Chapter 5 Benthic community response to flow and temperature in riffles

5.1 Introduction

Investigations that have used specific relationships between flow velocity and discharge to assess the impact of flow reduction on invertebrate fauna have found that species preferences for various flows could be used to develop a minimum flow level that would mitigate the effects of flow reduction (Fig. 5.1) (Gore 1978, Brunke et al. 2001). Invertebrate responses to flow reduction vary depending on the magnitude, duration and seasonality of the decline, the presence of other stressors and the characteristics of the affected system (Suren et al. 2003, Scherman et al. 2003, Boulton et al 2003, Suren and Riis 2010, Brooks et al. 2011a,b).

Macroinvertebrate responses to different flow velocities indicate that some species prefer particular habitat types (Brooks et al. 2011a), with water velocity a principal factor controlling faunal variation between mesohabitats (Pardo and Armitage 1997). Reduction in velocity may result in the loss of species from reaches under low flow conditions (Dewson et al. 2007b, Suren and Riis 2010). The impact of low flows on benthic macroinvertebrate communities may be magnified under increased silt deposition and loss of specific habitat types such as gravel runs or rock rapids in New Zealand streams (Dewson et al. 2007b, James et al. 2008). Brooks et al. (2011a) inferred the degree of possible impacts on macroinvertebrate taxa from water extraction using differences between predicted and observed assemblages from sites affected by upstream water extraction in a NSW coastal river. These results indicate that many riffle dependent species have a high likelihood of an adverse effect from water extraction (Brooks et al. 2011a).

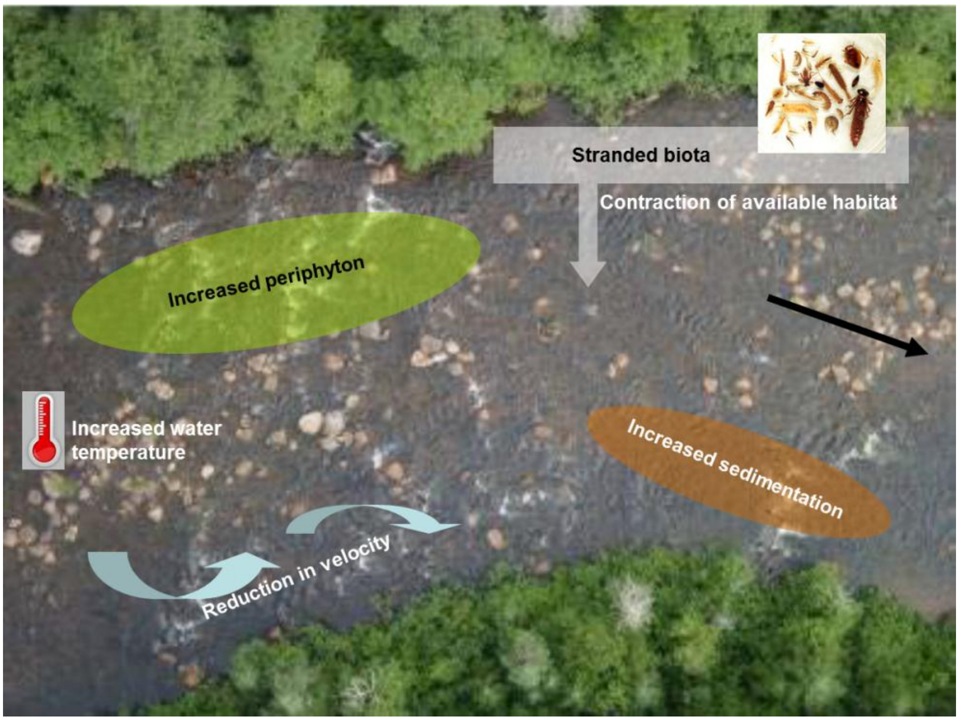


Figure 5.1 Impacts to riffle habitat under low flow conditions

Short-term discharge reductions have resulted in the accumulation of invertebrates in the decreased available habitats of small New Zealand streams, thereby increasing local invertebrate density (Dewson et al. 2007a, James et al. 2008, Sheldon et al. 2010). When studying Swedish streams, Englund and Malmqvist (1996) found that increasing flow expanded the habitable area within a riffle and therefore the production of lotic invertebrates. Other studies found that persistent drought and increased extraction have, in some cases, shown major reductions in invertebrate densities and recruitment in a number of systems both small and large (Wood and Petts 1994, Boulton 2003, Datry 2012). In the case of Californian Mediterranean-climate streams, most macroinvertebrate species were found to become temporally rare in their annual occurrence, reflecting variations in stream flow (Resh et al. 2013). Atkinson et al. (2014) found drought caused a large reduction in freshwater mussel populations in Oklahoma streams, particularly in areas with less forest cover. This loss of mussel biomass reduced nutrient recycling and storage, potentially altering nutrient availability within streams (Atkinson et al. 2014).

Changes in water quality and the degree of nutrient enrichment within a river are known to influence macroinvertebrate response during periods of low flow (Boulton and Lake 1992, Doyle et al. 2005, Lancaster et al. 2009, Baldwin et al. 2010) (Table 5.2). Studies suggest that water quality, particularly the degree of nutrient enrichment within a river, should be taken into account when assessing minimum flow requirements (Scherman et al. 2003, Gafner and Robinson 2007). Periphyton growth, particularly an increase in filamentous algae, may be a significant factor in how flow changes alter macroinvertebrate communities, with populations of more sensitive species such as mayflies and caddisflies declining while more tolerant species such as chironomids dominate (Biggs and Stokseth 1996, Suren et al. 2003, Haxton and Findlay 2008).

Table 5.1 Ecological consequences of high nutrient concentrations in riffles

Impact Type	Taxa	Ecological consequence	References
Experimental nutrient enrichment	Chironomidae	Domination of benthic macroinvertebrate population by Chironomidae (43%)	Blumenshine et al. 1997
Agriculture	Algae, Macroinvertebrates and Fish	Periphyton increased with TN, dominant invertebrate taxa and herbivorous fish increased with periphyton, sensitive invertebrate taxa decreased with increased periphyton	Griffith et al. 2009
Point source pollution (STP)	Macroinvertebrates	Macroinvertebrate density increased and richness decreased downstream of point source, EPT species were only present upstream of point source	Ortiz and Puig 2007
Agriculture and urbanisation	Macroinvertebrates and Fish	Fish Index of Biotic Integrity decreased with increasing nutrients, Macroinvertebrate richness and EPT individuals declined with increasing nutrients	Wang et al. 2007
Agriculture and flood pulse	Macroinvertebrate	Macroinvertebrate taxa abundance was more variable and dominated by vagile spp (e.g. chironomids) following flood pulse	Collier and Quinn 2003

River macroinvertebrate fauna appear to be more resistant to the effects of changed flow regimes if water quality condition did not deteriorate (Durance and Ormerod 2009, Lancaster et al. 2009, Dunbar et al. 2010). A study of two rivers in Victoria found that macroinvertebrate assemblages were altered in the reaches of one river due to the combined effects of decreased flow and longitudinal increases in salinity, while the macroinvertebrate fauna of another river appeared to be resistant to the effects of reduced flow, owing to limited decline in water quality (Lind et al. 2006). While research continues into the impacts of low flows, there appears to be a lack of reporting on primary ecosystem functions in relation to reduced flows (Poff and Zimmerman 2010) and a need for greater understanding of the associated ecological consequences (Rolls et al. 2011).

Seasonal variations exhibited within macroinvertebrate populations are also affected by changes in the natural flow regime (McMahon and Finlayson 2003). To maintain diverse invertebrate assemblages a range of flow conditions at various times is required to provide optimal hydraulic conditions for different groups of species (Gore et al. 2001). To assess the presence of these optimal conditions, an accurate characterisation of available habitats and associated flows is needed. A better understanding of seasonal changes in stream temperature and the associated changes in macroinvertebrate densities, including flow-dependent changes in macroinvertebrate and larval drift in altered flow environments is considered an important component in understanding minimum flow requirements (Gore et al. 2001). The importance of hydraulic characteristics, found to be useful in modelling the distribution of macroinvertebrates, will also vary depending on season and the development stage of organisms (Statzner et al. 1988). The importance of hydraulic characteristics in understanding flow requirements is further demonstrated in the "ecological limits of hydrologic alteration" (ELOHA) developed by Poff et al. (2010). This framework uses existing hydrologic techniques and environmental flow methods to support flow management at a regional scale (Poff et al. 2010).

Micro-habitat and hydraulic characteristics are thought to be significant factors in the structure of macroinvertebrate communities in riffle areas, through influencing their metabolism, feeding and behaviour (Lancaster and Hildrew 1993). The distribution of invertebrates has been found to significantly correlate with hydraulic variables, while hydraulic conditions are thought to represent a major physical gradient along which the benthic community is organized (Rempel et al. 2000) (Table 5.2). Lancaster and Hildrew (1993) determined that changes in macroinvertebrate microdistribution in an English stream were not a reaction to individual flow events, but a longer-term response to seasonal flow conditions. Evans and Norris (1997) investigated the influence of microhabitat variables on macroinvertebrate distribution and abundance in an Australian alpine stream, and found rock height, length, area and flow velocity were major determining factors in the macroinvertebrate community assemblage.

A study conducted in the unregulated Kangaroo River on the south coast of NSW, sampled macroinvertebrate communities within riffle habitats to assess if distribution related to small-scale differences in hydraulic conditions (Brooks et al. 2005). The study found that these differences were created by combinations of velocity, depth and substrate roughness and have an important role in the spatial distribution of macroinvertebrate assemblages in riffle habitats (Brooks et al. 2005). This finding highlights the importance of maintaining a variety of flow conditions within a riffle area. When flow volume is reduced, so too is the area and variety of conditions within a riffle area. This lack of variability reduces the area of favourable habitat for macroinvertebrates reliant on riffles.

Another study, also conducted in the Kangaroo River, investigated a number of issues relating to wetted perimeter breakpoints including the selection of appropriate normalising flows (Reinfelds et al. 2004). Breakpoint describes the point at which there is a break in discharge curves, below which wetted perimeter declines rapidly (Gippel and Stewardson 1998). The wetted perimeter–discharge relationship is a basic tool in the 'transect' approach and has been commonly used to assist with decisions regarding cease-to-pump thresholds and minimum environmental flows (Reinfelds et al. 2004). The study found that riffle wetted perimeter-discharge breakpoint analysis has merit and, when used to complement other techniques, can prove useful in setting environmental flows or cease-to-pump thresholds.

Table 5.2 Review of macroinvertebrate responses to velocity, duration and water temperature [3]

Order	Family	Flow category (m s^{-1})			Habitat preference	Likelihood of adverse effect by water extraction	Author
		<0.1	0.1-0.25	>0.25			
Trichoptera	Hydropsychidae	✓	✓	✓	Gravel runs/rock rapids; Present in high and low gradient riffles	Low	Brunke et al. (2001); Rabeni et al. (2002); Brooks et al. (2011)
	Glossosomatidae	✓	✓	✓	Gravel runs/rock rapids; Present in high and low gradient riffles	High	Poff and Ward (1992); Brooks et al. (2011a)
	Hydroptilidae				Gravel runs/rock rapids; Feeds on attached diatoms	Low	Tall et al. (2006); Humphries et al. (1996); Brooks et al. (2011a)
	Leptoceridae				Edge and run habitats; Macrophytes	High	Pardo and Armitage (1997); Humphries et al. (1996); Brooks et al. (2011)
Ephemeroptera	Baetidae		✓	✓	Gravel and higher velocities; Avoids sand/mud/silt	High	Rabeni et al. (2002); Ambuhl (1959); Storey and Lynas (2007); Brooks et al. (2011a)
	Caenidae	✓	✓	✓	High and low gradient riffles; Avoids sand/mud /silt	High	Rabeni et al. (2002); Storey and Lynas (2007); Brooks et al. (2011a)
Odonata	Gomphidae	✓			Macrophytes; Present in high and low gradient riffles	Low	Rabeni et al. (2002); Storey and Lynas (2007); Brooks et al. (2011a)
	Libellulidae	✓			Macrophytes; Present in edge waters	Low	Rabeni et al. (2002); Storey and Lynas (2007); Brooks et al. (2011a)

[3] Note: These responses do not not apply to all representatives of the family

Order	Family	Flow category (m s^{-1})			Habitat preference	Likelihood of adverse effect by water extraction	Author
		<0.1	0.1-0.25	>0.25			
Megaloptera	Corydalidae			✓	High gradient riffles	High	Rabeni et al. (2002); Brooks et al. (2011a)
Coleoptera	Psephenidae	✓	✓	✓	High and low gradient riffles Avoids sand/mud /silt	Moderate	Rabeni et al. (2002); Storey and Lynas (2007); Brooks et al. (2011a)
	Elmidae				Gravel riffles; Feeds on attached diatoms	High	Storey and Lynas (2007); Tall et al. (2006); Pardo and Armitage (1997); Brooks et al. (2011a)
	Hydrophilidae				Edge; Avoids sand/mud /silt	High	Storey and Lynas (2007); Brooks et al. (2011a)
Plecoptera	Gripopterygidae				High and low gradient riffles	Moderate	Humphries et al. (1996); Brooks et al. (2011a)
Diptera	Chironomidae	✓	✓	✓	Mud/silt/sand/gravel; High and low gradient riffles	Low	Rabeni et al. (2002); Storey and Lynas (2007); Brooks et al. (2011a)
	Simuliidae			✓	Gravel runs/rock rapids; High gradient riffles	Moderate	Rabeni et al. (2002); Storey and Lynas (2007); Brooks et al. (2011a)
Gastropoda		✓			Macrophytes Substrate	Moderate -High	Rabeni et al. (2002); Storey and Lynas (2007); Brooks et al. (2011a); Atkinson et al. (2014)
Oligochaeta		✓	✓		Mud/silt/sand; Low gradient riffles		Rabeni et al. (2002); (Pardo and Armitage (1997)
Hydracarina		✓	✓	✓	Macrophytes; High and low gradient riffles		Rabeni et al. (2002); Storey and Lynas (2007)

Gippel and Stewardson (1998) also looked at relationships among discharge, wetted perimeter, flowing water perimeter and blackfish habitat area in two regulated headwater streams located in the Melbourne catchment area. They found that, for these streams, the wetted perimeter relationships did not provide an optimum environmental flow, nor did it suggest a flow level that would maintain the macroinvertebrate community in its unregulated state if it was applied for a long period of time. They suggest that flowing water perimeter was preferable over wetted perimeter as a variable to define habitat suitable for macroinvertebrates.

Habitat methods are a natural extension of hydraulic methods with the assessment of flow requirements based on hydraulic conditions that meet specific biological requirements (Jowett 1997). Habitat suitability curves are the biological basis of habitat methods. Habitat suitability can be specified as seasonal requirements for different life stages. When considering multiple species, there can be conflicting habitat requirements (Table 5.2), with a decline in habitat for one species corresponding to an increase in habitat for another. In this case, habitat guilds or indicator species can be applied (Jowett and Richardson 1995). Typically, the flow-related changes in microhabitats are determined using data based on one or more hydraulic variable such as depth, velocity, substratum composition and sometimes more complex hydraulic indices. These variables are collected at multiple cross-sections within the river study reach. The simulated available habitat conditions are linked with information on the range of preferred to unsuitable microhabitat conditions for target species, life stages, assemblages, and are often depicted using seasonally defined habitat suitability index curves. These results can then be used to predict optimum flows.

