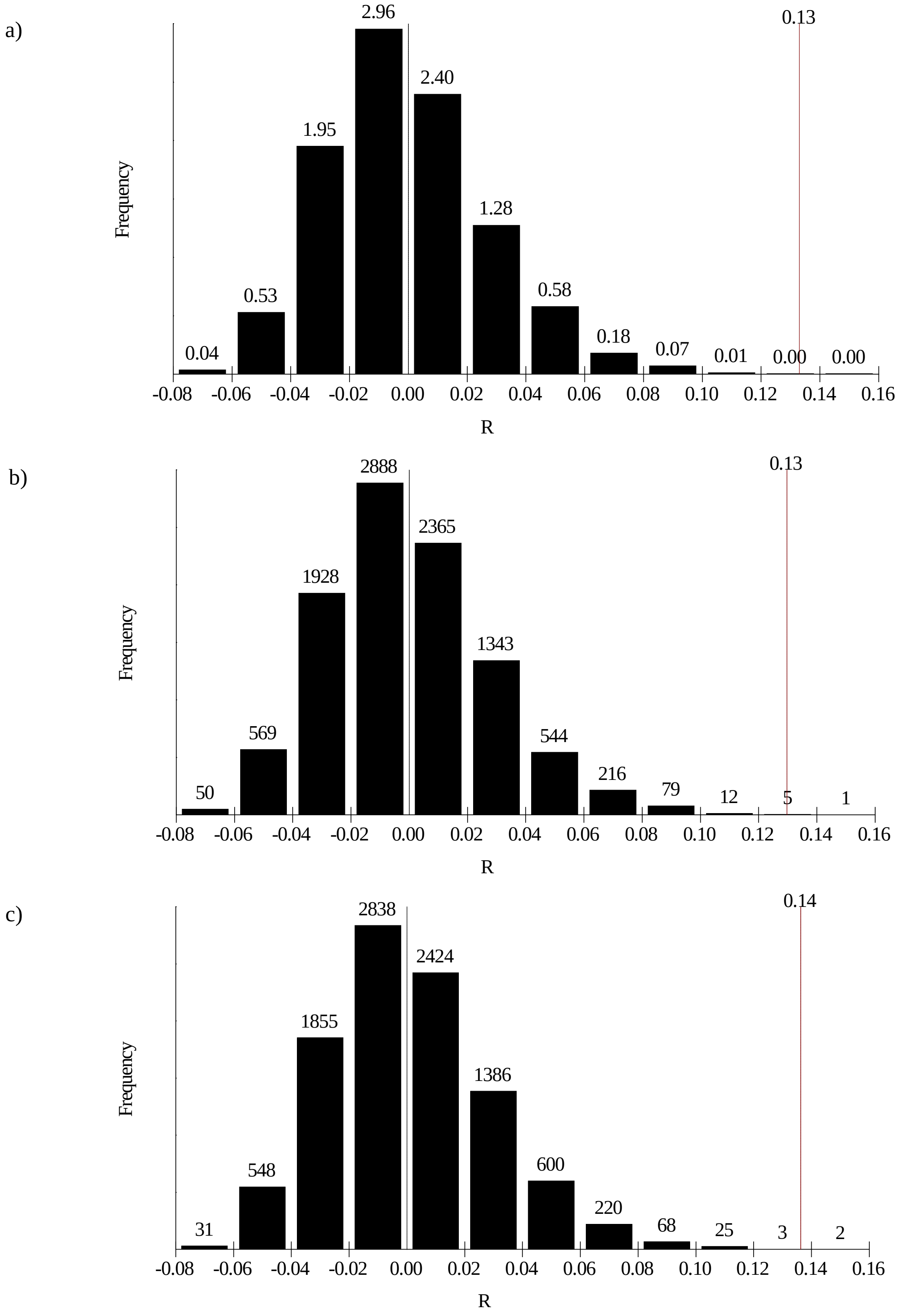
b)



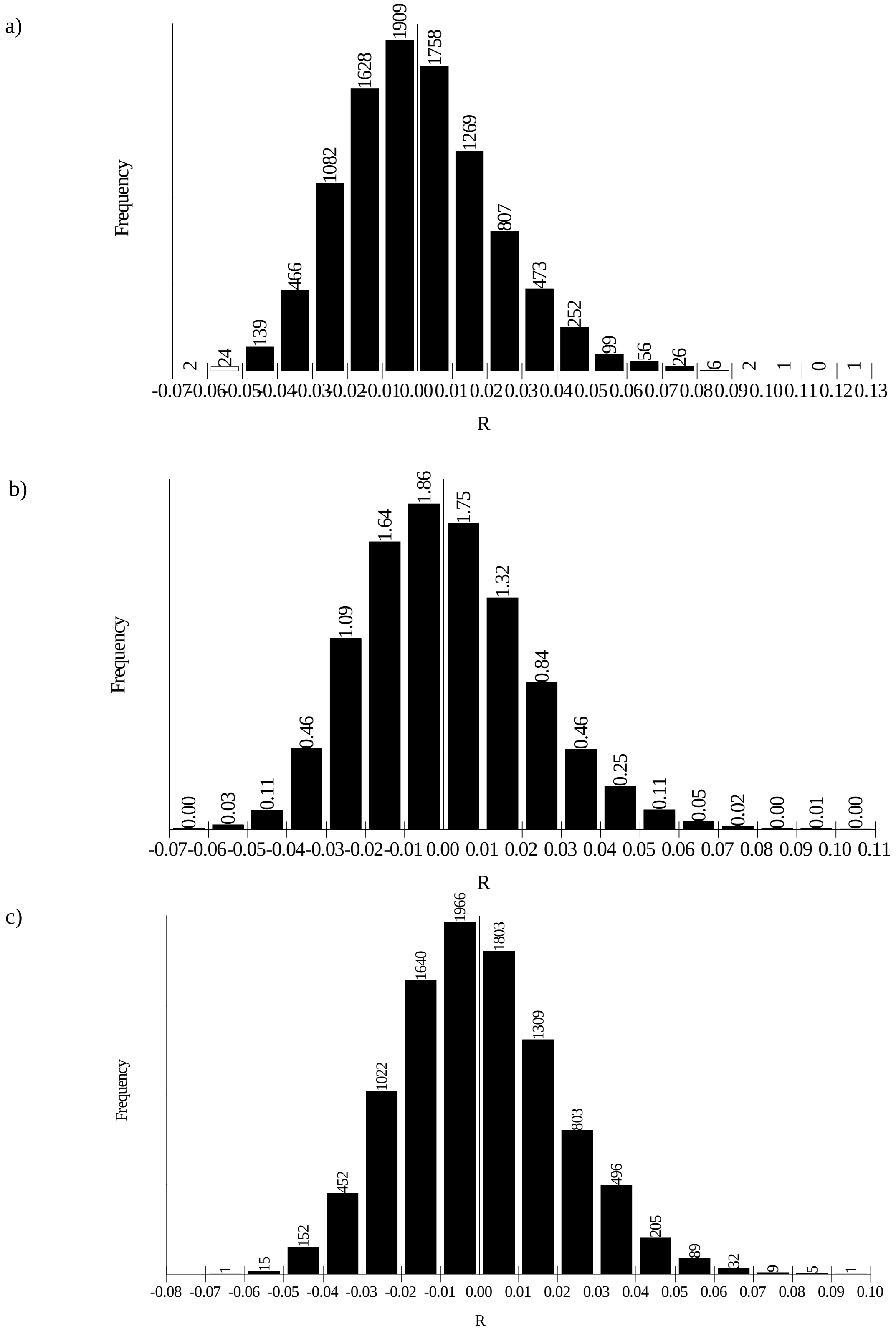
c)



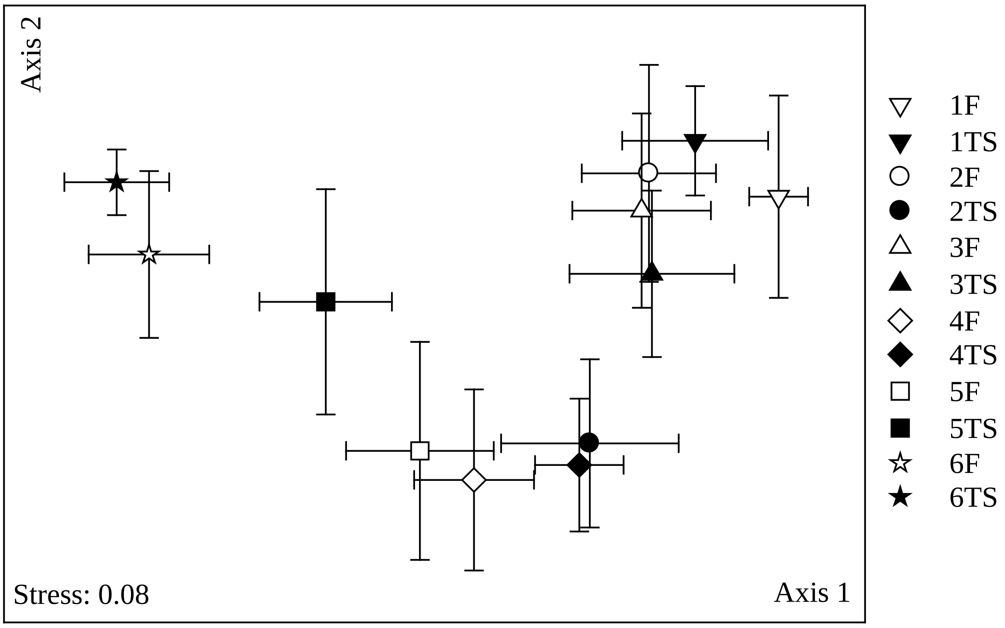
Appendix 38: Histograms for ANOSIM results comparing zooplankton egg bank assemblage structure between typical storage pots and floodplain pots based on a) presence/absence data (R = 0.13), b) absolute abundance $Log_{10}(x+1)$ (R = 0.13) and c) relative abundance $Log_{10}(x+1)$ (R = 0.14) across all sampling occasions.