However, Jowett's (1997) review of instream flow methods found differences between methods did not imply that one may be right and the other wrong, rather some methods may be better suited to some conditions than others. To do this effectively, an understanding of the morphological implications and ecological assumptions that underlie methods and management needs is required (Jowett 1997). In the development of the ELOHA framework, Poff et al. (2010) utilise available scientific information to develop ecologically based and socially acceptable goals and standards for the improved management of environmental flows at a regional scale. Poff and Zimmerman (2010) found that global literature could not be used to develop general quantitative relationships between changes in flow regime and ecological response. To better inform environmental flow management it was recommended that sampling programs targeting sites across gradients of flow alteration are developed to quantify ecological response to changes in flow (Poff and Zimmerman 2010).

Higher velocity habitats have been identified as crucial areas for macroinvertebrate diversity and therefore riffle maintenance is an important issue with regard to flow. Stream hydraulics, and therefore flow, is seen as a major determinant of macroinvertebrate community structure. The influence of hydraulic conditions on the distribution of macroinvertebrates has been established over larger spatial scales and between rivers of different sizes. However, there have only been a few studies investigating the relationship between hydraulic conditions and macroinvertebrates at a micro-scale.

During the development of the macro water sharing plan for the lower north coast unregulated and alluvial water courses, a need was identified for improved understanding of the river system to identify habitat values requiring protection under environmental flows (Bishop and Thurtell

2006, NSW Office of Water 2011). This would assist in the development of appropriate indicators that were reliant on habitats requiring protection under low flow conditions.

While some work has been undertaken in Australia on aspects such as the impact of flow on fish passage, much less has been completed on the impacts of flow on macroinvertebrates. Impacts on macroinvertebrates can be particularly relevant as flow influences habitat availability and changes in velocity and, as such, their distribution and abundance can be a useful tool for environmental flow setting and assessment (Growns 1998).

Decreases in the availability of critical physical habitat features have been identified by Bishop (2006) as relevant to the lower Manning system and potentially influenced by the Midcoast Water (MCW) Bootawa Dam pumping site and irrigators. While water quality is also thought to add to the impacts of low flows and disconnection, high concentrations of nutrients, resulting in excessive algal growth is thought to further exacerbate low flow influences on macroinvertebrate habitats. Responses of macroinvertebrates to low flows and possible cumulative impacts from increased nutrients sites in the lower Manning and a number of tributary sites with a range of nutrient concentrations were examined as part of this study.

As cobbled-based riffles were found to be the most common high-velocity habitat within the lower Manning River in terms of occurrence and surface area (Bishop 2005), the effect of flow on these areas requires investigation. These areas are also recognised as important habitats for macroinvertebrates, providing for much of the diversity within river systems.

The previous water quality study of “reference” sites in the Manning indicated that many tributaries had lower nutrient levels than that found in the mid to lower reaches of the Manning River (see Chapter 3). These sites included those that contain riffles which exhibited similar physical characteristics as macroinvertebrate sampling sites used in the Manning. An investigation of how macroinvertebrate communities respond to low flows at these sites which have lower nutrient levels in comparison to the lower Manning River may indicate that elevated nutrients may exacerbate the impact of low flows. This particularly relates to how land management and increased nutrients in a catchment will influence appropriate minimum flow thresholds for a particular river.

The investigation within the reference sites addresses a question posed as part of the thesis which refers to the examination of ecological responses to flow through measuring periphyton and macroinvertebrate responses to low flow conditions within riffles, under various nutrient regimes. Results from this investigation can then provide some guidance on the management implications of land use and water extraction and how management actions influence ecological responses within the Manning River.

5.1.1 Aims and hypotheses

This chapter examines ecological responses to low flow through the measurement of periphyton and macroinvertebrate responses within riffles, under various nutrient regimes. As other studies have indicated, flow alone may not be the most critical factor in macroinvertebrate community response. Other influences such as nutrients, temperature and relative sensitivities of taxa to those factors will determine the degree of ecological impact (Boulton 2003, Omerod et al. 2010, Poff and Zimmerman 2010, Vaughn 2010). While decreasing discharge results in a loss in aquatic habitat through a reduction in wetted area, velocity and depth, it may be changes to nutrient concentrations, increased water temperatures and lowered dissolved oxygen that have a greater influence on macroinvertebrate community structure (Gore 1977, Wood and Armitage 2004, Dewson et al. 2007b).

Hypotheses have been developed to assess spatial changes in riffle macroinvertebrate communities under summer low flows within different hydraulic classes and nutrient influences found in the Manning catchment. Firstly, it is predicted that macroinvertebrate diversity, abundance and community structure within riffles will decline under low flows. Secondly, spatial differences in macroinvertebrate community structure will be explained by differences in flow characteristics and nutrient concentrations. It is predicted that higher nutrient concentrations will further reduce macroinvertebrate taxa richness found in riffles as a result of increased periphyton biomass. Lastly, extraction during low flows will reduce macroinvertebrate community abundance and taxa richness by reducing habitat availability and variability.

5.1.2 Study sites

5.1.2.1 Manning River riffle sites

Six sites were chosen for this study. Three sites were selected upstream of major water users, and three sites downstream. Irrigators, predominantly producing pasture, are centred around Wingham on the lower Manning (between Killawarra gauge and Bungay – Fig. 5.2). MCW also has a pumping site located slightly upstream of the site labelled MCW riffle, extracting up to 24,822,000 $m^3 year^{-1}$ from the Manning River to fill Bootawa Dam, the main water supply for Taree, Forster, Tuncurry and other population centres in the region (Fig. 5.2).

Figure 5.2 Manning River macroinvertebrate riffle sites and location of Killawarra gauge site

Ida Lake riffle, the most upstream riffle site, is immediately upstream of Ida Lake (Fig. 5.3). This site is immediately downstream of the confluence of the Manning and Gloucester Rivers (Figure 5.2). The site upstream from major water extractors, but has some grazing and weed impacts. The riffle is approximately 40m above sea level within a confined floodplain (Table 5.3).

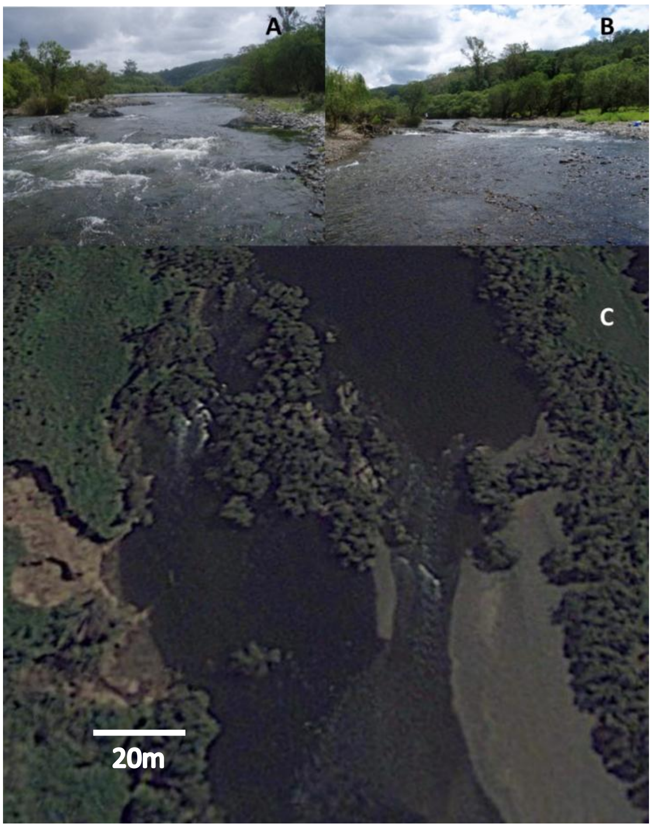


Figure 5.3 A. Ida Lake riffle in October 2008 (discharge ~ 7 m^3s^{-1} at Killawarra Bridge); B. Feb 2009 (discharge ~3.5 m^3s^{-1} at Killawarra Bridge)*; C. Aerial view
*Photographs taken looking downstream

Railbridge riffle, the second upstream site, is several kilometres downstream of Ida Lake and is referred to as Railbridge riffle (Fig. 5.2, Fig. 5.4). The riffle area is downstream of a shallow, macrophyte-dominated pool. Similar to the Ida Lake riffle, it is relatively unaffected by water extraction, but grazing and weeds have some influence at this site. Railbridge riffle is approximately 30m above sea level (Table 5.3).

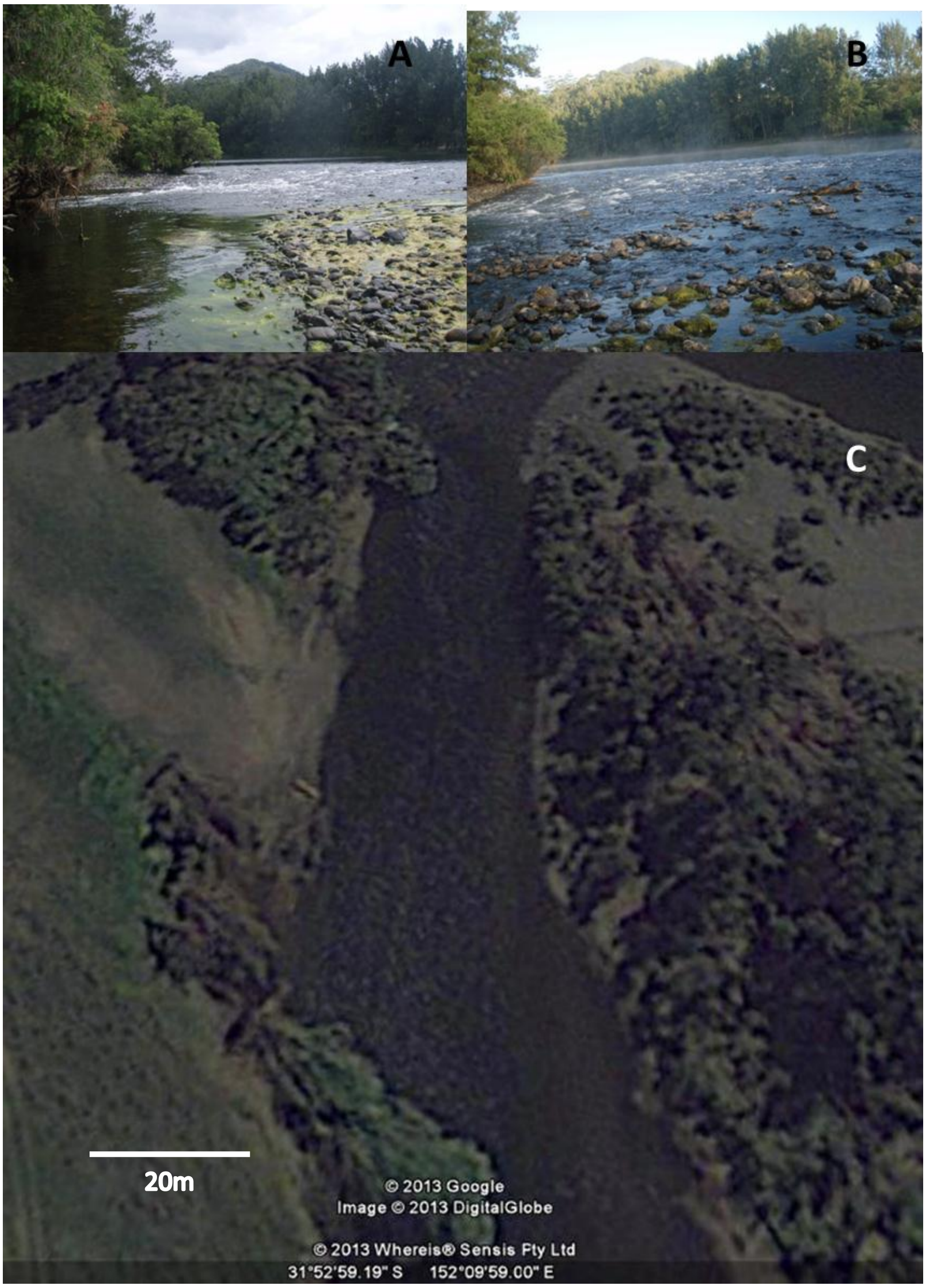


Figure 5.4 A. Railbridge riffle in October 2008 (discharge ~ 7 m^3s^{-1} at Killawarra Bridge); B. Feb 2009 (discharge ~3.5 m^3s^{-1} at Killawarra Bridge)*; C. Aerial view
*Photograph taken looking upstream

Woodside riffle is a few kilometres downstream of the Railbridge riffle and, as is the case with Ida Lake and Railbridge, is also above major water extractors (Fig. 5.2, Figure 5.5). This site is situated in a steep-sided, shaded section of the river, and also has weed and grazing impacts. Woodside riffle is located approximately 30m above sea level (Table 5.3).

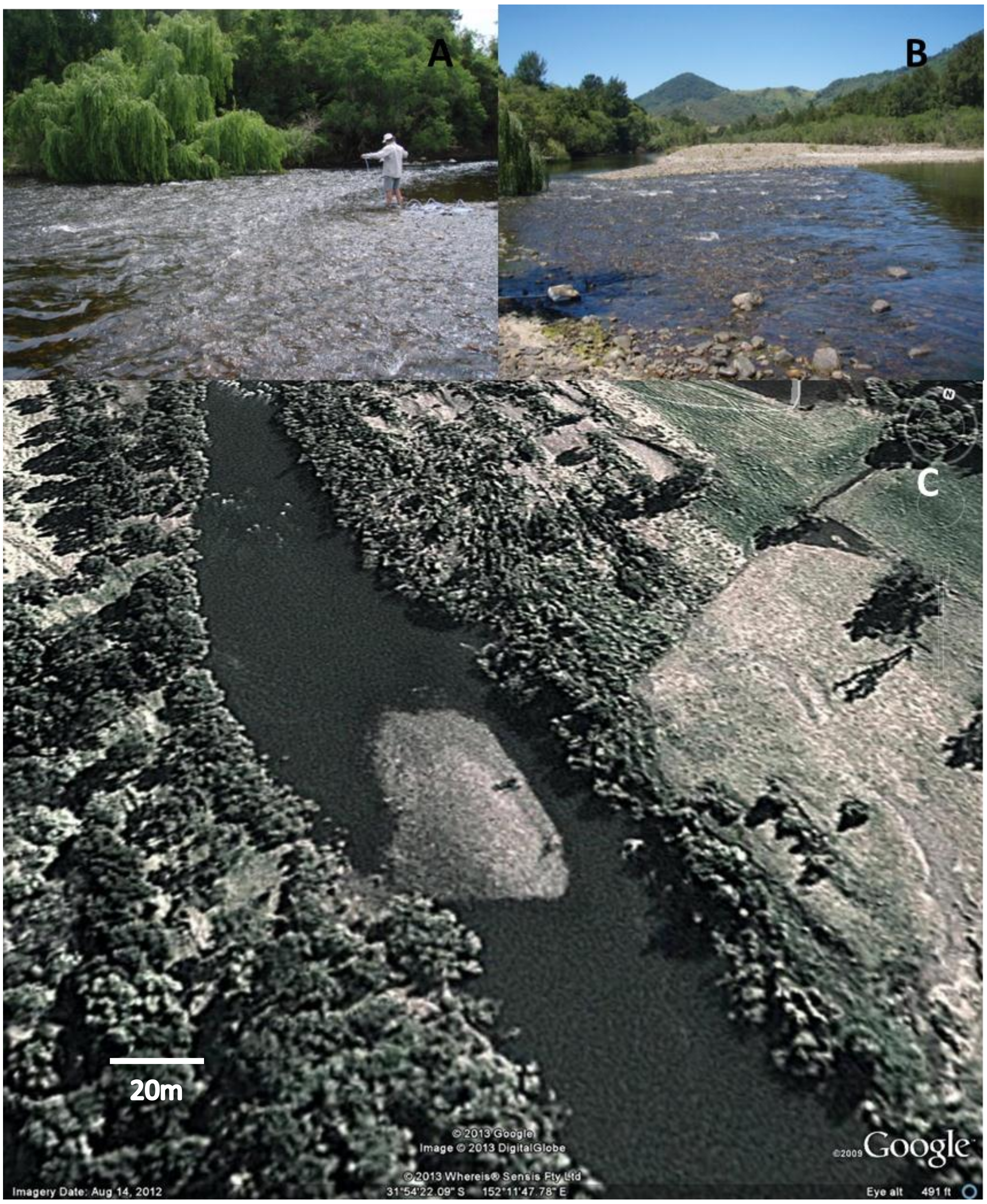


Figure 5.5 A. Woodside riffle in October 2008 (discharge ~ 7 m^3s^{-1} at Killawarra Bridge); B. Feb 2009 (discharge ~4 m^3s^{-1} at Killawarra Bridge)*; C. Aerial view
*Photograph taken looking downstream

Bungay riffle, is downstream of water extraction points but upstream of MCW extraction point. The riffle which is situated in close proximity to a dairy farm and, as a result of this proximity, has a higher level of grazing and nutrient impact (Figs. 5.2 and 5.6, Table 5.4). This site is approximately 10 m above sea level (Table 5.3).

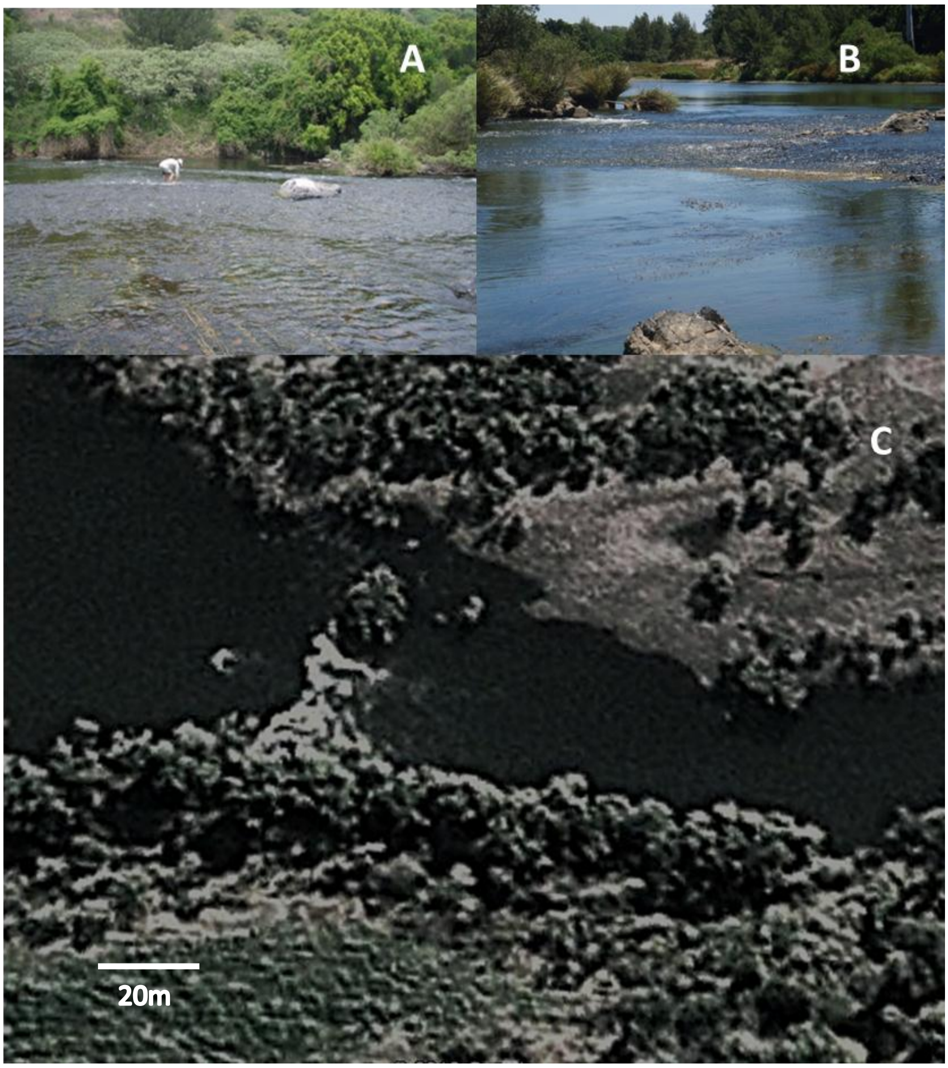


Figure 5.6 A. Bungay riffle in October 2008 (discharge ~ 7 m^3s^{-1} at Killawarra Bridge); B. Feb 2009 (discharge ~4 m^3s^{-1} at Killawarra Bridge)*; C. Aerial view
*Photograph taken looking downstream

MCW riffle, is immediately downstream of the MCW point of extraction for Bootawa Dam (Fig. 5.2). The riffle contains a substantially greater proportion of boulder-sized substrate than upstream sites and shares the grazing and weed impacts found at the other Manning sites. The MCW riffle is approximately 10m above sea level (Table 5.3).

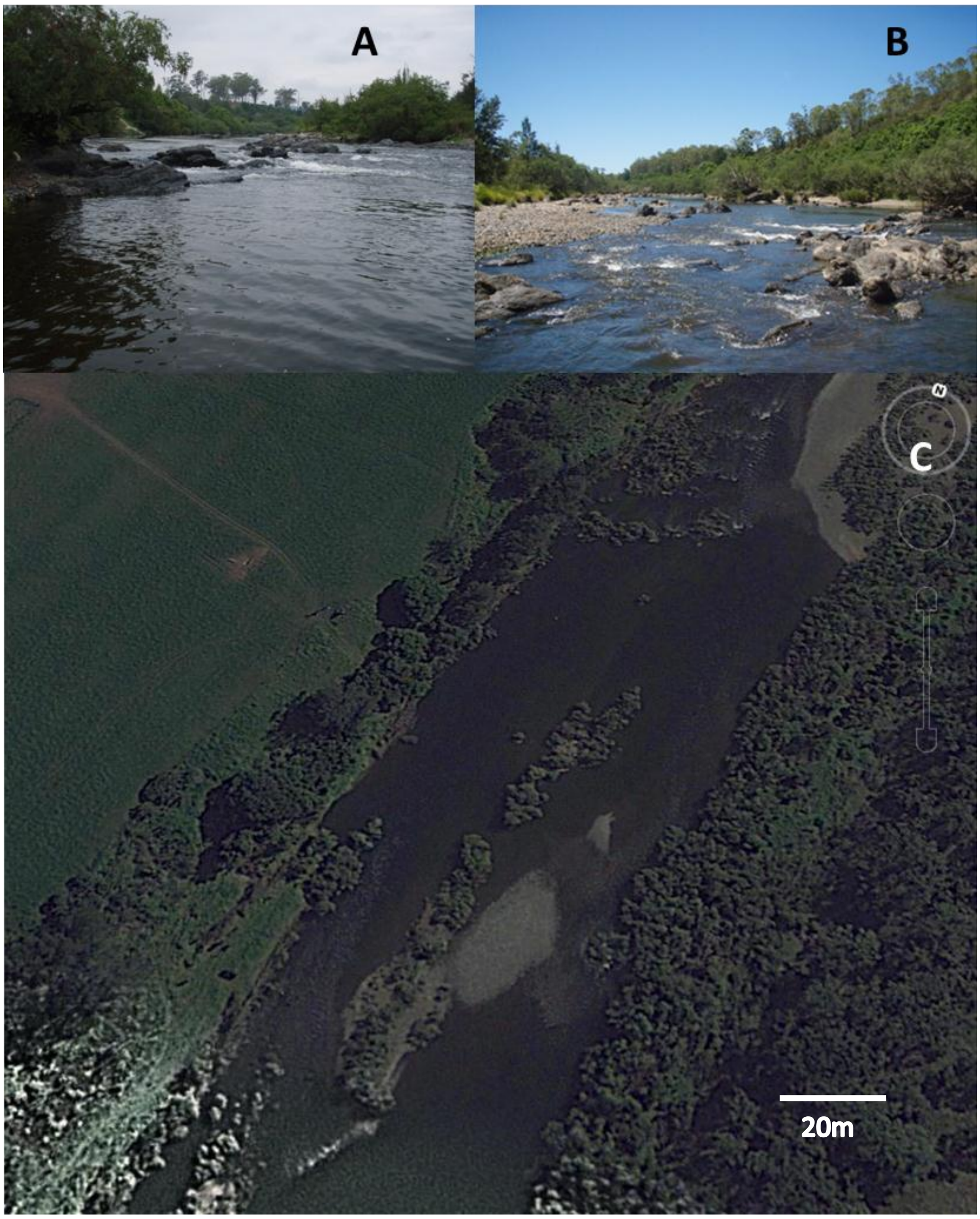


Figure 5.7 A. MCW riffle in October 2008 (discharge ~ 7 m^3s^{-1} at Killawarra Bridge); B. Feb 2009 (discharge ~ 4 m^3s^{-1} at Killawarra Bridge)*; C. Aerial view
*Photograph taken looking downstream

Abbotts riffle is located downstream of the MCW extraction point (Fig. 5.2). This site is immediately upstream of the king tide level described by Bishop (2006). Figure 5.8 illustrates the nature of the site. The physical attributes of the site 6 riffle are outlined in Table 5.3.

Average nutrient concentrations for two water quality sites, Killawarra Bridge and the MCW extraction site located in the lower Manning River, indicate nutrients frequently exceedance 80th percentile values derived from regionally-derived targets (Table 5.4).

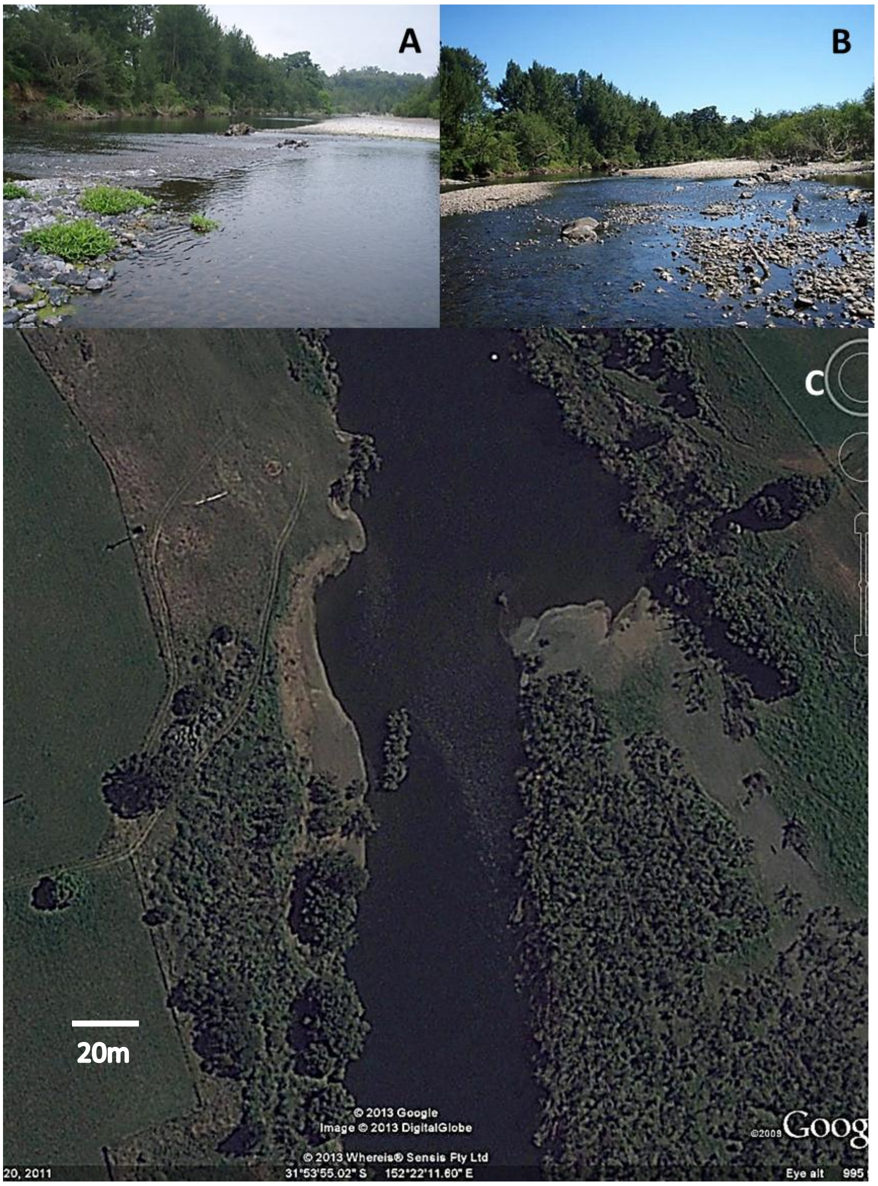


Figure 5.8 A. Abbotts riffle in October 2008 (discharge ~ 7 m^3s^{-1} at Killawarra Bridge); B. Feb 2009 (discharge ~4 m^3s^{-1} at Killawarra Bridge)*; C. Aerial view
*Photographs taken looking downstream

Table 5.3 Physical characteristics of Manning River riffle study reaches (measured at approximately 80th percentile discharge)

Site	Length (m)	Width (m)	Slope (m)	Altitude (m)
Ida Lake Riffle	42	15.8	0.65	40
Railbridge Riffle	65	35.9	0.62	30
Woodside Riffle	44	27.5	0.35	30
Bungay Riffle	59	43.0	0.32	10
MCW Riffle	43	39.2	0.39	10
Abbotts Riffle	48	45.1	0.40	10