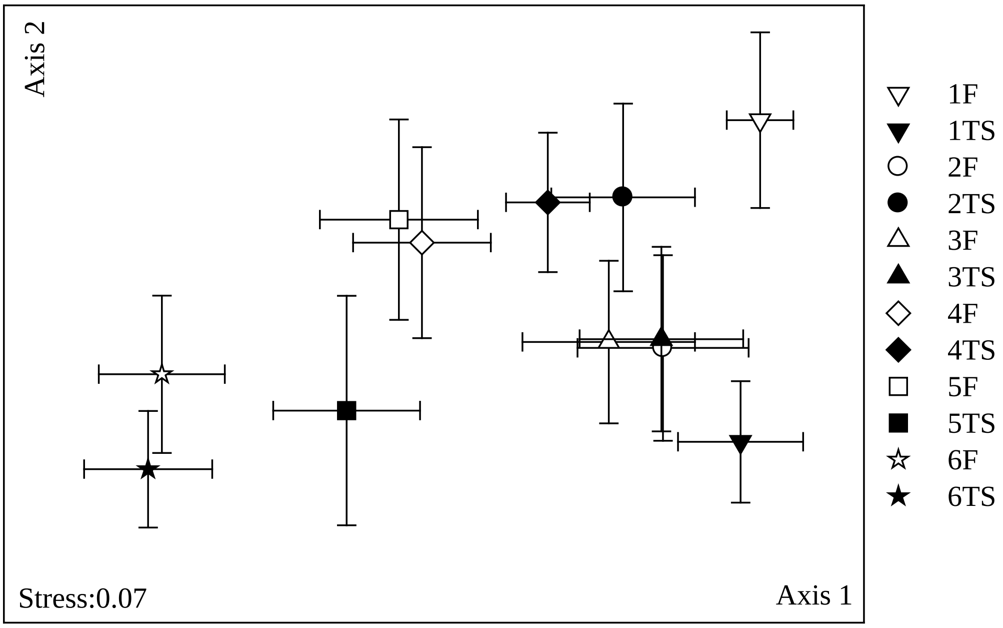
Appendix 39: Histograms for ANOSIM results comparing zooplankton egg bank assemblage structure between sampling occasions based on a) presence/absence data (R = 0.21), b) absolute abundance $\mathrm{Log}_{10}(x+1)$ (R = 0.22) and c) relative abundance $\mathrm{Log}_{10}(x+1)$ (R = 0.21).



Appendix 40: Non-metric multidimensional scaling plot of zooplankton assemblages emerging from soil collected from floodplain sites (F) and typical storage sites (TS) during the egg bank study. 1= Day 2, 2 = Day 7, 3 = Day 10 & 14, 4 = Day 17 & 21, 5 = Day 24 & 28, 6 = Day 75. Samples collected from each sites and time are represented as centroids (average x-y coordinates ± s.e.) Plot a) is based on log(*x*+1) transformed data standardised for relative abundance and b) is based on Presence/Absence data.



a)



b)

Appendix 41: Mean δ^{15}N (‰) of basal sources and consumers for samples collected in April 2008 (standard errors, and number of replicates are given in parentheses).

	NW4	**NW7**	**NW9**	**NW10**	**NW11**	**TS1**	**TS6**	**TS7**	**TS15**	**TS16**
Basal Sources										
Seston	-0.6 (0.4, 3)	2.7 (0.2, 3)	3.1 (0.7, 3)	1.3 (0.5, 3)	2.2 (1.3, 3)	0.5 (1.8, 3)	-0.6 (1.7, 3)	-2.6 (0.5, 3)	4.7 (0.8, 3)	2.1 (0.3, 3)
FPOM	4.8 (0.5, 3)	7.5 (0.3, 3)	1.9 (0.3, 3)	5.7 (0.1, 3)	5.4 (0.2, 3)	6.4 (0.8, 3)	7.0 (0.4, 3)	7.7 (0.5, 3)	7.1 (0.0, 3)	8.9 (0.3, 3)
CPOM	6.0 (0.2, 3)	4.2 (0.3, 3)	1.0 (0.6, 3)	4.7 (0.2, 3)	6.3 (0.4, 3)	7.6 (1.0, 3)	3.7 (0.3, 3)	5.0 (1.7, 3)	6.6 (0.7, 3)	8.2 (0.5, 3)
Biofilm	5.2 (0.7, 2)	5.7 (0.8, 3)		9.6 (0.6, 3)	10.5 (0.4, 2)	8.4 (0.2, 3)	6.1 (0.5, 3)	7.9 (0.4, 2)		5.8 (1.6, 3)
Floating algae		8.1 (-, 1)			11.9 (-, 1)				10.4 (0.1, 3)	
Algal Clump									8.8 (2.5, 3)	
Microinvertebrate Consumers										
Zooplankton	11.1 (0.1, 3)	11.7 (0.1, 3)	9.5 (0.6, 3)	10.8 (0.4, 3)	14.6 (0.2, 3)	15.3 (0.1, 3)	12.4 (1.0, 3)	13.4 (0.2, 3)	14.5 (0.2, 3)	13.5 (0.6, 3)
Primary Macroinvertebrate Consumers										
Chironomids				9.0 (0.4, 3)					12.0 (0.4, 3)	
Corixidae (Adults)	5.7 (-, 1)	9.3 (0.6, 2)	7.6 (1.3, 2)	8.0 (0.4, 3)	12.5 (-, 1)	7.4 (-, 1)		9.4 (2.4, 2)	9.1 (-, 1)	10.1 (1.8, 2)
Corixidae (Nymphs)		8.5 (-, 1)	8.2 (0.8, 2)	7.5 (1.1, 2)						11.8 (-, 1)
Physidae			6.1 (0.4, 3)	5.6 (0.6, 2)				10.5 (0.3, 3)		
Caenidae	8.5 (-, 1)			7.1 (-, 1)						
Baetidae			8.3 (0.1, 3)	8.9 (1.4, 2)	13.4 (-, 1)				12.5 (0.6, 3)	

	NW4	**NW7**	**NW9**	**NW10**	**NW11**	**TS1**	**TS6**	**TS7**	**TS15**	**TS16**
Trichoptera			7.9 (1.1, 3)	8.3 (0.9, 2)			7.6 (-, 1)	9.0 (1.1, 2)	12.9 (0.3, 3)	
Bivalvia		9.6 (0.1, 5)		10.9 (0.2, 6)						
Secondary Macroinvertebrate Consumers										
Notonectidae (Adults)	10.0 (0.3, 3)	9.4 (0.9, 3)	10.0 (0.3, 2)	8.9 (0.6, 3)	10.9 (0.1, 3)		9.8 (0.3, 3)	11.6 (0.6, 3)	12.3 (0.0, 3)	7.7 (1.4, 2)
Notonectidae (Nymphs)			10.0 (1.0, 2)	9.6 (0.6, 2)	14.1 (-, 1)		13.9 (0.8, 2)	14.9 (0.4, 2)	14.1 (0.3, 3)	15.3 (0.4, 2)
Nepidae					10.2 (0.6, 2)	9.9 (-, 1)	7.9 (0.3, 2)	9.7 (0.6, 3)	11.3 (-, 1)	8.5 (0.6, 3)
Odonata										
Epiproctophora	9.7 (-, 1)	12.3 (0.8, 2)	7.9 (0.4, 3)	8.1 (-, 1)	13.6 (-, 1)	14.1 (0.2, 3)	11.7 (-, 1)	10.5 (0.1, 3)	11.0 (0.3, 2)	
Zygoptera			10.8 (0.2, 3)	10.1 (0.4, 2)				12.7 (0.7, 3)	12.8 (-, 1)	
Naucoridae	6.4 (-, 1)									
Tanypodinae	10.0 (-, 1)	10.7 (-, 1)	9.9 (-, 1)	11.1 (-, 1)	9.2 (-, 1)				14.0 (-, 1)	
Spencerhydrus sp.		8.2 (-, 1)		8.5 (0.4, 3)	9.0 (-, 1)	10.5 (0.7, 3)	9.3 (0.9, 3)		8.9 (0.4, 3)	11.4 (-, 1)
Eretes sp.					10.2 (0.5, 2)			12.0 (0.2, 2)		
Hyphydrus sp.			9.2 (0.5, 3)					12.4 (-, 1)		
Macrobrachium sp. (Small)	12.7 (0.2, 5)	13.3 (0.1, 5)	11.9 (0.2, 5)	10.8 (0.2, 5)	12.6 (0.2, 5)	16.0 (0.3, 5)	14.5 (0.3, 5)	14.4 (0.4, 6)	15.6 (0.5, 4)	15.8 (0.4, 5)
Macrobrachium sp. (Large)	13.0 (0.3, 5)	12.5 (0.3, 5)		10.8 (0.3, 5)	12.5 (0.1, 5)	15.3 (0.8, 5)	13.6 (0.4, 5)		15.5 (0.5, 5)	15.8 (0.7, 5)
Cherax sp.	9.9 (0.3, 4)	9.5 (0.5, 4)	8.0 (0.3, 2)	9.1 (0.4, 5)	10.4 (0.3, 5)	14.2 (0.5, 5)	11.6 (0.2, 5)		13.2 (0.4, 6)	13.8 (0.5, 5)

	NW4	**NW7**	**NW9**	**NW10**	**NW11**	**TS1**	**TS6**	**TS7**	**TS15**	**TS16**
Acarina				9.3 (-, 1)					14.0 (0.9, 2)	
Fish										
L. unicolor				11.4 (0.3, 5)	13.5 (0.3, 5)	17.6 (0.4, 5)	13.9 (0.5, 5)	14.6 (0.5, 5)	13.7 (0.4, 6)	16.1 (1.1, 5)
N. erebi (small)	13.3 (0.3, 5)	14.1 (0.5, 5)	12.1 (0.3, 5)	11.7 (0.3, 5)	15.5 (-, 1)	16.6 (0.1, 2)	14.1 (0.5, 5)	17.0 (0.2, 3)		16.6 (0.7, 5)
N. erebi (medium)	12.3 (0.2, 5)	12.3 (0.4, 5)	12.1 (0.2, 3)	10.4 (0.2, 5)	13.0 (0.2, 5)		13.6 (0.5, 5)	13.8 (1.9, 2)		14.6 (0.3, 5)
N. erebi (large)	12.1 (0.5, 5)	12.2 (0.2, 5)		10.9 (0.2, 4)			12.8 (0.4, 5)	13.2 (1.3, 5)		16.8 (0.6, 4)
N. erebi (large) - bone	9.3 (1.6, 3)	11.0 (0.2, 5)		10.0 (0.3, 4)			12.5 (0.3, 5)			15.5 (0.4, 4)
C. carpio		11.4 (0.2, 5)		10.8 (0.3, 5)	13.1 (0.4, 5)	16.9 (0.2, 4)			13.3 (0.2, 5)	16.5 (0.4, 5)
C. carpio - bone				9.4 (0.6, 2)		15.6 (0.1, 3)				
M. ambigua	14.2 (0.3, 5)	14.5 (0.5, 5)	15.8 (-, 1)	11.9 (0.3, 5)	14.6 (0.2, 5)	15.8 (-, 1)	15.7 (0.2, 5)	15.1 (0.8, 5)	15.3 (0.1, 5)	17.6 (0.8, 5)
Hypseleotris spp.	14.8 (0.2, 3)	13.5 (0.2, 5)							15.1 (0.3, 3)	
R. semoni	14.0 (-, 1)	14.3 (0.2, 5)								
G. holbrooki	13.33 (0.2, 5)	13.17 (0.2, 5)		11.46 (0.2, 5)	15.30 (0.3, 5)	16.36 (0.7, 5)	13.75 (0.7, 3)	16.13 (0.9, 3)	12.11 (0.2, 5)	14.6 (0.5, 5)
M. fluviatilis		11.8 (0.1, 5)								
C. auratus				10.0 (0.0, 2)		16.3 (0.4, 5)			13.5 (0.2, 5)	