Table 5.4 Nutrient concentrations in the Manning catchment, 2001-2012

Site	**Nutrient** (mgL^{-1})	**Average** (mgL^{-1})	**Range** (mgL^{-1})	**80^{th} Percentile Exceedence***
Manning River at Killawarra (Located downstream of Woodside riffle)	Phosphorus	0.04	0.008-0.038	Samples exceed 80th percentile regional guidelines (0.03 mgL^{-1}) 55% of the time)
	Nitrogen	0.26	0.011-0.27	Samples within 80th percentile guidelines (0.30 mgL^{-1})
Manning River at MCW extraction site (Located immediately upstream of MCW riffle)	Phosphorus	0.03	0.007-0.22	Samples exceed 80th percentile guidelines (0.03 mgL^{-1}) 25% of the time
	Nitrogen	0.29	0.01-1.24	Samples exceed 80th percentile guidelines (0.3 mgL^{-1}) 25% of the time
Barrington River (Located downstream of Barrington River riffle sites)	Phosphorus	0.014	0.008-0.022	Samples within 80th percentile guidelines (0.025 mgL^{-1})
	Nitrogen	0.14	0.09-0.23	Samples within 80th percentile guidelines (0.25 mgL^{-1})
Gloucester River (Located downstream of Gloucester River riffle sites)	Phosphorus	0.022	0.011-0.042	Samples exceed 80th percentile guidelines (0.025 mgL^{-1}) 25% of the time
	Nitrogen	0.27	0.17-0.44	Samples exceed 80th percentile guidelines (0.25 mgL^{-1}) 30% of sampling occasions
Dingo Creek (Located upstream of Dingo Creek riffle sites)	Phosphorus	0.055	0.012-0.24	Samples exceed 80th percentile guidelines (0.03 mgL^{-1}) 35% of sampling occasions
	Nitrogen	0.35	0.11-1.14	Samples exceed 80th percentile guidelines (0.30 mgL^{-1}) 25% of sampling occasions

*Percentile exceedance compared to upland and lowland regionally derived water quality thresholds – see chapter 3

5.1.2.2 Tributary riffle sites

Three tributaries were chosen to investigate the influence of elevated nutrients on macroinvertebrate response to low flows. The Barrington River was chosen as its upper reaches are generally unaffected by major impacts such as grazing and have low nutrient concentrations, with both phosphorus and nitrogen levels within the 80^{th} percentile of regionally-derived threshold values (Chapter 3) (Table 5.4). Gloucester River, upstream of Gloucester Township, has some grazing and cropping impacts which are reflected in the higher nutrient levels when compared to the upper Barrington River (Table 5.4), with levels for nitrogen and phosphorus exceeding the 80^{th} percentile guidelines (Table 5.4). Dingo Creek was also found to have elevated nutrient concentrations, possibly from abattoir effluent being sprayed on nearby paddocks. Nitrogen and phosphorus levels exceeded threshold guidelines, with maximum nitrogen levels reaching 1.14 mgL^{-1} and maximum phosphorus levels reaching 0.24 mgL^{-1}. The relevant phosphorus and nitrogen ranges and exceedances for this site are detailed in Table 5.4.

Barrington River sites 1 to 3, were upstream of major grazing impacts, with intact, mostly native, riparian vegetation (Figs. 5.9 and 5.10). All sites were shallow with cobbled substrate. Physical characteristics relating to each riffle site are detailed in Table 5.5.

Figure 5.9 Barrington and Gloucester Rivers - macroinvertebrate site locations

Table 5.5 Physical characteristics of Tributary study reaches (measured at approximately 80th percentile discharge)

Site	Length (m)	Width (m)	Slope (m)	Altitude (m ASL)
Barrington site 1	38	21	0.6	150
Barrington site 2	23	13	0.55	130
Barrington site 3	35	18	0.5	130
Gloucester site 1	30	15	0.5	160
Gloucester site 2	21	12	0.5	130
Gloucester site 3	38	17	0.45	120
Dingo site 1	22	14	0.38	20
Dingo site 2	31	18	0.35	20
Dingo site 3	24	12	0.35	20

Figure 5.10 Barrington River macroinvertebrate site 2 - A. Riffle, late afternoon, looking downstream; B. Barrington River site 2, aerial view

Gloucester River sites 1 to 3, were all upstream of the township, with mostly intact riparian vegetation, though weed and grazing impacts exist (Figs. 5.9 and 5.11). Sites were between 145m and 200m above sea level (Table 5.5).

Figure 5.11 Gloucester River macroinvertebrate site 1 - A. Riffle, late afternoon, looking downstream; B. Gloucester River site 2, aerial view

Dingo Creek sites 1 to 3 were all in the vicinity of the abattoir effluent spray area, the last site only a short distance from the creeks confluence with the Manning River near Bungay Pool (Fig. 5.12). Bank vegetation was mostly intact but the surrounding riparian zone was impacted by weeds, grazing, cropping and abattoir effluent sprayed on nearby paddocks (Fig. 5.13). Sites were around 30m above sea level (Table 5.5).

Figure 5.12 Dingo Creek macroinvertebrate site locations

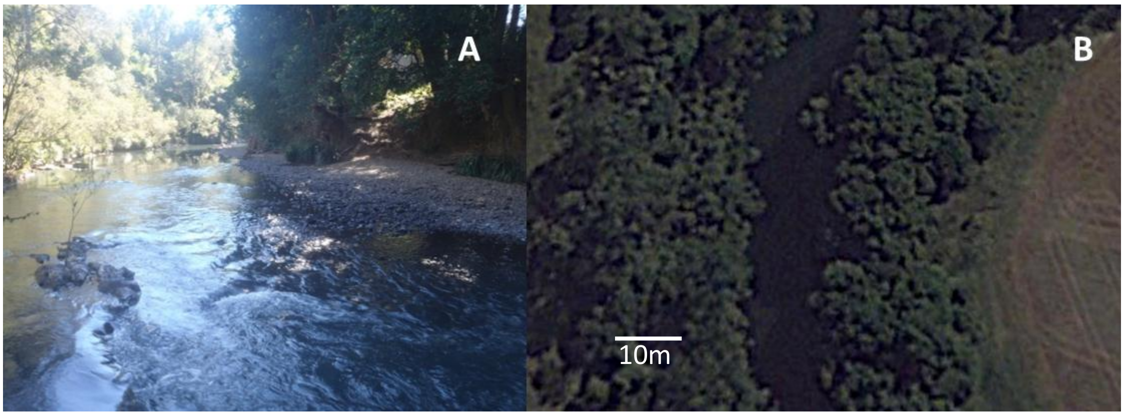


Figure 5.13 Dingo Creek macroinvertebrate site 1 - A. Riffle, early morning, looking downstream; B. Dingo Creek site 1, aerial view

5.2 Methods

5.2.1 Field sampling

Sites were selected according to riffle suitability, which included cobbled substrate, access and location within the lower Manning. There were four sites upstream of the MCW extraction point and two sites downstream of the extraction point (Fig. 5.2). A preliminary site assessment was completed at each of the riffles in October 2008 when discharge in the Manning River was around 7 $m^3 s^{-1}$ at Killawarra gauge (approximately 80th percentile flow). The preliminary assessment was undertaken to ensure the appropriateness of the sites for macroinvertebrate collection and to measure riffle velocity and depth at moderate flows. Channel profiles and velocity transects were undertaken to assess the type of habitats available under these moderate flow conditions.

Extraction within the Manning River is greatest around the township of Wingham with water predominantly extracted for pastures and other agricultural pursuits. While the total volume for irrigation extraction from the lower Manning is up to 8,058,000 $m^3 year^{-1}$, MCW extracts up to 7,194,000 m^3 $year^{-1}$ from the river downstream of the Wingham area, for storage in Bootawa Dam. Riffles upstream of high extraction areas provide important baseline information on hydraulic-macroinvertebrate relationships that are less affected by water extraction. It is also important that sites are located within the area of greatest agricultural extraction but not impacted by MCW extraction for water supply. The two most downstream sites were located below the MCW extraction point to determine the cumulative impacts of both irrigation and MCW extraction.

Ida Lake riffle, Railbridge riffle, Woodside riffle and tributary sites are upstream of major extraction points , Bungay riffle is subject to agricultural extraction only and MCW riffle and Abbotts riffle are downstream of both agricultural and water supply extraction. These sites were chosen to identify associations between environmental and ecological variables and possible impacts from extraction. The influence of extraction during low flows and possible impacts on riffle communities by reducing habitat availability and variability is also considered.

Median substrate size between the various riffles to allow comparisons was assessed by measuring the median axis of at least 49 bed particles (Newbury 2003). Surface flow characteristics, depth and velocity were used to determine flow classes (laminar, broken and chaotic). Laminar flow types were classified by the observation of an unbroken flow, with maximum velocity of up to 0.8 m s^{-1} measured during sampling. Broken flows were determined by more fragmented water activity, with maximum velocities reaching over 1 m s^{-1} during sampling. Chaotic flows were turbulent flows which reached maximum velocities of up to 1.2 m s^{-1} during sampling.

The impact of extraction during low flows on riffle communities through the reduction of habitat variability was assessed by comparing physical changes within Manning River riffles. Changes in velocities and depths within riffles under different flow conditions were determined using measurements taken every 1 m along 10 transects under moderate and low flow conditions.

Macroinvertebrate sampling was conducted in early February 2009 over a 4-day period. February was chosen as discharge in the Manning River at Killawarra was between 3.5 and 4 $m^3 s^{-1}$ (between 90th to 95th percentile discharge), water temperatures and periphyton production were high (Fig. 5.14). Macroinvertebrate sample selection methods were adapted from Brooks et al. (2005), where areas of different hydraulic conditions within each riffle were initially visually assessed based upon surface flow characteristics, depth and velocity.

Three samples were taken from three identified flow classes, which was primarily determined using a visual assessment of laminar, chaotic and broken flows, within each of the six riffles. The visually assessed conditions were used to ensure macroinvertebrate samples were collected from the three dominant hydraulic types found in each of the studied riffles.

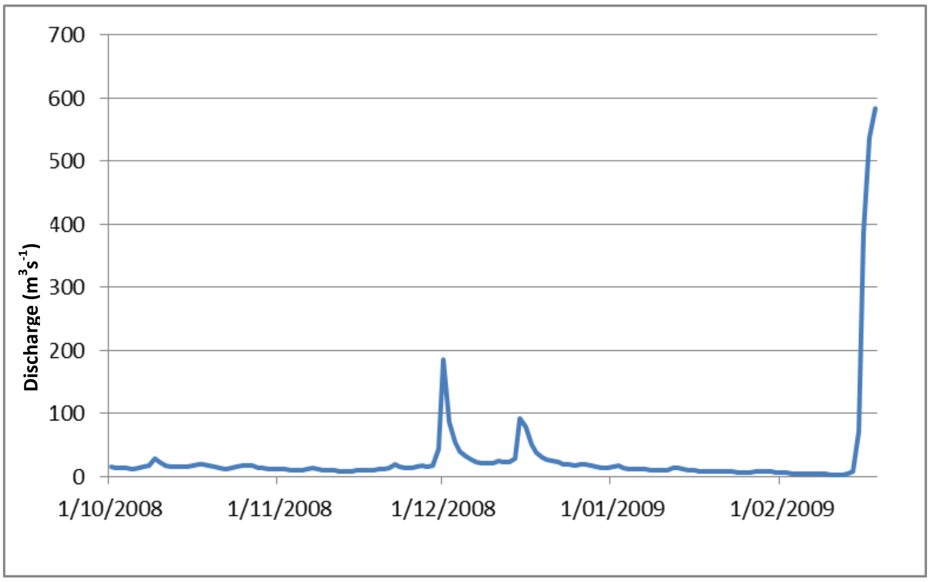


Figure 5.14 Manning River discharge at Killawarra gauge, October 2008 to February 2009

Tributary sites were selected based on long-term nutrient concentrations and riffle characteristics. The Barrington River, as indicated in Table 5.3, has low nutrient levels and a minimally disturbed catchment, with the upper catchment dominated by Barrington Tops National Park. This allows a comparison to be undertaken between the response of macroinvertebrates to low flows under low nutrient levels when compared to sites with higher nutrient levels (Table 5.3). The Gloucester River was selected as an intermediate nutrient site, with concentrations lower than that found in the lower Manning River, but higher than the Barrington River (Table 5.3). Dingo Creek was selected to represent higher nutrient levels in comparison to the other two tributary sites, with similar nutrient levels to the lower Manning (Table 5.3).

Sites within each of the tributaries were selected using the same criteria as that used for Manning riffle sites, cobbled substrates and representation of chaotic, laminar and broken flow types, as explained previously. The tributary sites were sampled in May 2012 following three weeks of low flows (Figure 5.15). The sampling period was selected to capture the influence of antecedent flow conditions. This approach is supported by Sheldon (2005), who suggested that reference data should be adjusted according to the hydrological conditions that precede macroinvertebrate sampling. There was also little difference in the macroinvertebrate taxa collected from riffle sites from the Barrington and Gloucester rivers during spring and autumn as part of rapid biological assessment conducted in 1995/96 as part of the AusRivAS program (Hose and Turak 2004).

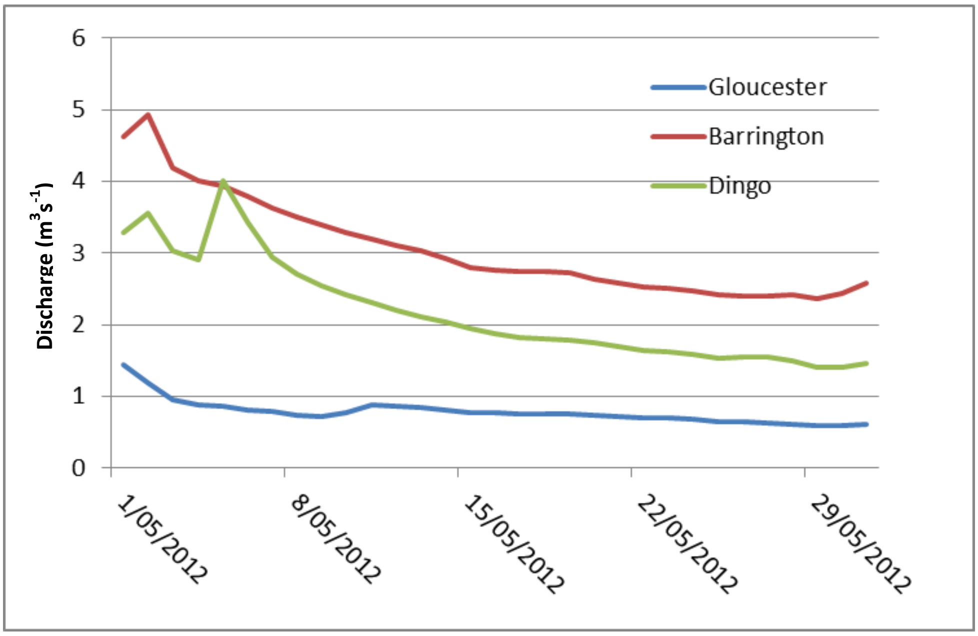


Figure 5.15 Barrington River discharge at Forbesdale gauge, Gloucester River at Gloucester gauge and Dingo Creek discharge at Belbourie gauge, May 2012

Macroinvertebrates were collected by randomly selecting three cobbled-sized rocks (total surface area ranging from 0.03 m^2 to 0.13 m^2) as independent replicates within each of the three different hydraulic zones. Chiasson (2009) found that a marked decrease in confidence limits occurred when macroinvertebrate samples increased fromtwo to three. Samples were collected from all sites on the same or consecutive days, under similar flow conditions. Identification of macroinvertebrates was to family or, where possible, genus level using available keys.

Substrate characteristics, velocity and periphyton were assessed for each riffle. Velocity was also measured using a Marsh McBirney flow-mate with an electromagnetic flow sensor at each cobble replicate prior to its collection and periphyton was also scraped from cobble replicates for chlorophyll-a analysis (Table 5.6). Water column samples were collected from fixed locations at each riffle site. Samples were collected for analysis of total nitrogen (TN mgL^{-1}), total phosphorus (TP mgL^{-1}), nitrite/nitrate (NO_3-N mgL^{-1}), ammonium (NH_4^+ mgL^{-1}), soluble reactive phosphorus (SRP mgL^{-1}), and turbidity (NTU) (see section 5.2.2 for details of laboratory analyses).

On each sampling occasion, pH, temperature, conductivity (µS cm^{-1}) and DO (mg L^{-1}) were measured using a Hydrolab DS 5X multiprobe which supported a Hach luminescent dissolved oxygen (LDO) sensor, temperature sensor, pH sensor and electrical conductivity sensor (Table 5.7). A Hach field turbidity meter was used in situ.

Table 5.6 Field sampling measurements and methods

Variable	Measurement	Method/Treatment/References
Substrate	Visual assessment of particle size % at each site; Surface area of sampled cobbles was measured following macroinvertebrate and periphyton collection	Surface area of cobbles was calculated by wrapping stones in foil, and using a linear relationship between foil mass and surface area (Calow 1972, Shelly 1979).
Velocity	Riffle transects – Velocity readings taken at a depth of 4cm at 1-m intervals across the width of the riffle, 20 transects were used per riffle; Velocity was measured 4 cm above sampled cobbles prior to removal for macroinvertebrate and periphyton collection	Velocity was measured using a Marsh-McBirney portable flow meter (Model 2000); Methods adjusted from Brooks et al. 2005.
Periphyton	Visual assessment , using % cover, across sampled riffle; Periphyton was scraped from sampled cobble following macroinvertebrate removal using a soft brush into 200 mL of distilled water and then homogenising.	100-mL subsamples were filtered through a glass fibre filter (e.g., Whatman® GFC), then folded and wrapped with aluminium foil to exclude light. The sample was stored in a cool, dark container until frozen. Section 5.2.2 provides details of laboratory analysis.
Macro-invertebrates	Cobbles were immediately placed in large sorting trays when removed from the stream. Macroinvertebrates were picked from each sampled cobble for 30 minutes.	Preserved in alcohol and identified to family or genus where possible (Growns et al. 1997).
Nutrients	Three grab samples taken mid water column at each riffle; Hydrolab was used to collect in-situ water quality information (see Table 5.8)	Water samples immediately chilled and frozen within 8 hours. Table 5.8 provides details of laboratory analyses (MCW 2010).

Table 5.7 Details of Hydrolab multiprobe and field turbidity meter

Sensor	Range	Accuracy	Resolution
Hach LDO	0-60 mg L^{-1}	± 0.1 mg L^{-1} at <8 mg L^{-1} ± 0.2 mg L^{-1} at >8 mg L^{-1}	0.01 mg L^{-1}
Temperature (30k ohm thermistor)	-5 to 50 °C	± 0.10°C	0.01°C
Conductivity (graphite electrodes)	0-100 mS cm^{-1}	± 0.5% reading + 0.001 mS cm^{-1}	0.001 mS cm^{-1}
pH (reference electrode)	0 to 14 pH units	± 0.2 units	0.01 units
Turbidity (Hach portable turbidity meter)	0 to 1000 NTU	≤ 0.02 NTU	0.01 NTU

5.2.2 Laboratory analysis

Nutrient field sampling and analyses followed the protocols described in Chapter 3, with all nutrient analyses undertaken by the MCW Laboratory. Details of the nutrients sampled are listed in Table 5.8.

Table 5.8 Water quality variables (source MCW 2010)

<table>
<tr><th>Variable</th><th>Units</th><th>Analysis Range</th><th>Method</th></tr>
<tr><td>Total Phosphorus</td><td rowspan="2">mgL^{-1}</td><td rowspan="2">0.002 - 1.00 mgL^{-1} as P</td><td rowspan="2">The analysis involves the oxidation of the phosphorus compounds to orthophosphate (PO_4^{3-}) by digesting with potassium persulphate.
Flow Injection Analyser used to process samples.</td></tr>
<tr><td>Soluble Reactive Phosphorus</td></tr>
<tr><td>Total Nitrogen</td><td></td><td>0.05 - 1. 00 mgL^{-1} as N</td><td rowspan="3">Ammonia nitrogen, nitrite nitrogen and organic nitrogen are converted to nitrate by alkaline persulphate digestion. Total nitrogen is then determined by measuring nitrate nitrogen contributed by the above nitrogen species plus any nitrate nitrogen originally in the sample, at 520nm.
Flow Injection Analyser used to process samples.</td></tr>
<tr><td>Nitrate/Nitrite</td><td>mgL^{-1}</td><td>Nitrate concentration 0.01- 1. 00 mgL^{-1} as N, and Nitrite concentration of 0.001 – 0.100 mgL^{-1} as N.</td></tr>
<tr><td>Ammonia</td><td>mgL^{-1}</td><td>0.01 to 1.00 mgL^{-1} NH_3-N/L.</td></tr>
<tr><td>Turbidity</td><td>NTU</td><td>0.02 to 1000 NTU</td><td>Method compares the intensity of light scattered by the sample under defined conditions with the intensity of light scattered by a standard reference suspension under the same conditions. The higher the intensity of scattered light, the higher the turbidity.</td></tr>
</table>

Analysis of chlorophyll-a was undertaken following Standard Methods (APHA 1992). The analysis involved the extraction of chlorophyll-a in acetone, the measurement of the extracted chlorophyll concentration using a spectrophotometer and the calculation of chlorophyll density (mg m^2) on substrates by determining the proportion of original sample that was assessed for chlorophyll.

5.2.3 Data analysis

Pairwise tests of Analysis of Similarity (ANOSIM) were carried out using the software package PRIMER version 6 (Clarke and Gorley 2006) to detect the differences between groupings of Manning River sites and reference sites using macroinvertebrate composition. ANOSIM with a one-way design was used to test differences in macroinvertebrate communities within the Manning River, upstream and downstream of the extraction site, and Manning River sites compared to reference sites.

To investigate how different flow types may influence macroinvertebrate diversity and abundance for firstly the Manning sites and then for reference sites, a hierarchical sampling design was used to test for differences in total abundance and taxon richness with a mixed-model PERMANOVA. PERMANOVA allows the partitioning variability according to one or more explanatory variables and the testing of interactions among factors (Clarke and Gorley 2006).

Data for the PERMANOVA were checked for normality and homogeneity of variances by inspecting residual plots (Quinn and Keough 2002) and log (x+1)-transformed. Factors in the model were site (random) and flow type – laminar, broken or chaotic - (fixed). The error term for each comparison was determined using methods described in Winer (1991) and Underwood (1997). The number of replicates used in each case was three for main effects. All statistical analyses were carried out using Primer 6 (Anderson et al. 2008).

Differences in benthic riffle communities were also used to determine whether higher nutrient levels were a factor in low flow impacts within riffles. Analyses were undertaken to detect differences in macroinvertebrate communities between sites exposed to extraction and/or higher nutrients compared to those found to have lower nutrient levels and/or no extraction pressures. To do this, macroinvertebrate community composition was compared using multidimensional scaling (MDS) and permutational analysis of variance (PERMANOVA; Anderson et al. 2008, McArdle and Anderson 2001). Sites, water quality, velocity, cobble surface area and chlorophyll-a variables were used as vectors. Data were 4^{th}-root transformed to limit weighting by rare or abundant taxa. Analyses were carried out using a Bray-Curtis distance measure.

A notable difference between site macroinvertebrate composition was the numerical domination of simuliids within Manning River riffles. To compare this numerical domination between sites, the percentage of simuliids was calculated for each site and compared to total number of macroinvertebrates. These results were then tabulated for comparison between sites.

Linear regressions were used to determine relationships between total nutrients (TP, TN), taxa richness and percentage of simuliids, commonly known as blackfly larvae, at each site. These analyses allowed comparisons of taxa richness and simuliid dominance between sites, allowing an assessment of community response to varying nutrient concentrations. Data for the linear regressions were checked for normality and homogeneity of variances by inspecting residual plots (Quinn and Keough 2002) and log (x+1)-transformed.

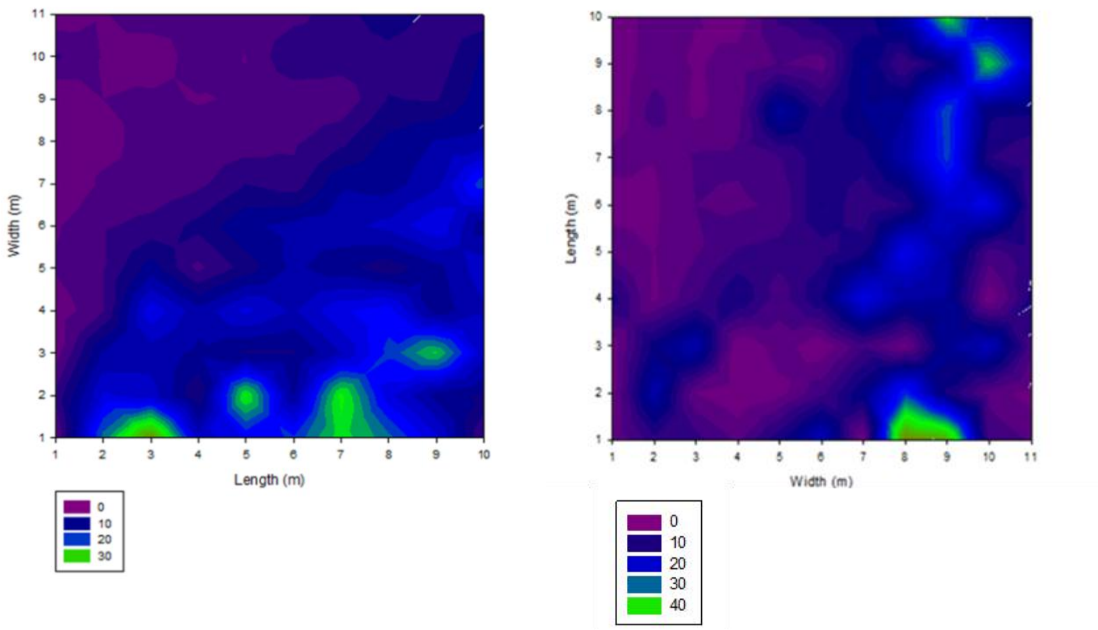
To assess how physical changes, particularly differences in periphyton levels between riffles, influence the composition of macroinvertebrate community, taxa were divided into functional feeding groups. The allocation of each taxon to each functional feeding groups was determined from the literature (Merritt and Cummins 2002, Uwadiae 2010). Community structure in relation to functional feeding groups was determined using the percentage of each function feeding group found at each reach. The composition of functional groups for each site was then tabulated for between site comparisons.

The impact of extraction during low flows on riffle communities through the reduction of habitat variability was assessed by combining the depth and velocity measured at each point and mapping these results to highlight differences in the types of habitats available within riffles. From these assessments, graphical representations were developed to identify different habitat types and changes to riffle habitats under moderate and low flow types. By doing this, an indication of the extent of habitat loss under low flow conditions was determined.

5.3 Results

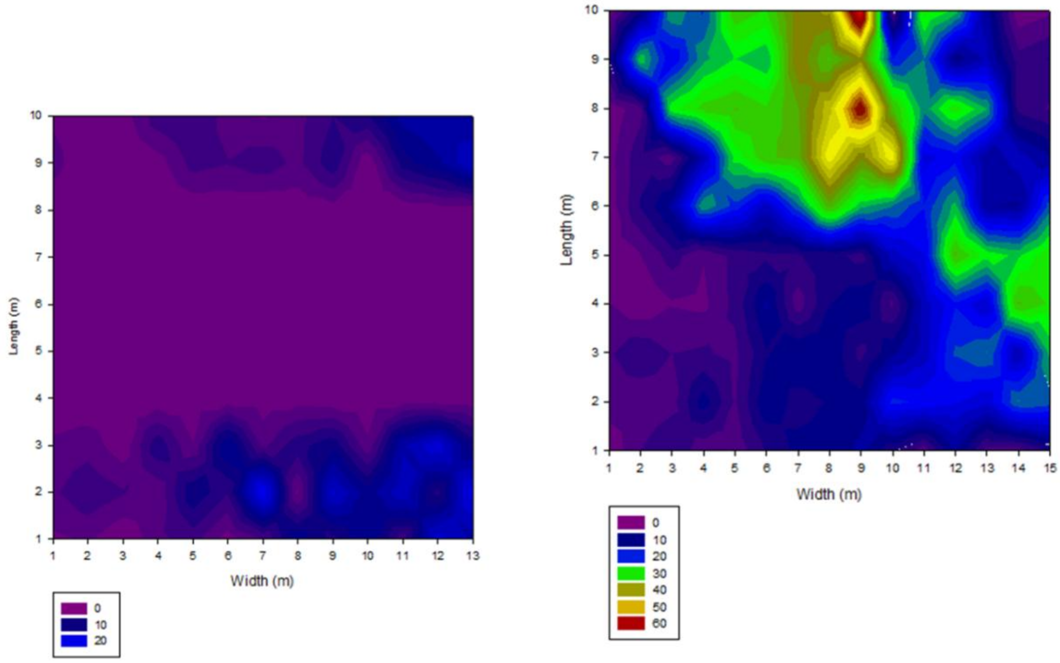
5.3.1 Ecohydraulic mapping

Preliminary hydraulic assessment and mapping of selected riffles was carried out prior to macroinvertebrate sampling in October 2008, with measurements taken during flows of around 7 m^3s^{-1} at Killawarra gauging station. The second assessment was carried out during the macroinvertebrate sampling in February 2009, when discharge was around 3.5-4 m^3s^{-1} at Killawarra. The contour graphs combine depth and velocity measured in the riffles to indicate the variety of micro-habitats available under different flows (Figs. 5.16 to 5.21). The contour graphs show that the higher velocity and deeper habitats, indicated by the brighter colours, reduce significantly under low flow conditions. The variety of habitats available at each riffle declined under low flow conditions. The Railbridge riffle in particular demonstrates the reduction in velocity and depth under a reduced discharge rates (Fig. 5.17).