Appendix 42: Mean δ^{13}C (‰) of basal sources and consumers for samples collected in April 2008 (standard errors, and number of replicates are given in parentheses)

	NW4	**NW7**	**NW9**	**NW10**	**NW11**	**TS1**	**TS6**	**TS7**	**TS15**	**TS16**
Basal Sources										
Seston	-26.4	-27.5	-33.6	-26.4	-25.0	-21.0	-21.8	-22.7	-25.3	-24.1
	(0.1, 3)	(0.7, 3)	(0.9, 3)	(0.0, 3)	(0.1, 3)	(0.2, 3)	(0.1, 3)	(0.1, 3)	(0.1, 3)	(0.1, 3)
FPOM	-26.5	-27.5	-27.8	-27.2	-26.0	-21.7	-22.6	-25.2	-22.5	-23.9
	(0.5, 3)	(0.2, 3)	(0.2, 3)	(0.4, 3)	(0.3, 3)	(0.6, 3)	(0.4, 3)	(0.6, 3)	(0.4, 3)	(0.4, 3)
CPOM	-26.8	-27.0	-27.6	-26.9	-26.3	-20.7	-22.7	-25.7	-25.9	-24.4
	(0.4, 3)	(0.6, 3)	(0.2, 3)	(0.9, 3)	(0.4, 3)	(2.3, 3)	(0.9, 3)	(0.4, 3)	(1.3, 3)	(1.1, 3)
Biofilm	-28.0	-27.6		-19.8	-15.6	-14.6	-19.2	-22.8		-24.2
	(1.2, 2)	(0.3, 3)		(0.7, 3)	(0.8, 2)	(0.5, 3)	(0.1, 3)	(2.8, 2)		(2.7, 3)
Floating algae		-22.7			-26.3				-12.9	
		(-, 1)			(-, 1)				(0.8, 3)	
Algal Clump									-25.5	
									(1.8, 3)	
Microinvertebrate Consumers										
Zooplankton	-34.7	-34.5	-35.9	-32.9	-31.5	-21.0	-23.9	-24.7	-29.6	-32.6
	(0.4, 3)	(0.0, 3)	(0.1, 3)	(0.2, 3)	(0.2, 3)	(0.9, 3)	(0.2, 3)	(0.1, 3)	(0.1, 3)	(0.6, 3)
Primary Macroinvertebrate Consumers										
Chironomids				-29.5					-21.3	
				(1.1, 3)					(2.9, 3)	
Corixidae (Adults)	-21.5	-24.6	-28.3	-27.8	-25.1	-19.9		-25.3	-25.9	-20.1
	(-, 1)	(1.4, 2)	(0.2, 2)	(0.4, 3)				(1.2, 2)	(-, 1)	(1.4, 2)
Corixidae (Nymphs)		-27.1	-31.8	-29.2						-23.5
		(-, 1)	(1.2, 2)	(0.6, 2)						(-, 1)
Physidae			-29.0	-21.4				-23.9		
			(0.3, 3)	(0.2, 2)				(1.4, 3)		
Caenidae	-29.3			-31.1						
	(-, 1)			(-, 1)						
Baetidae			-32.6	-26.4	-22.6				-19.8	
			(0.3, 3)	(1.0, 2)	(-, 1)				(2.4, 3)	

	NW4	**NW7**	**NW9**	**NW10**	**NW11**	**TS1**	**TS6**	**TS7**	**TS15**	**TS16**
Trichoptera		-25.5 (-, 1)	-29.5 (1.7, 3)	-24.3 (0.7, 2)			-16.4 (-, 1)	-22.9 (1.9, 2)	-19.1 (1.1, 3)	
Bivalvia		-33.4 (0.1, 5)		-31.0 (0.2, 6)						
Secondary Macroinvertebrate Consumers										
Notonectidae (Adults)	-27.1 (1.5, 3)	-29.8 (0.8, 3)	-33.7 (0.4, 2)	-30.7 (0.4, 3)	-29.3 (0.4, 3)		-28.2 (1.0, 3)	-26.7 (1.0, 3)	-27.0 (0.2, 3)	-28.5 (0.5, 2)
Notonectidae (Nymphs)			-33.2 (1.0, 2)	-31.8 (0.3, 2)	-30.5 (-, 1)		-20.9 (0.2, 2)	-23.0 (0.2, 2)	-26.9 (0.2, 3)	-30.8 (0.4, 2)
Nepidae					-25.4 (0.3, 2)	-23.7 (-, 1)	-28.8 (1.7, 2)	-24.7 (0.9, 3)	-27.5 (-, 1)	-24.9 (1.2, 3)
Odonata										
Epiproctophora	-28.0 (-, 1)	-21.7 (0.0, 2)	-31.0 (0.3, 3)	-30.3 (-, 1)	-17.3 (-, 1)	-17.2 (0.2, 3)	-16.7 (-, 1)	-24.9 (0.7, 3)	-25.9 (0.1, 2)	
Zygoptera			-34.5 (0.5, 3)	-28.6 (0.3, 2)				-21.7 (1.4, 3)	-23.2 (-, 1)	
Naucoridae	-25.3 (-, 1)									
Tanypodinae	-27.2 (-, 1)	-27.4 (-, 1)	-33.2 (-, 1)	-27.4 (-, 1)	-27.3 (-, 1)				-23.8 (-, 1)	
Spencerhydrus sp.		-24.2 (-, 1)		-28.1 (1.0, 3)	-25.3 (-, 1)	-27.6 (0.3, 3)	-25.3 (0.2, 3)		-26.0 (0.4, 3)	-24.4 (-, 1)
Eretes sp.					-26.5 (0.2, 2)			-25.2 (0.3, 2)		
Hyphydrus sp.			-33.6 (0.2, 3)					-25.0 (-, 1)		
Macrobrachium sp. (Small)	-25.2 (0.3, 5)	-25.2 (0.2, 5)	-28.9 (0.3, 5)	-26.7 (0.1, 5)	-25.5 (0.3, 5)	-15.5 (0.2, 5)	-17.6 (0.4, 5)	-21.8 (0.6, 6)	-22.1 (0.3, 4)	-21.2 (0.8, 5)
Macrobrachium sp. (Large)	-23.2 (0.3, 5)	-24.3 (0.2, 5)		-26.3 (0.2, 5)	-24.7 (0.3, 5)	-17.5 (1.4, 5)	-17.0 (0.4, 5)		-21.3 (0.2, 5)	-22.0 (0.8, 5)
Cherax sp.	-22.7 (0.4, 4)	-23.9 (0.8, 4)	-28.6 (0.4, 2)	-25.1 (0.4, 5)	-23.7 (0.1, 5)	-18.1 (0.3, 5)	-16.7 (0.2, 5)		-20.5 (0.4, 6)	-20.6 (0.4, 5)

	NW4	**NW7**	**NW9**	**NW10**	**NW11**	**TS1**	**TS6**	**TS7**	**TS15**	**TS16**
Acarina				-23.1 (-, 1)					-25.4 (1.9, 2)	
Fish										
L. unicolor				-26.5 (0.7, 5)	-25.2 (0.4, 5)	-17.8 (0.8, 5)	-18.6 (0.9, 5)	-19.4 (0.6, 5)	-22.4 (0.5, 6)	-24.3 (1.0, 5)
N. erebi (small)	-29.8 (0.2, 5)	-30.0 (0.3, 5)	-32.9 (0.1, 5)	-29.9 (0.1, 5)	-29.7 (-, 1)	-17.1 (0.3, 2)	-24.5 (0.1, 5)	-24.3 (0.2, 3)		-23.5 (1.6, 5)
N. erebi (medium)	-30.7 (0.4, 5)	-30.4 (0.3, 5)	-33.1 (0.1, 3)	-29.9 (0.2, 5)	-29.8 (0.1, 5)		-20.5 (0.4, 5)	-27.0 (3.7, 2)		-19.9 (0.7, 5)
N. erebi (large)	-30.4 (0.4, 5)	-29.6 (0.2, 5)		-31.4 (0.2, 4)			-21.3 (0.5, 5)	-24.3 (1.5, 5)		-24.7 (1.9, 4)
N. erebi (large) - bone	-31.2 (0.3, 3)	-29.5 (0.2, 5)		-31.9 (0.3, 4)			-21.0 (0.6, 5)			-24.5 (1.9, 4)
C. carpio		-24.2 (0.5, 5)		-27.3 (0.4, 5)	-26.6 (0.8, 5)	-17.8 (0.1, 4)			-20.7 (0.4, 5)	-21.0 (0.7, 5)
C. carpio - bone				-27.0 (0.5, 2)		-20.6 (0.4, 3)				
M. ambigua	-28.4 (1.1, 5)	-28.2 (0.8, 5)	-23.1 (-, 1)	-28.1 (0.6, 5)	-27.6 (0.6, 5)	-16.8 (-, 1)	-22.1 (0.7, 5)	-21.4 (1.2, 5)	-22.8 (0.2, 5)	-25.6 (1.4, 5)
Hypseleotris spp.	-26.9 (0.1, 3)	-26.8 (0.3, 5)							-22.5 (0.9, 3)	
R. semoni	-27.2 (-,1)	-28.0 (0.2, 5)								
G. holbrooki	-26.41 (0.8, 5)	-25.29 (0.5, 5)		-27.08 (0.7, 5)	-26.69 (0.6, 5)	-17.75 (0.5, 5)	-20.83 (1.3, 3)	-18.02 (1.0, 3)	-22.26 (0.3, 5)	-24.57 (1.0, 5)
M. fluviatilis		-23.8 (0.4, 5)								
C. auratus				-28.2 (0.9, 2)		-17.4 (0.6, 5)			-23.5 (0.3, 5)	

Appendix 43: Percent contribution of primary sources to consumers at the natural wetland (NW) and typical storage (TS) sites in April 2008. Results were calculated using IsoSource Mixing Model (Phillips and Gregg, 2003) using $\delta^{13}C$ values of source and consumer tissues