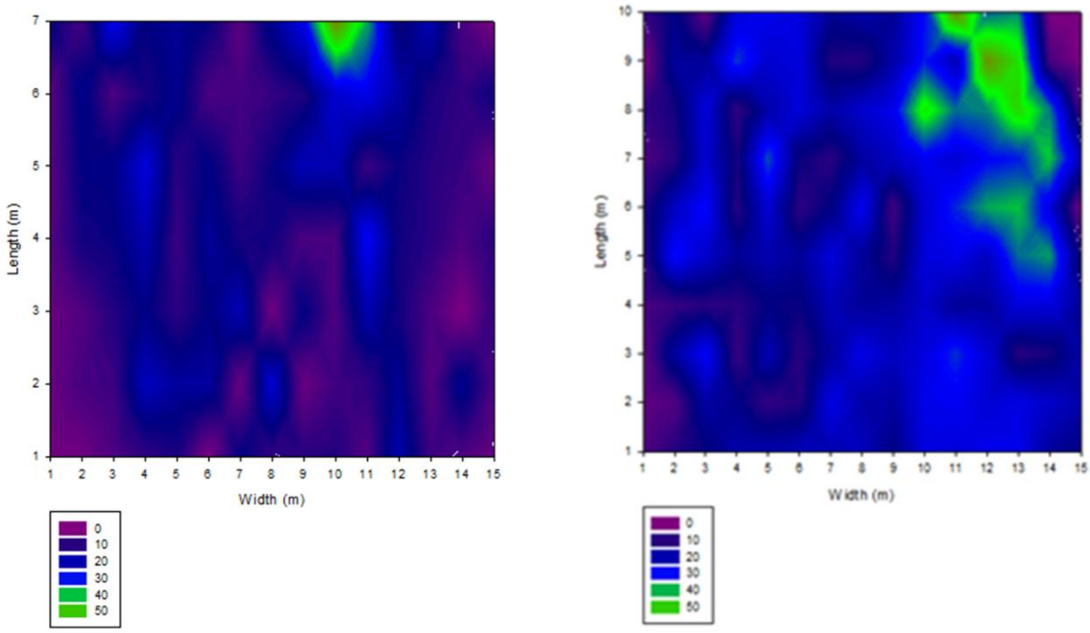
a) Ida Lake @ 3.5 m^3s^{-1} b) Ida Lake @ 7 m^3s^{-1}

Figure 5.16 Velocity and depth at Ida Lake riffle under a) 3.5 m^3s^{-1} at Killawarra and b) 7 m^3s^{-1} at Killawarra (units = velocity m s^{-1} x water depth m)



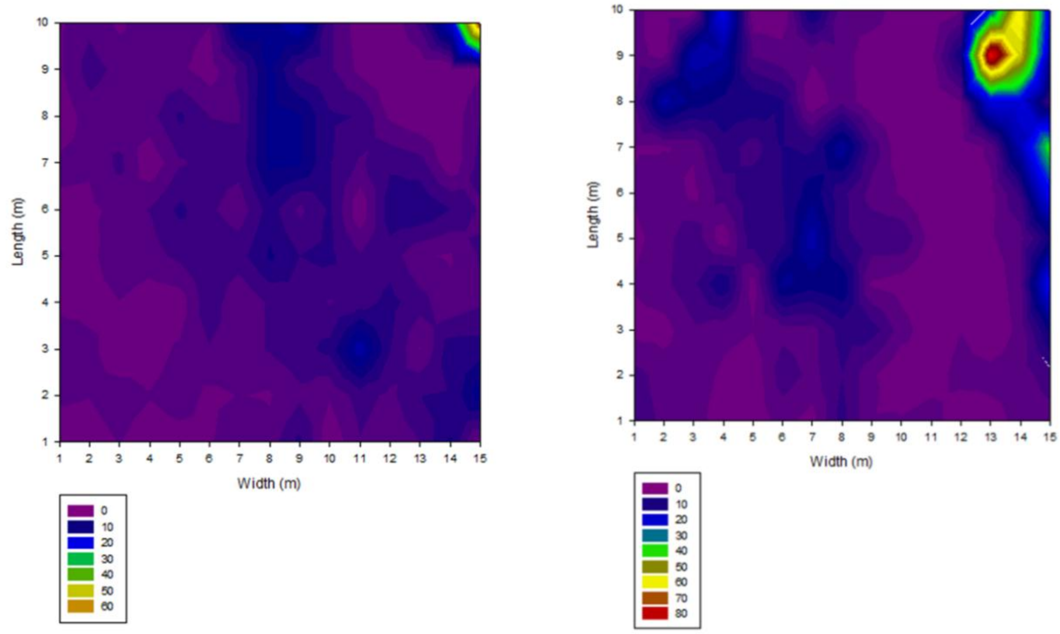
a) Railbridge @ 3.5 m^3s^{-1} b) Railbridge @ 7 m^3s^{-1}

Figure 5.17 Velocity and depth at Railbridge riffle under a) 3.5 m^3s^{-1}at Killawarra and b) 7 m^3s^{-1}at Killawarra (units = velocity m s^{-1} x water depth m)



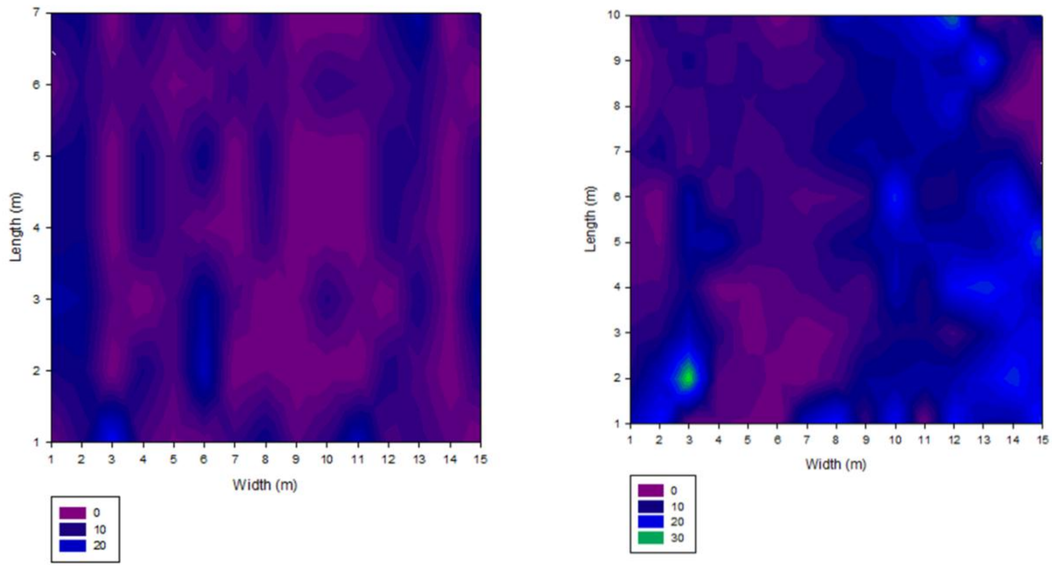
a) Woodside @ 4 m^3s^{-1} b) Woodside @ 7 m^3s^{-1}

Figure 5.18 Velocity and depth at Woodside riffle under a) 4 m^3s^{-1}at Killawarra and b) 7 m^3s^{-1} at Killawarra (units = velocity m s^{-1} x water depth m)



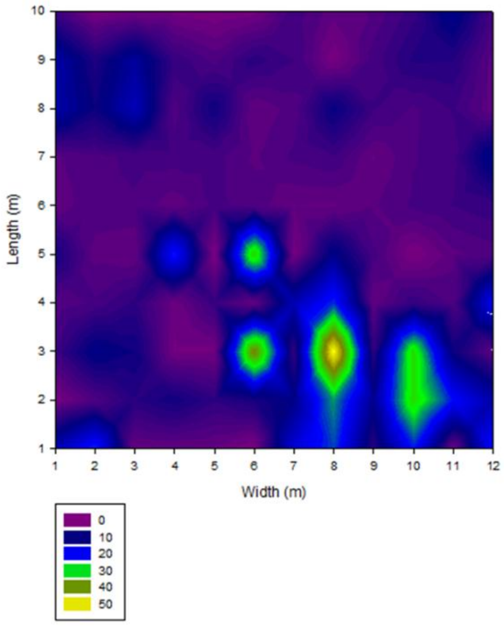
a) Bungay @ 4 m^3s^{-1} b) Bungay @ 7 m^3s^{-1}

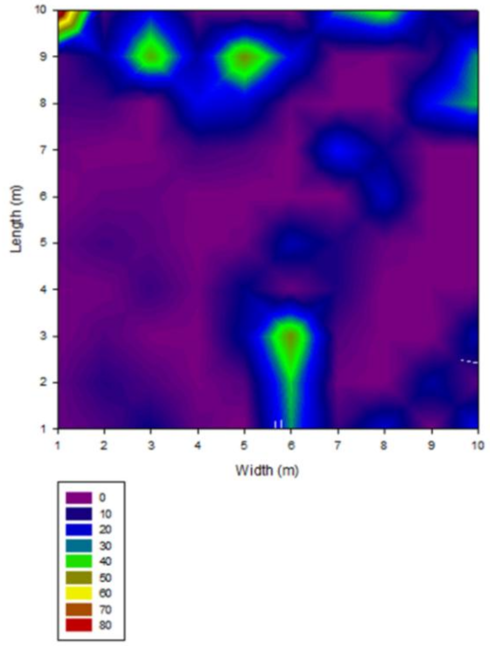
Figure 5.19 Velocity and depth at Bungay riffle under a) 4 m^3s^{-1} at Killawarra and b) 7 m^3s^{-1} at Killawarra (units = velocity m s^{-1} x water depth m)



a) MCW @ 4 m^3s^{-1} b) MCW @ 7 m^3s^{-1}

Figure 5.20 Velocity and depth at Midcoast Water riffle under a) 4 m^3s^{-1} at Killawarra and b) 7 m^3s^{-1} at Killawarra (units = velocity m s^{-1} x water depth m)





a) Abbotts @ 4 m^3s^{-1} b) Abbotts @ 7 m^3s^{-1}

Figure 5.21 Velocity and depth at Abbotts riffle under a) 4 m^3s^{-1} at Killawarra and b) 7 m^3s^{-1} at Killawarra (units = velocity m s^{-1} x water depth m)

5.3.2 Relationships between nutrients, chlorophyll-a and macroinvertebrates in riffles

Total phosphorus and nitrogen concentrations measured at the time of sampling show higher concentrations in the Manning riffles when compared to the reference riffles (Figs. 5.22 and 5.23). The negative correlation between taxa richness and nutrient concentrations indicates that increased nutrients may reduce taxa richness within affected riffles (Figs. 5.22 and 5.23). A positive relationship exists between nutrient concentrations and simuliid numbers (TP: r^2 = 0.89, TN: r^2 = 0.71) (Figs. 5.24 and 5.25). This positive relation is likely to be driven by the high number of simuliids found at all lower Manning River sites.

Differences between periphyton chlorophyll-a in the Manning River and reference sites indicates that chlorophyll-a was not a significant influence on macroinvertebrate numbers or taxa (Fig. 5.26). Figure 5.27 demonstrates the observed differences in algal structure at Manning River sites in comparison to reference sites. Filamentous algae were prevalent across all sampled lower Manning River riffles during the summer low-flow period. Reference sites were observed to support diatom-dominated periphyton with little filamentous algae present despite experiencing similar physical conditions as the lower Manning sites.

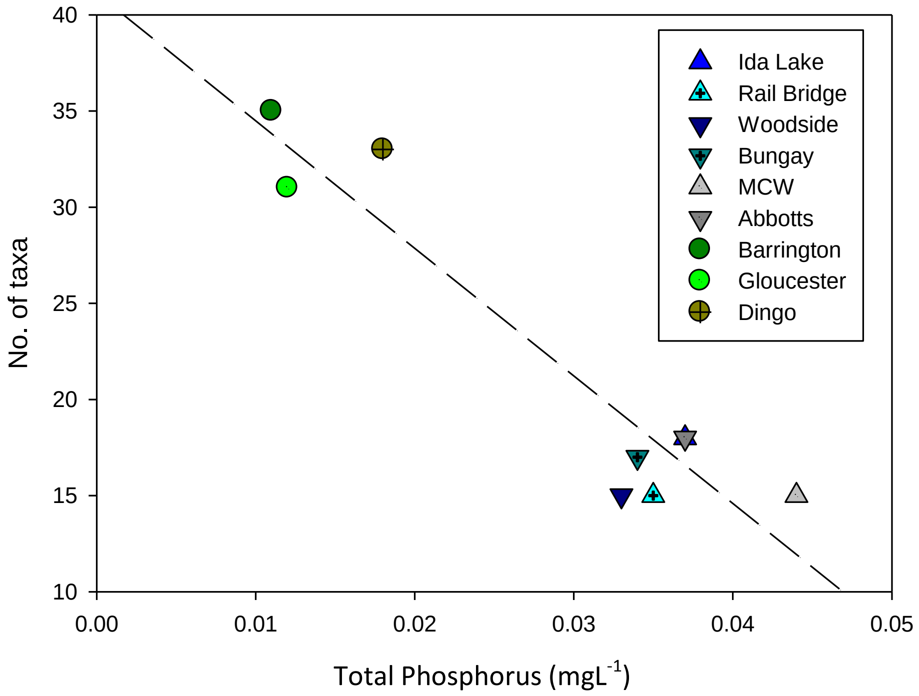


Figure 5.22 Macroinvertebrate taxa richness for Manning River and reference sites vs total phosphorus (r^2=0.89) (Data for each site are represented as separate data points)

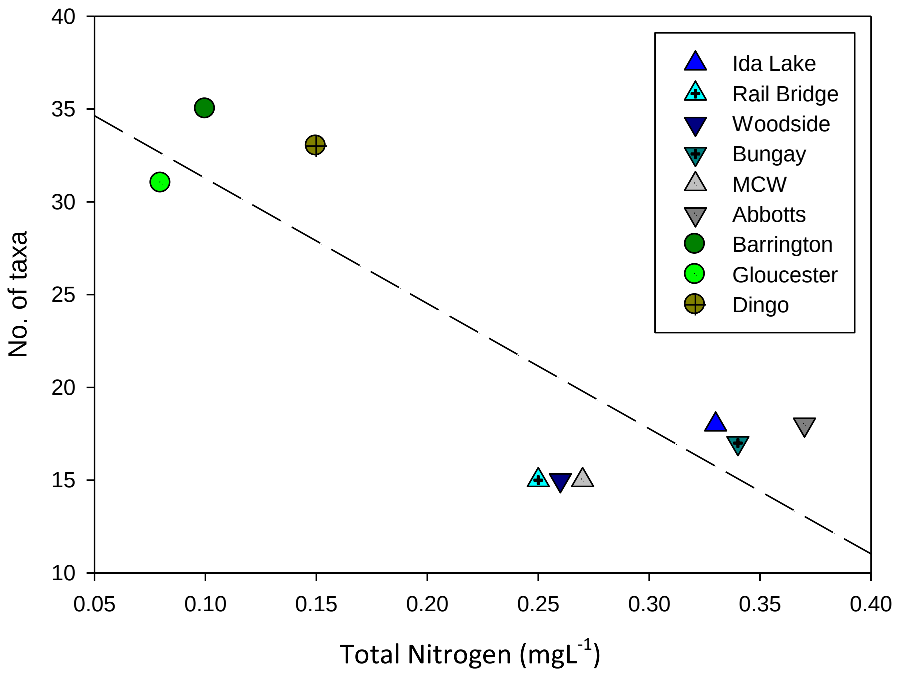


Figure 5.23 Macroinvertebrate taxa richness for Manning River and reference sites vs total nitrogen (r^2=0.71). (Data for each site are represented as separate data points)

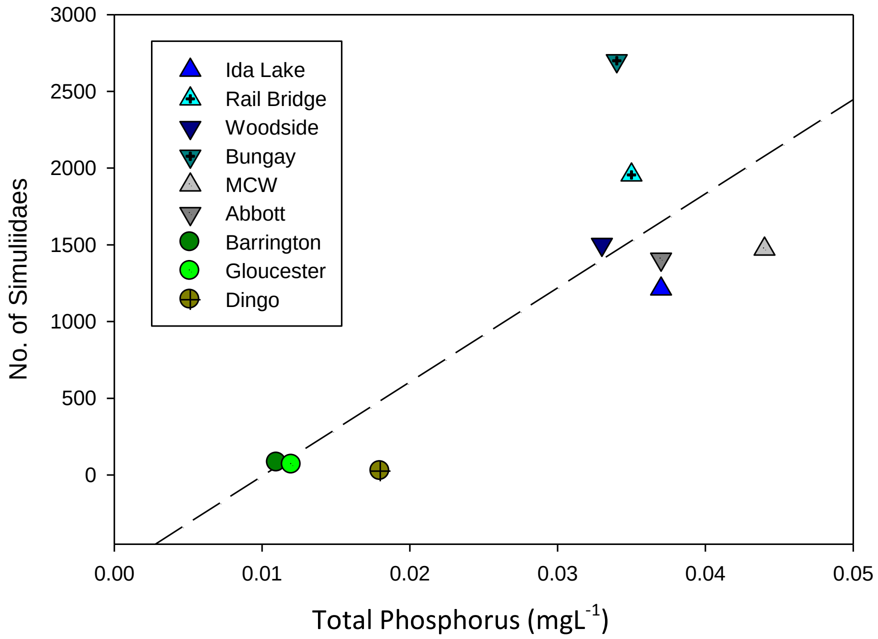


Figure 5.24 Simuliidae abundance for Manning River and reference sites vs total phosphorus (r^2=0.63). (Data for each site are represented as separate data points).