	Algal clump				Biofilm				Floating algae				Seston				Zooplankton				CPOM			
NW4	Mean	SD	Min	Max	Mean	SD	Min	Max	Mean	SD	Min	Max	Mean	SD	Min	Max	Mean	SD	Min	Max	Mean	SD	Min	Max
Caenidae					26.8	19.3	0.0	81.0					21.6	15.6	0.0	65.0	28.8	3.4	19.0	36.0	22.8	16.5	0.0	69.0
Corixidae A																								
Epiproctophora					33.3	23.9	0.0	100.0					26.8	19.3	0.0	81.0	11.4	4.2	0.0	20.0	28.4	20.5	0.0	86.0
Naucoridae																								
Notonectidae A					18.7	12.4	0.0	53.0					39.9	22.0	0.0	92.0	3.4	2.3	0.0	10.0	38.0	24.8	0.0	97.0
Tanypodinae					21.0	13.8	0.0	58.0					37.9	21.6	0.0	91.0	3.9	2.5	0.0	11.0	37.2	24.1	0.0	96.0
Cherax sp.																								
Macrobrachium L																								
Macrobrachium S																								
Fish																								
G. holbrooki					2.0	1.9	0.0	7.0					89.5	6.1	74.0	100.0	0.1	0.3	0.0	1.0	8.3	6.7	0.0	26.0
Hypseliotris spp.					12.4	8.8	0.0	39.0					48.0	22.5	0.0	94.0	2.0	1.7	0.0	7.0	37.6	25.9	0.0	100.0
M. ambigua					31.3	22.5	0.0	95.0					25.2	18.2	0.0	77.0	16.7	4.0	5.0	25.0	26.7	19.3	0.0	81.0
N. erebi L					21.7	15.7	0.0	66.0					17.5	12.7	0.0	53.0	42.4	2.8	34.0	48.0	18.5	13.4	0.0	56.0
N. erebi LB					17.6	12.8	0.0	54.0					14.2	10.4	0.0	43.0	53.1	2.3	46.0	58.0	15.0	11.0	0.0	46.0
N. erebi M					20.0	14.5	0.0	61.0					16.0	11.7	0.0	49.0	47.0	2.6	39.0	52.0	17.0	12.4	0.0	52.0
N. erebi S					24.4	17.6	0.0	74.0					19.6	14.2	0.0	60.0	35.1	3.1	26.0	42.0	20.8	15.1	0.0	63.0
R. semoni					21.0	13.8	0.0	58.0					37.9	21.6	0.0	91.0	3.9	2.5	0.0	11.0	37.2	24.1	0.0	96.0
Avg.					20.9	14.8	0.0	62.2					32.8	16.3	6.2	74.7	20.7	2.6	14.1	26.8	25.6	17.8	0.0	72.3
NW7																								
Corixidae A					9.7	7.9	0.0	40.0	65.4	5.2	55.0	84.0	10.0	8.2	0.0	41.0	3.8	3.3	0.0	16.0	11.1	9.0	0.0	45.0
Corixidae N					22.6	17.9	0.0	91.0	19.4	11.6	0.0	63.0	23.3	18.4	0.0	94.0	9.2	7.4	0.0	38.0	25.5	20.1	0.0	100.0
Spencerhydrus sp.					8.0	6.6	0.0	33.0	71.6	4.4	63.0	87.0	8.2	6.8	0.0	34.0	3.1	2.7	0.0	13.0	9.1	7.5	0.0	37.0
Epiproctophora																								

	Algal clump				Biofilm				Floating algae				Seston				Zooplankton				CPOM			
NW7	Mean	SD	Min	Max	Mean	SD	Min	Max	Mean	SD	Min	Max	Mean	SD	Min	Max	Mean	SD	Min	Max	Mean	SD	Min	Max
Notonectidae A					17.2	13.7	0.0	70.0	9.8	8.0	0.0	40.0	16.9	13.5	0.0	68.0	40.2	5.4	30.0	60.0	15.8	12.7	0.0	64.0
Tanypodinae					24.8	19.4	0.0	97.0	15.3	11.8	0.0	60.0	25.0	19.7	0.0	100.0	10.7	7.8	0.0	40.0	24.2	18.8	0.0	96.0
Trichoptera					14.5	11.6	0.0	59.0	48.2	7.6	33.0	76.0	15.0	12.0	0.0	61.0	5.8	4.8	0.0	24.0	16.6	13.2	0.0	67.0
Mussels					3.9	3.4	0.0	16.0	2.0	1.9	0.0	9.0	3.7	3.3	0.0	16.0	86.8	1.5	84.0	92.0	3.5	3.1	0.0	15.0
Cherax sp.					6.4	5.3	0.0	26.0	77.4	3.6	70.0	90.0	6.6	5.5	0.0	27.0	2.4	2.2	0.0	11.0	7.3	6.0	0.0	30.0
Macrobrachium L					8.2	6.8	0.0	34.0	70.7	4.5	62.0	87.0	8.5	7.0	0.0	35.0	3.2	2.8	0.0	14.0	9.4	7.7	0.0	38.0
Macrobrachium S					12.9	10.4	0.0	53.0	53.9	6.8	40.0	79.0	13.3	10.7	0.0	54.0	5.1	4.3	0.0	22.0	14.8	11.8	0.0	60.0
Fish																								
C. carpio					7.8	6.4	0.0	32.0	72.3	4.3	64.0	88.0	8.0	6.6	0.0	33.0	3.0	2.7	0.0	13.0	8.9	7.3	0.0	36.0
G. holbrooki					13.3	10.7	0.0	54.0	52.6	7.0	39.0	78.0	13.7	11.0	0.0	56.0	5.3	4.5	0.0	22.0	15.2	12.1	0.0	61.0
Hypseliotris sp.					20.7	16.4	0.0	84.0	25.9	10.7	5.0	66.0	21.4	17.0	0.0	86.0	8.4	6.9	0.0	35.0	23.6	18.7	0.0	95.0
M. ambigua					23.0	18.2	0.0	92.0	13.2	10.6	0.0	54.0	22.5	17.8	0.0	90.0	20.2	7.1	8.0	47.0	21.1	16.7	0.0	85.0
M. fluviatilis					5.5	4.7	0.0	23.0	80.3	3.2	74.0	91.0	5.7	4.8	0.0	24.0	2.0	1.9	0.0	9.0	6.4	5.3	0.0	26.0
N. erebi L					17.9	14.3	0.0	72.0	10.2	8.3	0.0	42.0	17.5	14.0	0.0	71.0	37.9	5.6	28.0	59.0	16.4	13.1	0.0	66.0
N. erebi LB					18.4	14.6	0.0	74.0	10.5	8.5	0.0	43.0	18.0	14.3	0.0	72.0	36.2	5.7	26.0	58.0	16.9	13.5	0.0	68.0
N. erebi M					14.9	11.9	0.0	60.0	8.5	6.9	0.0	35.0	14.6	11.7	0.0	59.0	48.5	4.7	40.0	66.0	13.7	11.0	0.0	55.0
N. erebi S					16.2	13.0	0.0	65.0	9.3	7.5	0.0	38.0	15.9	12.7	0.0	64.0	43.8	5.1	35.0	63.0	14.9	11.9	0.0	60.0
R. semoni					23.6	18.7	0.0	95.0	13.5	10.9	0.0	55.0	23.1	18.3	0.0	93.0	18.1	7.3	5.0	46.0	21.7	17.2	0.0	87.0
Avg.					14.5	11.6	0.0	58.5					14.5	11.7	0.0	58.9	19.7	4.7	12.8	37.4	14.8	11.8	0.0	59.6
NW9																								
Baetidae													41.6	24.5	0.0	85.0	30.0	17.5	0.0	60.0	28.4	7.0	15.0	41.0
Corixidae A													5.5	3.5	0.0	12.0	3.5	2.5	0.0	8.0	90.9	1.3	88.0	93.0
Corixidae N													35.2	20.6	0.0	71.0	24.8	14.7	0.0	51.0	40.1	5.9	29.0	51.0
Hyphydrus sp.													49.0	28.5	0.0	100.0	37.4	20.4	0.0	73.0	13.6	8.1	0.0	28.0
Epiproctophora													28.5	16.9	0.0	58.0	20.5	12.1	0.0	42.0	51.0	4.9	42.0	60.0
Notonectidae A													48.7	28.5	0.0	100.0	37.8	20.4	0.0	73.0	13.5	8.1	0.0	28.0
Notonectidae N													47.6	27.8	0.0	96.0	33.6	19.9	0.0	68.0	18.8	7.9	4.0	33.0

	Algal clump				Biofilm				Floating algae				Seston				Zooplankton				CPOM			
NW9	Mean	SD	Min	Max	Mean	SD	Min	Max	Mean	SD	Min	Max	Mean	SD	Min	Max	Mean	SD	Min	Max	Mean	SD	Min	Max
Tanypodinae													47.6	96.0	27.8	0.0	33.6	68.0	19.9	0.0	18.8	33.0	7.9	4.0
Trichoptera													16.0	9.5	0.0	33.0	11.0	6.8	0.0	23.0	73.0	2.8	67.0	78.0
Zygoptera													31.2	18.4	0.0	65.0	60.3	13.2	35.0	83.0	8.5	5.3	0.0	18.0
Physidae													11.3	7.0	0.0	24.0	7.9	5.0	0.0	17.0	80.7	2.2	76.0	85.0
Cherax sp.													8.5	5.3	0.0	18.0	6.0	3.8	0.0	13.0	85.5	1.7	82.0	89.0
Macrobrachium S													11.1	6.6	0.0	23.0	7.5	4.7	0.0	16.0	81.4	2.1	77.0	85.0
Fish																								
M. ambigua																								
N. erebi M													46.2	26.8	0.0	93.0	32.3	19.2	0.0	66.0	21.5	7.7	7.0	35.0
N. erebi S													44.9	26.3	0.0	91.0	31.9	18.8	0.0	65.0	23.2	7.6	9.0	37.0
Avg.													31.5	23.1	1.9	57.9	25.2	16.5	3.7	43.9	43.3	7.0	33.6	51.0
NW10																								
Acarina					59.2	5.8	49.0	75.0					16.9	12.3	0.0	51.0	8.3	6.2	0.0	26.0	15.6	11.4	0.0	47.0
Baetidae					19.2	11.1	0.0	50.0					33.4	23.9	0.0	100.0	16.6	12.1	0.0	50.0	30.8	22.1	0.0	93.0
Caenidae					4.5	3.5	0.0	14.0					9.5	7.1	0.0	29.0	75.6	4.0	68.0	86.0	10.5	7.7	0.0	32.0
Chironominae					8.6	6.4	0.0	26.0					17.7	12.9	0.0	54.0	54.3	7.2	41.0	74.0	19.4	14.1	0.0	59.0
Corixidae A					12.6	9.3	0.0	39.0					26.1	18.8	0.0	79.0	32.9	10.4	14.0	62.0	28.4	20.5	0.0	86.0
Corixidae N					9.1	6.8	0.0	28.0					18.9	13.7	0.0	57.0	51.4	7.6	38.0	72.0	20.6	14.9	0.0	62.0
Spencerhydrus sp.					12.0	8.8	0.0	37.0					24.7	17.8	0.0	75.0	36.3	9.9	18.0	64.0	27.0	19.4	0.0	82.0
Epiproctophora					6.5	4.9	0.0	20.0					13.5	9.9	0.0	41.0	65.2	5.6	55.0	80.0	14.8	10.8	0.0	45.0
Notonectidae A					5.3	4.1	0.0	17.0					11.3	8.3	0.0	34.0	71.1	4.7	63.0	84.0	12.3	9.0	0.0	37.0
Notonectidae N					2.6	2.2	0.0	9.0					5.8	4.4	0.0	18.0	85.3	2.5	81.0	92.0	6.3	4.8	0.0	19.0
Tanypodinae					13.7	10.1	0.0	42.0					28.3	20.4	0.0	85.0	27.1	11.3	7.0	58.0	30.9	22.2	0.0	93.0
Trichoptera					44.7	7.7	31.0	66.0					22.9	16.5	0.0	69.0	11.3	8.3	0.0	35.0	21.1	15.3	0.0	64.0
Zygoptera					10.7	8.0	0.0	33.0					22.2	16.0	0.0	67.0	42.9	8.9	27.0	68.0	24.2	17.5	0.0	73.0
Physidae					79.4	3.1	74.0	88.0					8.6	6.4	0.0	26.0	4.1	3.2	0.0	13.0	7.9	5.9	0.0	24.0
Cherax sp.					34.7	9.0	19.0	60.0					27.0	19.5	0.0	81.0	13.4	9.8	0.0	41.0	24.9	18.0	0.0	75.0