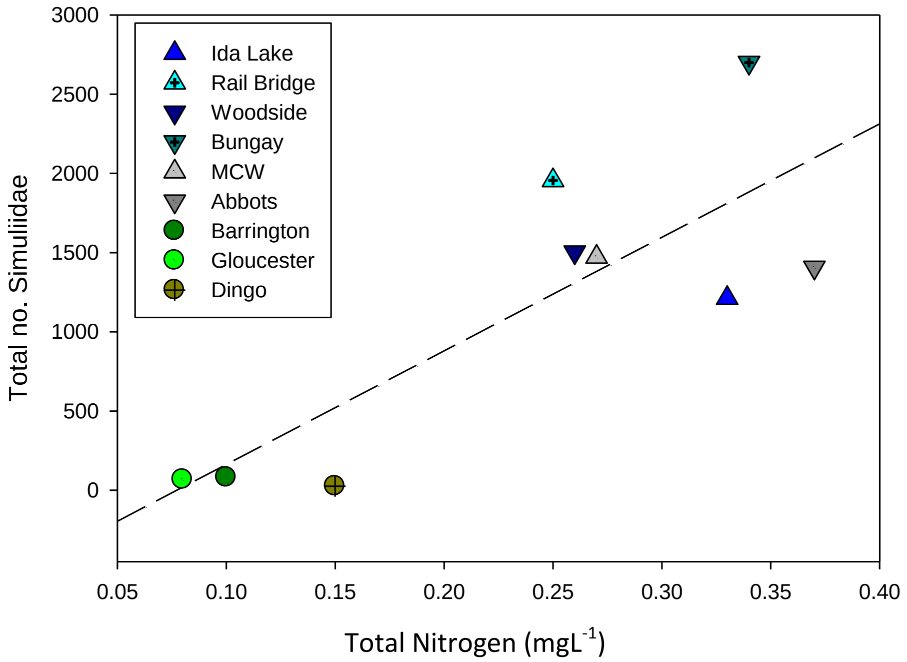


Figure 5.25 Simuliidae abundance for Manning River and reference sites vs total nitrogen (r^2=0.66). (Data for each site are represented as separate data points)

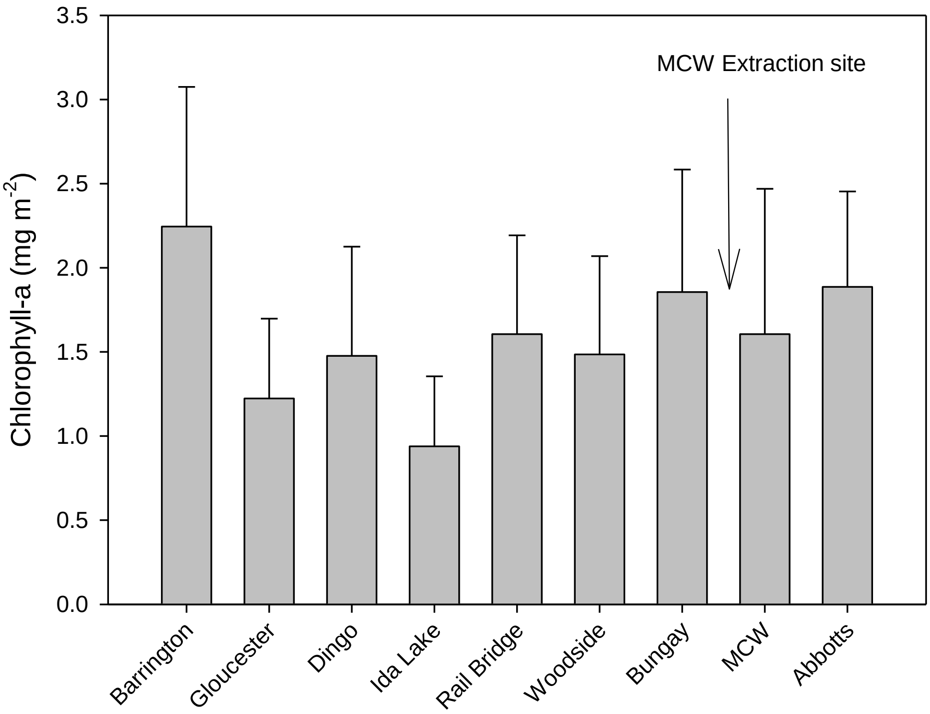


Figure 5.26 Average and SD of periphyton chlorophyll-a at Manning River and reference sites

Figure 5.27 (A) Filamentous algae frequently observed at Manning River riffles – Bungay Pool; (B) Filamentous-free cobbles observed at reference sites - Barrington River

Figure 5.28 shows macroinvertebrate abundance and proportion of simuliids collected from the Manning and reference sites. Simuliids were numerically dominant at Manning sites, with very few simuliid individuals collected at the reference sites (Fig. 5.28). Total abundance varied across all sites in the Manning and reference sites, ranging from less than 1500 individuals per site at Abbotts to over 3000 at Bungay with 88% of individuals being simuliids (Fig. 5.29).

There were no significant differences found comparing total abundance for a particular flow type, chaotic, broken or laminar, for the Manning sites (Table 5.9). There was a significant difference in macroinvertebrate abundance between sites indicating differences between the macroinvertebrate communities found at some of the Manning River sites (Table 5.9).

PERMANOVA results for the reference sites indicated a significant difference between sites and between flow types (Table 5.10). The significant differences between flow type for reference sites suggests that the more sensitive species found at these sites may utilise specific flow types, thereby providing some differentiation, whereas the lack of some of the more sensitive species in the Manning River prevented the determination of community differences between flow types. This is supported by the reduced number of taxa found at the Manning sites to the number of taxa found at the reference sites (Fig. 5.30). The dominance of the simuliids (from 58% to 88% of all taxa collected) found in the Manning indicates conditions were highly favourable for this species at some Manning sites.

To test for significant differences in macroinvertebrate taxa richness among all sites, PERMANOVA results showed that significant differences occurred across Manning and reference sites, however flow types did not demonstrate significant differences when all sites were combined (Table 5.11).

The analysis of functional feeding groups within each site also indicates that collector/filterers dominated macroinvertebrate community structure at all Manning River sites while collector/gatherers and omnivores dominated the reference sites (Fig. 5.31). Very few predators were collected from all sites, while shredders and omnivores were absent from the Manning River sites (Fig. 5.31).

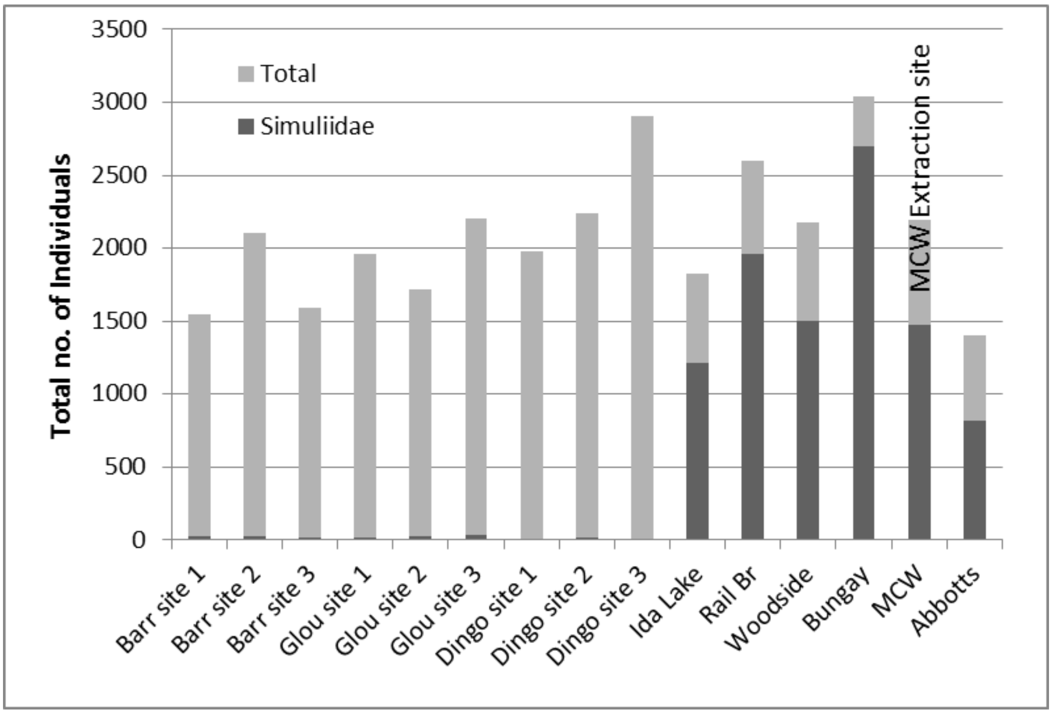


Figure 5.28 Macroinvertebrate Simuliidae and community abundance, Manning River and reference riffle sites

Table 5.9 Results of PERMANOVA main test for significant differences in total macroinvertebrate abundance between Manning River sites and flow types

Source	df	SS	MS	Pseudo-F	P (perm)
Sites	5	5751.7	1150.3	2.645	0.001
Flow Type	2	796.08	398.04	0.44529	0.873
Site x Flow	10	8938.9	893.89	2.0554	0.001
Residual	36	15657	434.9		
Total	53	31143			

Table 5.10 Results of PERMANOVA main test for significant differences in total macroinvertebrate abundance between reference sites and flow types

Source	df	SS	MS	Pseudo-F	P (perm)
Sites	8	23793	2974.1	13.068	0.001
Flow Type	2	3790.9	1895.4	3.6984	0.001
Site x Flow	16	8200.1	512.5	2.252	0.001
Residual	54	12289	227.58		
Total	80	48073			

Table 5.11 Results of PERMANOVA main test for significant differences in total macroinvertebrate abundance between all sites and flow types

Source	df	SS	MS	Pseudo-F	P (perm)
Sites	8	$1.3186e^5$	16482	50.157	0.001
Flow Type	2	1458.1	729.07	0.9411	0.458
Site x Flow	16	14140	883.74	2.6894	0.001
Residual	108	35489	328.6		
Total	134	$1.8461e^5$			

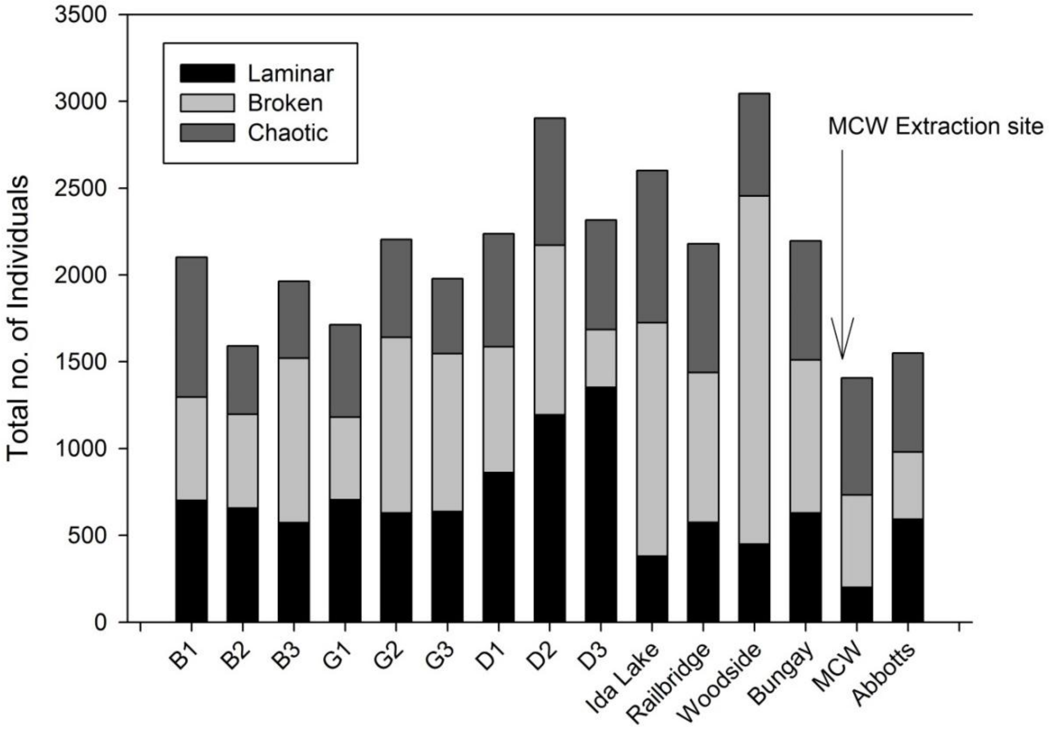


Figure 5.29 Macroinvertebrate total abundance within flow type, Manning River riffle and reference sites