	Algal clump				Biofilm				Floating algae				Seston				Zooplankton				CPOM			
NW10	Mean	SD	Min	Max	Mean	SD	Min	Max	Mean	SD	Min	Max	Mean	SD	Min	Max	Mean	SD	Min	Max	Mean	SD	Min	Max
Macrobrachium L					19.8	11.1	0.0	50.0					33.1	23.8	0.0	100.0	16.5	12.0	0.0	50.0	30.6	22.0	0.0	92.0
Macrobrachium S					15.9	11.2	0.0	47.0					32.2	22.8	0.0	96.0	18.6	12.4	0.0	53.0	33.3	23.6	0.0	98.0
Mussel					4.6	3.6	0.0	15.0					9.8	7.3	0.0	30.0	74.8	4.1	67.0	86.0	10.8	7.9	0.0	33.0
Fish																								
C. auratus					11.7	8.7	0.0	36.0					24.2	17.5	0.0	73.0	37.6	9.7	20.0	64.0	26.5	19.0	0.0	80.0
C. carpio B					14.7	10.8	0.0	45.0					30.1	21.7	0.0	91.0	22.2	12.1	1.0	56.0	33.0	23.6	0.0	99.0
C. carpio					13.9	10.2	0.0	43.0					28.6	20.6	0.0	86.0	26.2	11.4	6.0	58.0	31.3	22.5	0.0	94.0
G. holbrooki					14.6	10.7	0.0	45.0					30.0	21.6	0.0	90.0	22.7	12.0	1.0	56.0	32.7	23.5	0.0	99.0
L. unicolor					17.6	11.3	0.0	49.0					33.5	23.9	0.0	99.0	17.1	12.3	0.0	52.0	31.7	22.6	0.0	95.0
M. ambigua					11.9	8.8	0.0	36.0					24.4	17.6	0.0	74.0	36.9	9.8	19.0	64.0	26.7	19.3	0.0	81.0
N. erebi L					3.5	2.8	0.0	11.0					7.5	5.7	0.0	23.0	80.7	3.3	75.0	89.0	8.3	6.2	0.0	25.0
N. erebi LB					2.3	2.0	0.0	8.0					5.0	4.0	0.0	16.0	87.1	2.3	83.0	93.0	5.6	4.3	0.0	17.0
N. erebi M					7.5	5.7	0.0	23.0					15.7	11.5	0.0	48.0	59.7	6.4	48.0	77.0	17.1	12.5	0.0	52.0
N. erebi S					7.5	5.6	0.0	23.0					15.6	11.4	0.0	47.0	59.9	6.4	48.0	77.0	17.0	12.4	0.0	52.0
Avg.					16.7	7.3	6.2	37.0					20.6	14.9	0.0	62.1	41.3	8.1	27.9	63.6	21.4	15.5	0.0	64.6
NW11																								
Baetidae					38.1	6.6	24.0	56.0					25.1	18.2	0.0	76.0	14.6	10.7	0.0	44.0	22.1	16.0	0.0	66.0
Corixidae A					16.8	8.8	0.0	41.0					33.7	24.1	0.0	100.0	19.8	14.3	0.0	60.0	29.7	21.3	0.0	89.0
Spencerhydrus sp.					15.0	8.9	0.0	39.0					33.3	23.5	0.0	96.0	20.7	14.6	0.0	61.0	31.0	21.9	0.0	92.0
Eretes sp.					10.3	7.7	0.0	32.0					25.9	18.7	0.0	78.0	31.3	13.9	3.0	68.0	32.4	23.2	0.0	97.0
Epiproctophora					84.7	1.8	81.0	89.0					6.3	4.8	0.0	19.0	3.5	2.8	0.0	11.0	5.5	4.2	0.0	17.0
Nepidae					14.6	0.0	39.0	8.8					33.0	0.0	95.0	23.2	21.1	0.0	62.0	14.6	31.4	0.0	92.0	22.0
Notonectidae A					4.4	3.5	0.0	14.0					11.4	8.4	0.0	35.0	69.9	6.3	57.0	86.0	14.4	10.5	0.0	43.0
Notonectidae N					2.0	1.7	0.0	7.0					5.5	4.2	0.0	17.0	85.6	3.2	79.0	94.0	6.9	5.2	0.0	21.0
Tanypodinae					8.5	6.4	0.0	26.0					21.5	15.6	0.0	65.0	43.1	11.6	19.0	74.0	26.8	19.4	0.0	81.0
Cherax sp.					29.0	7.6	13.0	49.0					28.9	20.8	0.0	87.0	16.9	12.3	0.0	51.0	25.3	18.3	0.0	76.0
Macrobrachium L					20.4	8.5	3.0	43.0					32.3	23.2	0.0	97.0	18.9	13.7	0.0	57.0	28.5	20.4	0.0	85.0

	Algal clump				Biofilm				Floating algae				Seston				Zooplankton				CPOM			
NW11	Mean	SD	Min	Max	Mean	SD	Min	Max	Mean	SD	Min	Max	Mean	SD	Min	Max	Mean	SD	Min	Max	Mean	SD	Min	Max
Macrobrachium S					13.7	8.7	0.0	38.0					32.3	22.5	0.0	93.0	21.8	14.7	0.0	63.0	32.2	22.5	0.0	94.0
Fish																								
C. carpio					10.1	7.5	0.0	31.0					25.4	18.3	0.0	77.0	32.8	13.6	5.0	69.0	31.7	22.8	0.0	95.0
G. holbrooki					9.9	7.3	0.0	30.0					24.8	17.9	0.0	75.0	34.4	13.3	7.0	70.0	31.0	22.2	0.0	93.0
L. unicolor					15.7	8.9	0.0	40.0					33.6	23.9	0.0	97.0	20.3	14.5	0.0	61.0	30.4	21.6	0.0	91.0
M. ambigua					7.9	5.9	0.0	24.0					20.0	14.5	0.0	60.0	47.3	10.8	25.0	76.0	24.9	18.0	0.0	75.0
N. erebi M					3.4	2.8	0.0	11.0					9.2	6.8	0.0	28.0	75.9	5.1	65.0	89.0	11.5	8.4	0.0	35.0
N. erebi S					3.6	2.9	0.0	11.0					9.4	7.0	0.0	29.0	75.1	5.3	64.0	89.0	11.9	8.8	0.0	36.0
Avg.					17.1	5.9	8.9	32.8					22.9	15.1	5.3	64.0	36.3	10.0	21.4	63.2	23.8	15.8	5.1	67.1
TS1																								
Corixidae A					16.0	1.4	12.0	19.0					27.5	19.8	0.0	84.0	27.5	19.8	0.0	84.0	29.1	20.9	0.0	88.0
Spencerhydrus sp.																								
Epiproctophora					58.4	1.1	56.0	60.0					13.5	9.9	0.0	42.0	13.5	9.9	0.0	42.0	14.5	10.5	0.0	44.0
Nepidae																								
Cherax sp.					44.5	1.1	41.0	47.0					18.2	13.3	0.0	56.0	18.2	13.3	0.0	56.0	19.1	13.9	0.0	59.0
Macrobrachium L					54.0	1.1	51.0	56.0					15.1	11.0	0.0	46.0	15.1	11.0	0.0	46.0	15.8	11.5	0.0	49.0
Macrobrachium S					53.8	1.1	51.0	56.0					15.2	11.1	0.0	46.0	15.2	11.1	0.0	46.0	15.8	11.7	0.0	49.0
Fish																								
C. auratus					55.5	1.0	53.0	57.0					14.5	10.6	0.0	45.0	14.5	10.6	0.0	45.0	15.5	11.3	0.0	47.0
C. carpio					49.3	1.1	46.0	51.0					16.6	12.1	0.0	51.0	16.6	12.1	0.0	51.0	17.5	12.6	0.0	54.0
C. carpio B					4.6	1.5	0.0	7.0					31.2	22.4	0.0	95.0	31.2	22.4	0.0	95.0	33.1	23.7	0.0	100.0
C. carpio G					39.8	1.2	37.0	42.0					19.7	14.3	0.0	60.0	19.7	14.3	0.0	60.0	20.7	15.0	0.0	63.0
C. carpio K					22.3	1.3	19.0	25.0					25.5	18.4	0.0	77.0	25.5	18.4	0.0	77.0	26.7	19.3	0.0	81.0
G. holbrooki					50.1	1.1	47.0	52.0					16.3	11.9	0.0	50.0	16.3	11.9	0.0	50.0	17.3	12.5	0.0	53.0
L. unicolor					49.9	1.1	47.0	52.0					16.4	12.0	0.0	50.0	16.4	12.0	0.0	50.0	17.3	12.6	0.0	53.0
M. ambigua					65.1	1.0	63.0	67.0					11.5	8.4	0.0	35.0	11.5	8.4	0.0	35.0	12.0	8.9	0.0	37.0

	Algal clump				Biofilm				Floating algae				Seston				Zooplankton				CPOM			
TS1	Mean	SD	Min	Max	Mean	SD	Min	Max	Mean	SD	Min	Max	Mean	SD	Min	Max	Mean	SD	Min	Max	Mean	SD	Min	Max
N. erebi S					61.0	1.1	59.0	63.0					12.9	9.4	0.0	39.0	12.9	9.4	0.0	39.0	13.3	9.8	0.0	41.0
Avg.					44.6	1.2	41.6	46.7					18.2	13.2	0.0	55.4	18.2	13.2	0.0	55.4	19.1	13.9	0.0	58.4
TS6																								
Spencerhydrus sp.																								
Epiproctophora																								
Nepidae																								
Notonectidae A																								
Notonectidae N					49.5	6.5	31.0	66.0					22.1	16.0	0.0	69.0	11.9	8.8	0.0	38.0	16.5	12.0	0.0	52.0
Trichoptera																								
Cherax sp.																								
Macrobrachium L																								
Macrobrachium S																								
Fish																								
G. holbrooki					52.5	6.2	35.0	68.0					20.8	15.1	0.0	65.0	11.2	8.3	0.0	35.0	15.5	11.3	0.0	49.0
L. unicolor																								
M. ambigua					17.3	9.2	0.0	41.0					32.7	22.5	0.0	92.0	21.2	14.3	0.0	62.0	28.8	19.8	0.0	85.0
N. erebi L					39.9	7.6	18.0	59.0					26.3	19.0	0.0	82.0	14.2	10.4	0.0	44.0	19.6	14.2	0.0	61.0
N. erebi LB					47.7	6.7	29.0	64.0					22.9	16.6	0.0	71.0	12.3	9.1	0.0	39.0	17.1	12.4	0.0	53.0
N. erebi M					61.7	5.1	47.0	74.0					16.8	12.3	0.0	53.0	9.0	6.7	0.0	29.0	12.5	9.2	0.0	40.0
N. erebi S																								
Avg.					44.8	6.9	26.7	62.0					23.6	16.9	0.0	72.0	13.3	9.6	0.0	41.2	18.3	13.2	0.0	56.7
TS7																								
Corixidae A					4.7	3.8	0.0	17.0					4.5	3.6	0.0	16.0	14.3	10.7	0.0	49.0	76.4	7.6	51.0	89.0
Eretes sp.					5.9	4.6	0.0	20.0					5.6	4.4	0.0	20.0	17.5	12.9	0.0	59.0	71.0	8.9	41.0	86.0
Hyphydrus sp.					7.9	6.0	0.0	27.0					7.6	5.8	0.0	26.0	23.4	17.0	0.0	77.0	61.1	11.5	23.0	80.0

	Algal clump				Biofilm				Floating algae				Seston				Zooplankton				CPOM			
TS7	Mean	SD	Min	Max	Mean	SD	Min	Max	Mean	SD	Min	Max	Mean	SD	Min	Max	Mean	SD	Min	Max	Mean	SD	Min	Max
Epiproctophora					9.3	7.0	0.0	31.0					9.0	6.7	0.0	30.0	27.3	19.8	0.0	89.0	54.4	13.3	11.0	76.0
Nepidae					11.7	8.6	0.0	38.0					11.2	8.3	0.0	37.0	33.5	24.0	0.0	100.0	43.6	16.0	0.0	69.0
Notonectidae A																								
Notonectidae N					42.6	26.4	0.0	96.0					46.2	25.3	0.0	93.0	6.8	4.6	0.0	20.0	4.4	3.1	0.0	13.0
Trichoptera					42.1	27.3	0.0	100.0					51.3	26.3	0.0	97.0	4.1	3.2	0.0	14.0	2.5	2.1	0.0	9.0
Zygoptera																								
Physidae					26.3	16.0	0.0	66.0					25.5	15.4	0.0	63.0	27.6	17.4	0.0	65.0	20.5	11.4	0.0	43.0
Macrobrachium S																								
Fish																								
G. holbrooki																								
L. unicolor																								
M. ambigua																								
N. erebi L					18.8	12.1	0.0	53.0					18.1	11.7	0.0	51.0	34.2	22.4	0.0	84.0	28.9	14.7	0.0	55.0
N. erebi M																								
N. erebi S					17.3	11.4	0.0	50.0					16.6	11.0	0.0	48.0	35.1	23.4	0.0	88.0	31.0	15.4	0.0	58.0
Avg.					18.7	12.3	0.0	49.8					19.6	11.9	0.0	48.1	22.4	15.5	0.0	64.5	39.4	10.4	12.6	57.8
TS15																								
Acarina	25.0	19.7	0.0	100.0					6.9	4.8	0.0	25.0	25.1	19.7	0.0	98.0	18.8	14.8	0.0	75.0	24.3	19.1	0.0	97.0
Baetidae	13.8	11.0	0.0	55.0					48.6	2.7	44.0	59.0	14.0	11.3	0.0	56.0	10.3	8.3	0.0	42.0	13.4	10.7	0.0	54.0
Chironominae	16.7	13.3	0.0	67.0					37.6	3.3	32.0	50.0	17.0	13.6	0.0	68.0	12.5	10.1	0.0	50.0	16.2	12.9	0.0	65.0
Corixidae A	23.3	18.4	0.0	93.0					5.3	4.5	0.0	22.0	22.0	17.4	0.0	89.0	23.9	14.4	0.0	78.0	25.5	20.1	0.0	100.0
Spencerhydrus sp.	22.5	17.8	0.0	90.0					5.1	4.3	0.0	22.0	21.3	16.9	0.0	86.0	26.3	14.0	0.0	78.0	24.8	19.6	0.0	100.0
Epiproctophora	22.7	18.0	0.0	91.0					5.2	4.4	0.0	22.0	21.5	17.0	0.0	86.0	25.7	14.1	0.0	78.0	25.0	19.8	0.0	100.0
Nepidae	12.9	10.4	0.0	52.0					2.8	2.5	0.0	12.0	12.2	9.8	0.0	49.0	57.9	8.2	43.0	88.0	14.2	11.4	0.0	57.0
Notonectidae A	15.9	12.7	0.0	64.0					3.5	3.1	0.0	15.0	15.0	12.0	0.0	60.0	48.2	9.9	30.0	85.0	17.5	13.9	0.0	70.0
Notonectidae N	16.9	13.5	0.0	68.0					3.8	3.3	0.0	16.0	16.0	12.8	0.0	64.0	44.7	10.6	25.0	84.0	18.6	14.8	0.0	75.0
Tanypodinae	21.7	17.2	0.0	87.0					18.9	4.2	12.0	35.0	22.1	17.5	0.0	88.0	16.3	13.0	0.0	65.0	21.0	16.7	0.0	84.0

	Algal clump				Biofilm				Floating algae				Seston				Zooplankton				CPOM			
TS15	Mean	SD	Min	Max	Mean	SD	Min	Max	Mean	SD	Min	Max	Mean	SD	Min	Max	Mean	SD	Min	Max	Mean	SD	Min	Max
Trichoptera	12.2	9.9	0.0	49.0					54.5	2.4	50.0	63.0	12.4	10.0	0.0	50.0	9.1	7.4	0.0	37.0	11.8	9.6	0.0	47.0
Zygoptera	20.5	16.3	0.0	82.0					23.4	4.0	16.0	38.0	20.9	16.5	0.0	84.0	15.4	12.3	0.0	62.0	19.9	15.8	0.0	80.0
Cherax sp.	15.0	12.0	0.0	60.0					44.0	3.0	39.0	55.0	15.3	12.2	0.0	61.0	11.2	9.1	0.0	45.0	14.5	11.7	0.0	58.0
Macrobrachium L	16.7	13.4	0.0	67.0					37.5	3.3	32.0	50.0	17.0	13.6	0.0	68.0	12.5	10.1	0.0	50.0	16.2	12.9	0.0	65.0
Macrobrachium S	18.3	14.5	0.0	73.0					31.6	3.6	25.0	45.0	18.6	14.8	0.0	75.0	13.7	11.0	0.0	55.0	17.8	14.1	0.0	71.0
Fish																								
C. auratus	21.0	16.6	0.0	84.0					21.6	4.1	14.0	37.0	21.4	16.9	0.0	86.0	15.7	12.6	0.0	63.0	20.4	16.2	0.0	82.0
C. carpio	15.5	12.4	0.0	62.0					42.2	3.0	37.0	54.0	15.7	12.6	0.0	63.0	11.6	9.3	0.0	47.0	15.0	12.0	0.0	60.0
G. holbrooki	18.7	14.9	0.0	75.0					30.2	3.6	24.0	44.0	19.0	15.1	0.0	76.0	14.0	11.2	0.0	56.0	18.1	14.4	0.0	73.0
Hypseleotris spp.	19.1	15.1	0.0	76.0					28.7	3.7	22.0	43.0	19.4	15.4	0.0	78.0	14.3	11.4	0.0	58.0	18.5	14.7	0.0	74.0
L. unicolor	18.9	15.0	0.0	76.0					29.4	3.7	23.0	43.0	19.2	15.3	0.0	77.0	14.1	11.3	0.0	57.0	18.3	14.6	0.0	73.0
M. ambigua	19.7	15.6	0.0	79.0					26.5	3.8	20.0	41.0	20.0	15.9	0.0	80.0	14.7	11.8	0.0	59.0	19.1	15.2	0.0	77.0
Avg.	18.4	14.7	0.0	73.8					24.2	3.6	18.6	37.7	18.3	14.6	0.0	73.4	20.5	11.2	4.7	62.5	18.6	14.8	0.0	74.4
TS16																								
Corixidae A																								
Corixidae N																								
Spencerhydrus sp.					32.3	23.0	0.0	98.0					32.9	23.4	0.0	97.0	2.1	1.0	0.0	4.0	32.7	23.8	0.0	100.0
Nepidae					29.9	21.6	0.0	92.0					30.6	22.1	0.0	92.0	7.9	1.0	5.0	10.0	31.6	22.4	0.0	95.0
Notonectidae A					16.2	11.7	0.0	49.0					15.8	11.5	0.0	49.0	51.0	0.8	49.0	52.0	17.0	11.7	0.0	51.0
Notonectidae N					7.7	5.6	0.0	23.0					7.7	5.6	0.0	22.0	77.8	0.8	77.0	79.0	6.7	5.3	0.0	23.0
Cherax sp.																								
Macrobrachium L																								
Macrobrachium S																								
Fish																								
C. carpio																								
G. holbrooki					31.5	22.8	0.0	96.0					31.3	22.7	0.0	96.0	3.7	1.0	1.0	6.0	33.4	23.4	0.0	99.0

	Algal clump				Biofilm				Floating algae				Seston				Zooplankton				CPOM			
TS16	Mean	SD	Min	Max	Mean	SD	Min	Max	Mean	SD	Min	Max	Mean	SD	Min	Max	Mean	SD	Min	Max	Mean	SD	Min	Max
L. unicolor					36.1	24.2	0.0	99.0					35.5	23.4	0.0	99.0	0.8	0.8	0.0	2.0	27.6	19.9	0.0	89.0
M. ambigua					27.8	20.0	0.0	84.0					28.0	20.0	0.0	83.0	16.3	1.0	14.0	18.0	28.0	20.9	0.0	86.0
N. erebi L					31.5	22.5	0.0	95.0					31.7	22.5	0.0	94.0	5.2	1.0	2.0	7.0	31.6	23.3	0.0	98.0
N. erebi LB					32.4	23.3	0.0	97.0					31.5	22.7	0.0	97.0	2.5	1.0	0.0	4.0	33.6	23.7	0.0	100.0
N. erebi M																								
N. erebi S																								
Avg.					27.3	19.4	0.0	81.4					27.2	19.3	0.0	81.0	18.6	0.9	16.4	20.2	26.9	19.4	0.0	82.3

A = adult, N = nymph, S = small, M = medium = L = large, B= bone
Where no data is given IsoSource was unable to compute solutions

Appendix 44: Mean Euclidean distances between δ^{13}C values of each food web component at natural wetlands (NW) and typical storages (TS).

	NW		**TS**	
	Mean	**s.e.**	**Mean**	**s.e.**
Algae	5.6991	0.46119	6.4030	0.37605
CPOM	1.0552	0.07427	3.0761	0.24873
FPOM	0.9663	0.06606	1.6798	0.12062
Seston	3.2635	0.30773	1.8979	0.12373
Zooplankton	1.8994	0.12653	4.9251	0.36182
Corixidae A	2.8676	0.35383	3.9493	0.59217
Trichoptera	3.8689	0.69475	3.7001	0.60064
Cherax sp.	2.0281	0.12292	2.1139	0.09872
Spencerhydrus sp.	2.8206	0.49635	1.4452	0.14035
Epiproctophora	6.2417	0.87792	4.9838	0.62775
Notonectidae A	2.7091	0.21384	1.5477	0.15759
Notonectidae N	1.6604	0.35089	4.5568	0.48847
Macrobrachium sp. L	1.4296	0.07389	3.3481	0.16925
Macrobrachium sp. S	1.7100	0.07706	3.3464	0.13382
C.carpio	2.1284	0.14322	1.9987	0.14210
G.holbrooki	1.7998	0.09001	3.6153	0.17316
L.unicolor	1.6401	0.18771	3.4569	0.13388
M.ambigua	2.1007	0.12087	3.2475	0.16650
N. erebi L	1.0980	0.07996	3.6165	0.27623
N. erebi L B	1.4761	0.11741	3.5160	0.44913
N. erebi M	1.2564	0.06971	3.2110	0.41518
N. erebi S	1.4450	0.09093	3.4344	0.27910

Appendix 45: Mean Euclidean distances between $\delta^{15}N$ values of each food web component at natural wetlands (NW) and typical storages (TS).

	NW		**TS**	
	Mean	**s.e.**	**Mean**	**s.e.**
Algae	2.9654	0.22035	2.9343	0.17090
CPOM	2.2930	0.17185	2.5539	0.17569
FPOM	2.2380	0.16108	1.2271	0.09825
Seston	2.0280	0.13295	3.6141	0.24277
Zooplankton	2.0542	0.15512	1.5087	0.11007
Corixidae A	2.3127	0.27677	2.5210	0.42603
Trichoptera	1.7492	0.28220	3.1121	0.50659
Cherax sp.	1.2138	0.05913	1.5432	0.07155
Spencerhydrus sp.	0.6509	0.12146	1.5709	0.15118
Epiproctophora	2.9545	0.36627	1.9130	0.23925
Notonectidae A	1.2231	0.09164	2.1645	0.21857
Notonectidae N	2.4523	0.56067	0.9741	0.11056
Macrobrachium L	1.1340	0.06130	1.7686	0.08932
Macrobrachium S	1.0811	0.04445	1.2396	0.05112
C. carpio	1.3680	0.09722	2.1104	0.15362
G. holbrooki	1.6996	0.08786	2.2581	0.11048
L. unicolor	1.5879	0.14597	2.2739	0.08859
M. ambigua	1.5341	0.07877	1.6874	0.09210
N. erebi L	1.0694	0.07257	2.9060	0.22079
N. erebi L B	1.4189	0.17904	1.9734	0.22346
N. erebi M	1.1960	0.05788	1.4348	0.11700
N. erebi S	1.5259	0.07276	1.8835	0.12646

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Supplementary materials: publications arising from thesis

The conceptual model and explanation presented in Chapter 1 formed the basis of an original refereed conference paper, published in the proceedings of the 9th Annual Environmental Research Event Hobart, Tasmania, 29 November – 2 December, 2005. It is attached below in its original published format.

Drivers affecting the aquatic biodiversity and the ecological value of water storages on irrigation properties: a conceptual model

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Abstract

Floodplains along river systems of the northern Murray-Darling Basin contain a diversity of wetland habitats, maintained through highly variable patterns of flood inundation. However, in catchments such as the Border Rivers, the development of irrigation farms on floodplain areas and subsequent flow regulation has reduced the extent and frequency of floods and led to a decrease in the distribution of wetlands. Due to the flow variability of the system, irrigators use on-farm water storages to store water until required for irrigation. With the decline in natural wetlands, these water storages may now represent a key form of artificial aquatic habitat within the Border Rivers. A conceptual model was developed to explore the drivers affecting the biodiversity of on-farm storages. It is believed that ring tanks with a greater selection of habitat types and more complex morphology will support a more abundant and diverse animal community. Potential management options to improve storage biodiversity include adding coarse woody debris to the banks, planting aquatic vegetation and limiting recycled tailwater to only one storage on the property.

Keywords: biodiversity, Border Rivers catchment, irrigation, modelling, on-farm water storages, wetlands

1. Introduction

Wetlands are an important part of floodplain ecosystems across semi-arid regions of Australia. In a flood event these wetlands are connected with the main river channel and, following floodwater recession, remain behind as isolated ecosystems. During the in between dry periods permanent wetlands provide vital refuge for aquatic plants and animals (1,2) and when flooding occurs, may provide an important source for colonisation of temporary waterbodies (3) and the adjacent river channel.

In the northern Murray-Darling Basin, water resource development has led to many natural wetlands contracting in size (4) or disappearing altogether (5). Along the Macintyre River, there is evidence that the growth of irrigation and river regulation has had significant implications for patterns of floodplain inundation, both by reducing the magnitude of flow events and the flooding frequency of certain channels (6,7). Regulation of Whalen Creek, an anabranch of the Macintyre that supplies water to numerous wetlands and billabongs, has reduced flows to this creek by between 30-70% (5). With this decline in natural wetlands, artificial waterbodies (on-farm water storages) on the floodplain may now represent a key form of aquatic habitat.

There has been little investigation into the ecological value of on-farm storages in Australia to fish and macroinvertebrate populations. Jarman and Montgomery (8) investigated the use of on-farm wetlands and storages by waterbirds in the Lower Gwydir Valley and Bowmer *et al.* (9) considered the potential of irrigation drains to be used as wetlands. Brooks (10) looked at the macroinvertebrate community structure in irrigation channels in the Gwydir Valley, but concentrated on possible relationships between pesticide concentrations and macroinvertebrate communities. No comparisons with natural wetlands were made. In other parts of the world artificial wetlands have been recognised as a possible alternative habitat. In the US for example, Knutson *et al.* (11) found that constructed farm ponds may help sustain amphibian populations in landscapes where natural wetland habitat is rare.

There is the potential that, if properly managed, on-farm water storages in the northern Murray-Darling Basin may represent alternative habitats and refuge for fish and macroinvertebrates in the region. Effective management to maximise the ecological value of these on-farm storages will require an understanding of the processes at work in nearby natural wetlands and the factors affecting fish and macroinvertebrate populations. The present study aims to investigate these factors and identify possible drivers of population structure in on-farm storages. The findings will be used to outline management options to add to the Best Management Practices (BMP) manual for the cotton industry and will also have transferable applications to farmers wishing to use their on-farm storages for aquaculture purposes.

2. Study Area

The present study was undertaken in the Border Rivers catchment of north west New South Wales and southern Queensland. The first broadacre flood irrigation of cotton in the catchment occurred in the late 1970s and today this industry is of major economic importance, employing a large (approximately 2,500) number of people (12). Broadacre irrigation is one of the main users of water within the catchment, extracting an average of 40% of the total Border Rivers inflows (13). Summer rainfall patterns dominate the region, with 55% of the annual rainfall occurring from November to March. However, annual median rainfalls vary considerably. For example, records from Boggabilla show that annual rainfalls have ranged from a low of 174 mm in 1902 to a high of 1,041 mm in 1950 (4). Due to this variability, irrigators use large on-farm storages to store water pumped from rivers during regulated and unregulated flows, collect rainfall runoff and receive recycled tailwater. In 2002, on-farm storages across the catchment had a total capacity of approximately 459,000 ML (B. McCollum, pers. comm.).

There are many significant wetlands in the Border Rivers, particularly downstream of Boggabilla where the majority of broadacre irrigation occurs. These wetlands include lagoons or water holes in flood channels, such as the Telephone/Malgarai, Rainbow, Maynes and Boobera lagoons, Whalan Creek and the Morella Watercourse (4). There are also numerous anabranch channels and associated billabongs along the Macintyre River channel itself (6). The larger lagoons have naturally high commence-to-flow levels and generally contain some permanent water.

3. Typical On-farm Storages

A large number of constructed on-farm storages occur in the Border Rivers catchment, built to overcome the problem of an unreliable natural water supply. Water stored in these on-farm storages is mainly used for the irrigation of row crops, particularly of cotton. Site visits were carried out to survey characteristics of 100 of these storages and interviews with landholders provided information about management practices, storage dimensions and water sources. A two-way cluster analysis of the water source and morphological data found that the majority could be placed in one group (Table 1). This group comprised storages containing water sourced from allocated flows, unallocated flows and overland flows and also recycled tailwater. Site visits revealed that the majority had little or no vegetation growing on the embankment and aquatic vegetation was also rare. The banks were devoid of snags and substrate was generally a fine, clay sediment. As most of the storages in the catchment were in this group, it was these characteristics that were used to construct a conceptual model of the factors driving the biodiversity of on-farm water storages in the region (Fig 1). These are hereafter referred to as ‘typical storages’.

Table 1: Mean values for characteristics of typical on-farm water storages in the Border Rivers catchment

	Age (Years)	Height (m)	Area (Ha)	Perimeter (km)	Capacity (ML)	Distance from source (km)*
Mean	9.05	4.66	38.44	2.51	1553.24	2.71
SE	0.888	0.114	2.287	0.076	98.75	0.353

*Distance from source = distance to river from which flows are pumped

4. Conceptual Model – regulation of biodiversity between wetlands and storages

4.1 Riparian vegetation

The majority of on-farm storages visited had little or no vegetation on the embankment. Many property holders use broad-spectrum herbicides to kill any vegetation that emerges on the embankment or regularly grade the top of the embankment while others use selective herbicides to kill broad leaf species but allowing grasses to grow. The reasoning behind this is to prevent damage to the wall of the embankment from growing roots, which may cause leaks in the storage. In contrast natural wetlands have a multi-layered band of vegetation along their edge (14). The importance of linkages between the riparian zone and the aquatic ecosystem has been well documented (15, 16,17). Riparian vegetation adds organic carbon to the aquatic food web in the form of litter (e.g. leaves, bark, fruit: 18, 17).

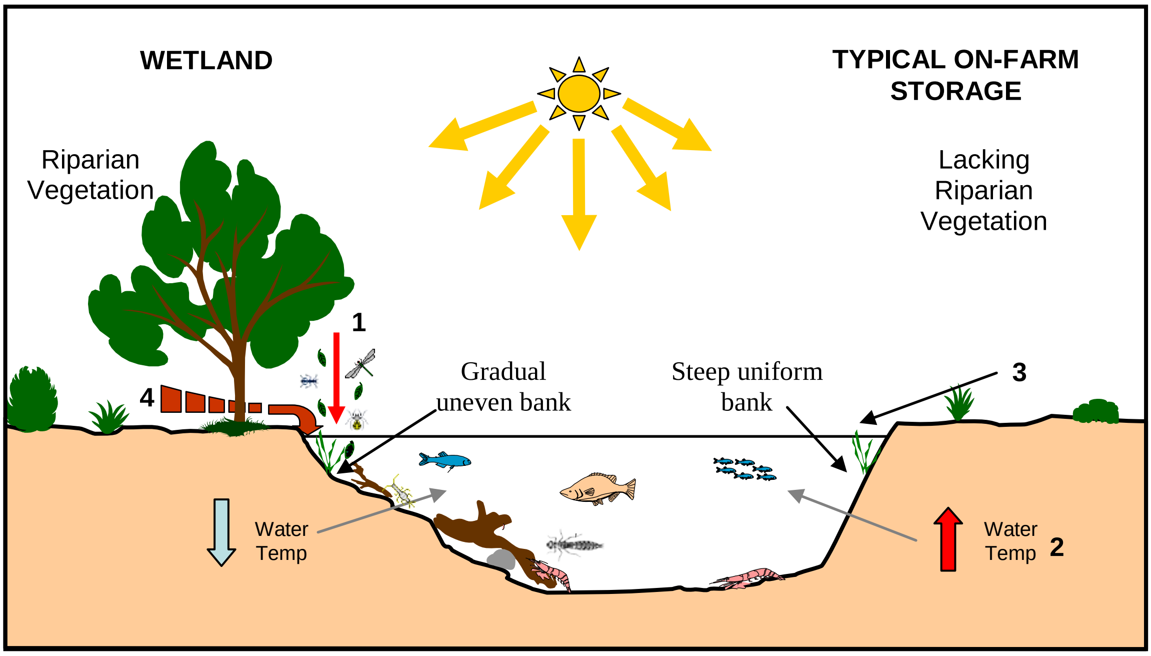

Fig 1: Conceptual model comparing potential drivers affecting biodiversity between natural wetlands and on-farm water storages. 1. Inputs of leaf litter and terrestrial invertebrates. 2. Rise in water temperature due to lack of shading. 3. No source of logs and branches. 4. Filtration of sediments and nutrients (adapted from 19)

Native vegetation provides inputs of leaf litter all year round, with a slight peak around late summer, providing a continuous food source for macroinvertebrates (20). Terrestrial invertebrates falling from riparian vegetation also add energy and nutrients to the aquatic food web (18). The lack of riparian vegetation on constructed storages will lead to a reduction in the amount of energy and nutrients from leaves and terrestrial invertebrates available to the aquatic food webs therein. As well as leaf litter, branches and whole trees may fall into the water, providing important habitat for fish, plants and macroinvertebrates (15, 18, 21). The general absence of trees on storage embankments means that there is no source of coarse or large woody debris. This was apparent from site visits, with the majority of storages being devoid of snags on their banks.

Riparian vegetation also provides shading during the day, moderating fluctuations in water temperature. Where riparian vegetation is absent, water temperatures are likely to rise affecting the balance of algal and aquatic plant communities and their contribution to energy production (22). This also results in a shift in aquatic production dominated by microalgae such as diatoms to larger, less palatable species such as green algae and macrophytes (19). In natural wetlands, riparian vegetation also acts as a buffer, filtering sediments and nutrients before they reach the water. The importance of this to on-farm storages is likely to be minimal as typical storages are raised above the ground and, therefore, the only run-off will be off the top of the embankment itself.

4.2 Morphology

The morphology of aquatic ecosystems such as surface area, volume, perimeter, depth and channel complexity is thought to affect the overall health of the ecosystem. Thoms *et al.* (23) found that in dryland rivers, increased channel complexity increased the surface area available for organic matter to provide a food source and the amount of habitat for lower order aquatic organisms. Morphological differences between natural wetlands and the more uniform on-farm storages could lead to differences in species composition, diversity and abundance.

Benthic algae have been found to be extremely important contributors to waterhole (24) and lake (25) food webs. In dryland river waterholes, as found in the Border Rivers, the water is very turbid restricting the photosynthesis of benthic algae to a narrow photic zone around the bank. The steeper the slope of the bank, the smaller the width of this photic zone (24). If the slope of on-farm water storages is steeper than in natural wetlands it is likely that rates of primary production will be reduced. Keast and Harker (26) also found that fish biomass and, to a lesser extent, benthic invertebrate biomass was concentrated towards the margins of a lake up to a depth of 2.5 m. Fish distribution has also been associated with lake morphological characteristics. Jackson and Harvey (27) found that large, deep lakes differed from shallow lakes in their fish species diversity. Olden *et al.* (28) stated that surface area, volume and shoreline perimeter were positively correlated with habitat diversity which in turn affects species diversity (29). Unfortunately evaporation from storages can be a significant loss to irrigation farms and property holders will be aiming to reduce the amount of surface area to minimise these losses.

4.3 Habitat availability and substrate type

The amount and type of habitat available is often considered as a primary limiting factor in population and community recovery in degraded ecosystems (30). The availability of habitat is believed to be important to fish and macroinvertebrate communities for a number of reasons. Habitat can provide shelter from the elements and refuge from predators while providing suitable spawning sites for certain species and in some cases can also be a source of food. Both species abundance and diversity have shown positive correlations with the abundance of coarse woody debris (CWD) (31) with CWD providing a refuge from predation for epibenthic fish and invertebrates (32). Fish abundance has been found to increase when CWD has been added to streams (33). The complexity of the habitat available has also been observed to be positively correlated with the number of macroinvertebrate species present. Addition of grooves to artificial snags increased the number of macroinvertebrate species associated with them (34).

The presence of hard substrate matter has been found to increase the number of benthic diatom species found in dryland river waterholes with species numbers double that found on soft substrates (35). The lack of hard substrates in on-farm storages could affect the amount of organic carbon available to their aquatic food webs. Several studies have also found that substrate characteristics are a major factor in determining the structure of freshwater macroinvertebrate communities (36, 37, 38). Although it is possible that response to substrate will depend on the species (37), in general, high proportions of fine substrate were found to reduce species richness (38). However, these studies were carried out in either streams or rivers. In lakes, ponds and ditches, which are more comparable to storages, the habitat preference of macroinvertebrates has been found to be influenced chiefly by the vegetation where there was a definite relation between vegetation and the distribution of macroinvertebrates (39). As the majority of storages had very little or no aquatic vegetation, it is likely that substrate composition will still be an important factor in determining macroinvertebrate species composition.

4.4 Source of water

The source of the water used to fill storages is likely to have implications for biodiversity and water quality. Floodplain wetlands are filled during overland flows when water runs across the land after rainfall, either before it enters a watercourse or after it leaves a watercourse as floodwater or after it rises to the surface naturally from underground. In contrast, storages can also be filled using tailwater and water pumped from the river during regulated and unregulated flows.

When fields are irrigated, any excess water is collected in the tail drain and known as tailwater. Tailwater is then recycled back to the storage dam or supply channel where it can be used for further irrigations. As the water travels along the crop rows it is likely to pick up chemical residues from recent spraying, either in solution or adsorbed onto soil particles. Endosulfan is the most well known pesticide in the cotton industry but is only one of many used throughout the season. Brooks (10) investigated two cotton properties in NSW and found that levels of endosulfan in the water column were always above the 1992 ANZECC trigger value for 99% of ecosystem protection of 0.1 μg/L (40). However, concentrations were generally lowest in the storage dams compared with the irrigation channels. This study was also carried out before guidelines were introduced for the 1999/00 cotton season to limit endosulfan use. Other potential contaminants include herbicides, other insecticides, defoliators, wetting agents, conditioners and fertilisers. Some of these are likely to reduce algae in storages which are an important base of food-webs in dryland river catchments.

Water accessed during unregulated and regulated flows is usually pumped from the river either directly into the storage or via a supply channel. Any fish or macroinvertebrates in the pumped water pass through at least one pump before entering the storage. Due to their size it is more likely that fish will be injured or killed as they go through the pump, although numbers of fish do survive this process and become established in the storage. These include carp, bony bream, spangled perch and yellowbelly

(S. Lutton, unpublished data). There are a number of fish screens which can be used to prevent the movement of fish into irrigation offtakes, although they are uncommon in Australia (41).

4.5 Stock access

Livestock watering and grazing are known to have detrimental effects on natural wetlands. Jansen and Healey (42) found that grazing stock reduced the quality and availability of aquatic vegetation. This led to a decrease in the abundance and diversity of frog communities in Australian floodplain wetlands due to changes in habitat quality. Breakdown of banks by cattle trampling has led to increased erosion along streambanks and subsequent effects on water quality (43). Godwin and Miner (44) found that offstream watering can reduce water quality impacts from manure which is a source of bacteria, nitrogen and phosphorous. A number of property holders allowed livestock to access the storages on their property and this may be having an effect on the already limited vegetation and on water quality.

5. Discussion

Storages differ intrinsically from natural wetlands by their constructed nature and lack many of the attributes that native species have become adapted to, such as coarse woody debris, riparian vegetation and macrophytes. These differences are likely to result in on-farm storages having a lower biodiversity of fish and macroinvertebrates than natural wetlands. However, storages represent a significant proportion of the floodplain aquatic habitat in the Border Rivers catchment and changes could be made to utilise this more effectively.

Altering the morphology of existing storages would most likely be impractical and cost prohibitive, although could be addressed during the design and construction of new storages. Planting large riparian vegetation may lead to damage to the embankment and storage walls and is not likely to be undertaken by many property holders. However, there are a variety of different management options that could be put into place relatively easily. Planting aquatic vegetation is one option that could be implemented to improve the biodiversity value of on-farm storages by providing habitat and a food source. Coarse woody debris could also be added to the inner banks of the storage to provide a hard substrate and further habitat. It is possible that the presence of chemicals in tailwater may be having a detrimental affect on aquatic species. On properties where there is more than one storage, tailwater returns could be limited to only one to reduce chemical contamination of the remaining storages. To prevent damage to the bank and any vegetation present, property holders should restrict the access of livestock to the storage.

The benefits of implementing these management options not only include improving the aquatic biodiversity of the storage but increasing habitat availability for mobile fauna such as waterbirds, and boosting the potential for aquaculture production. In order to test the predictions of this conceptual model an aquatic sampling programme comparing storages of different characteristics and natural wetlands is required. Once management options are undertaken, ongoing monitoring to record any changes in biodiversity will be essential.

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