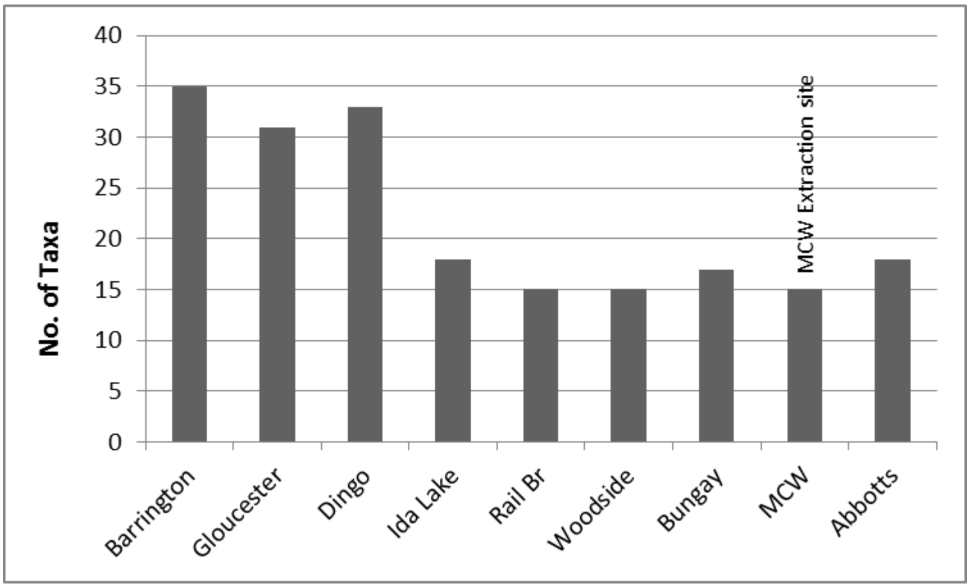


Figure 5.30 Total macroinvertebrate taxa richness at Manning River and reference sites

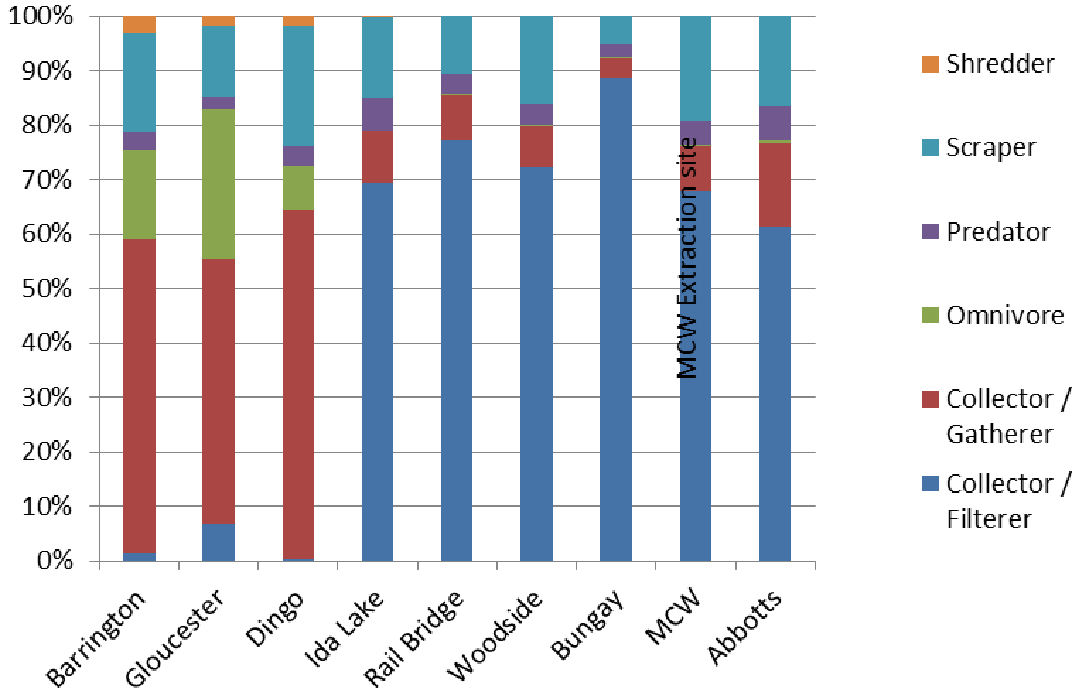


Figure 5.31 Functional feeding group distribution in the Manning River and reference sites

5.3.3 Multivariate analysis of benthic macroinvertebrate communities

Multivariate analysis indicates that the Bungay riffle macroinvertebrate community's domination by simuliids (88% of total abundance) was a feature that separated it from the other Manning River sites (Fig. 5.32). In terms of water chemistry, Bungay had higher levels of N and turbidity (Fig 5.33). Laminar habitats from Ida Lake also demonstrated differences in macroinvertebrate community structure when compared to other sites and flow types (Figs 5.32 and 5.33). There was no relationship between taxa richness or abundance and environmental variables, indicating that flow features may have a greater influence on macroinvertebrate populations at this site (Figs. 5.32 and 5.33).

The reference sites were distinctly grouped when macroinvertebrate community structure with environmental variables were compared (Fig. 5.34). Taxa richness and abundance associated with Dingo Creek, a lower altitude site, correlate with increasing with EC, depth and nutrients (Fig. 5.34). The macroinvertebrate richness and abundance of the two upland sites, Barrington and Gloucester Rivers, were more closely associated with higher DO and pH (Fig. 5.34). Taxa presence also separated the three reference sites, with Dingo Creek having a greater proportion of Ephemeroptera, including Caenidae and Baetidae and Hydropsychidae than the other two reference sites (Fig. 5.35). The Barrington River site had higher numbers of Plecoptera, Simuliidae and Psephenidae species, while the Gloucester River aligned with a number of Trichopteran taxa and chironomids (Fig. 5.35).

ANOSIM was used to determine differences in macroinvertebrate communities between upstream and downstream of the MCW extraction site in the Manning River, and differences between reference sites and Manning River sites. There was no significant difference between upstream and downstream sites of the MCW extraction site (Global R = -0.072; Significance level of sample statistic p = 0.908). When comparing reference and Manning River sites, a significant result was obtained (Global R = 0.986; Significance level of sample statistic p = 0.001). These results indicate that factors, other than MCW extraction, may have a greater influence on macroinvertebrate communities within the lower Manning during low flows. In comparison, the significant difference between reference sites and Manning River sites indicate that sites which have very little extraction and more intact riparian areas, respond differently under low flows.

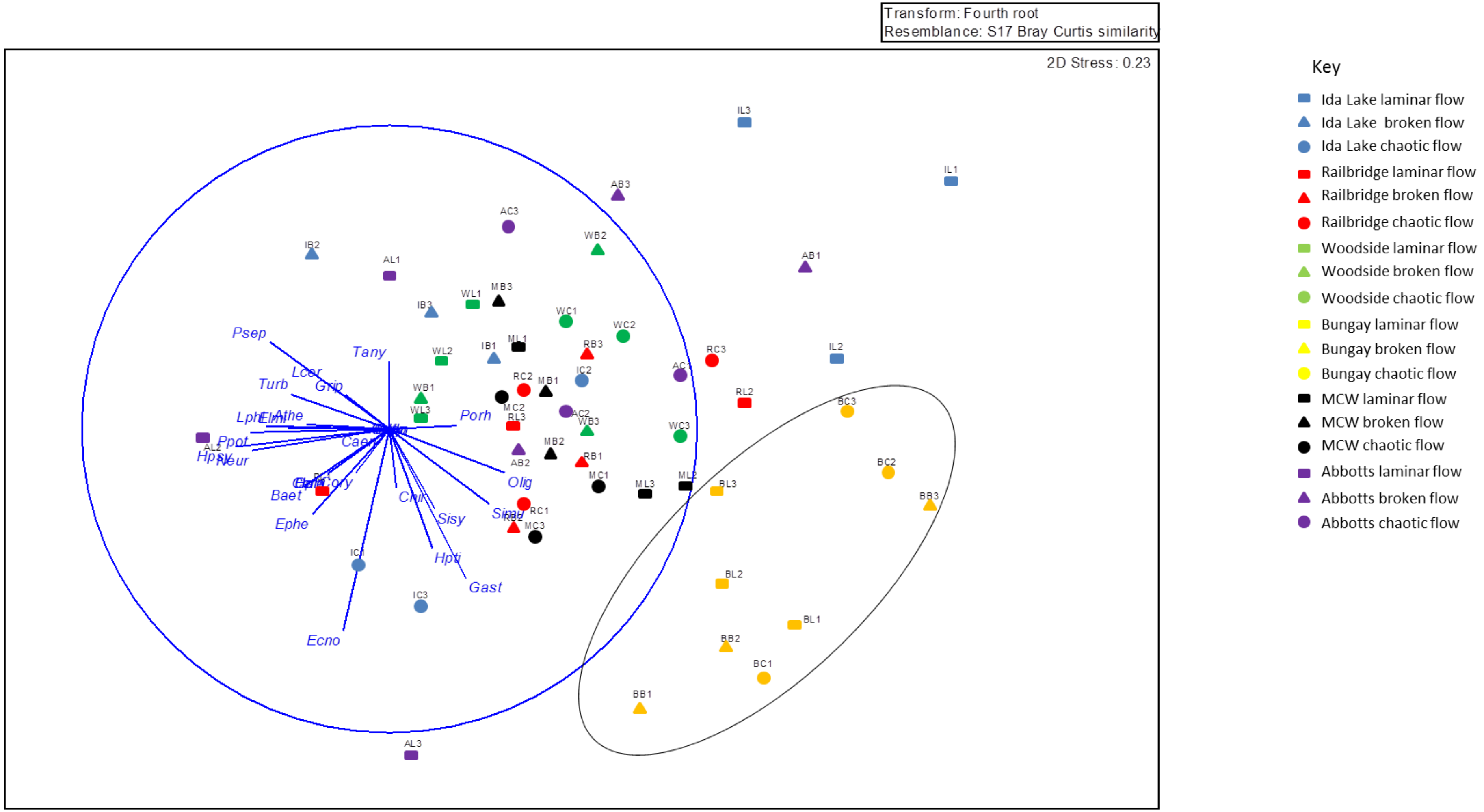


Figure 5.32 MDS plots comparing macroinvertebrate communities collected from 3 flow types at Manning River sites

Macroinvertebrate Key: Turb=Turbellaria; Olig=Oligochaeta; Hphi=Hydrophilidae; Elmi=Elmidae; Psep=Psephenidae; Cera=Ceratopogonidae; Simu=Simuliidae; Athe=Athericidae; Tany=Tanypodinae; Orth=Orthocladiinae; Chir=Chironominae; Baet=Baetidae; Lphi=Leptophlebiidae; Ephe=Ephemerellidae; Caen=Caenidae; Cory=Corydalidae; Neur=Neurothidae; Sisy=Sisyridae; Grip=Gripopterygidae; Hpti=Hydroptilidae; Ppot=Philopotamidae ;Hpsy=Hydropsychidae; Poly=Polycentropodidae; Ecno=Ecnomidae; Porh=Philorheithridae; Cono=Conoesucidae; Odon=Odontoceridae; Calo=Calocidae; Lcer=Leptoceridae; Gast=Gastropoda

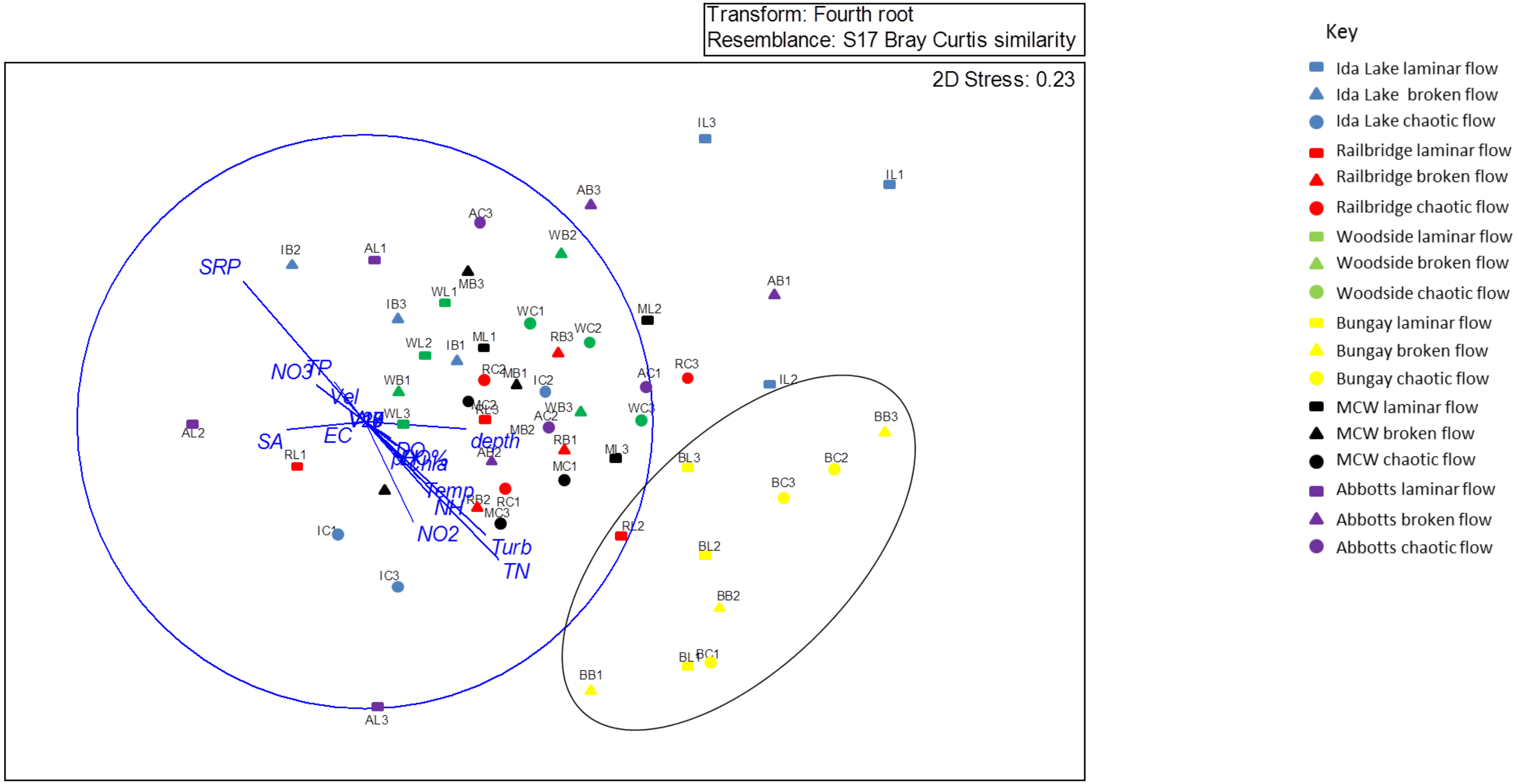


Figure 5.33 MDS plots comparing macroinvertebrate communities collected from 3 flow types at Manning River sites and associations with environmental variables

Environmental Variable Key:Depth=Water Depth; SA=Surface Area;Vel=Water Velocity; Turb=Turbidity; DO=Dissolved Oxygen; DO%=Saturated Dissolved Oxygen;pH; Temp=Water temperature; Chla=Chlorophyll-a; TN=Total Nitrogen, NO2=Nitrite; NO3=Nitrate; NH=Ammonium; TP=Total Phosphorus; SRP=Soluble Reactive Phosphorus.

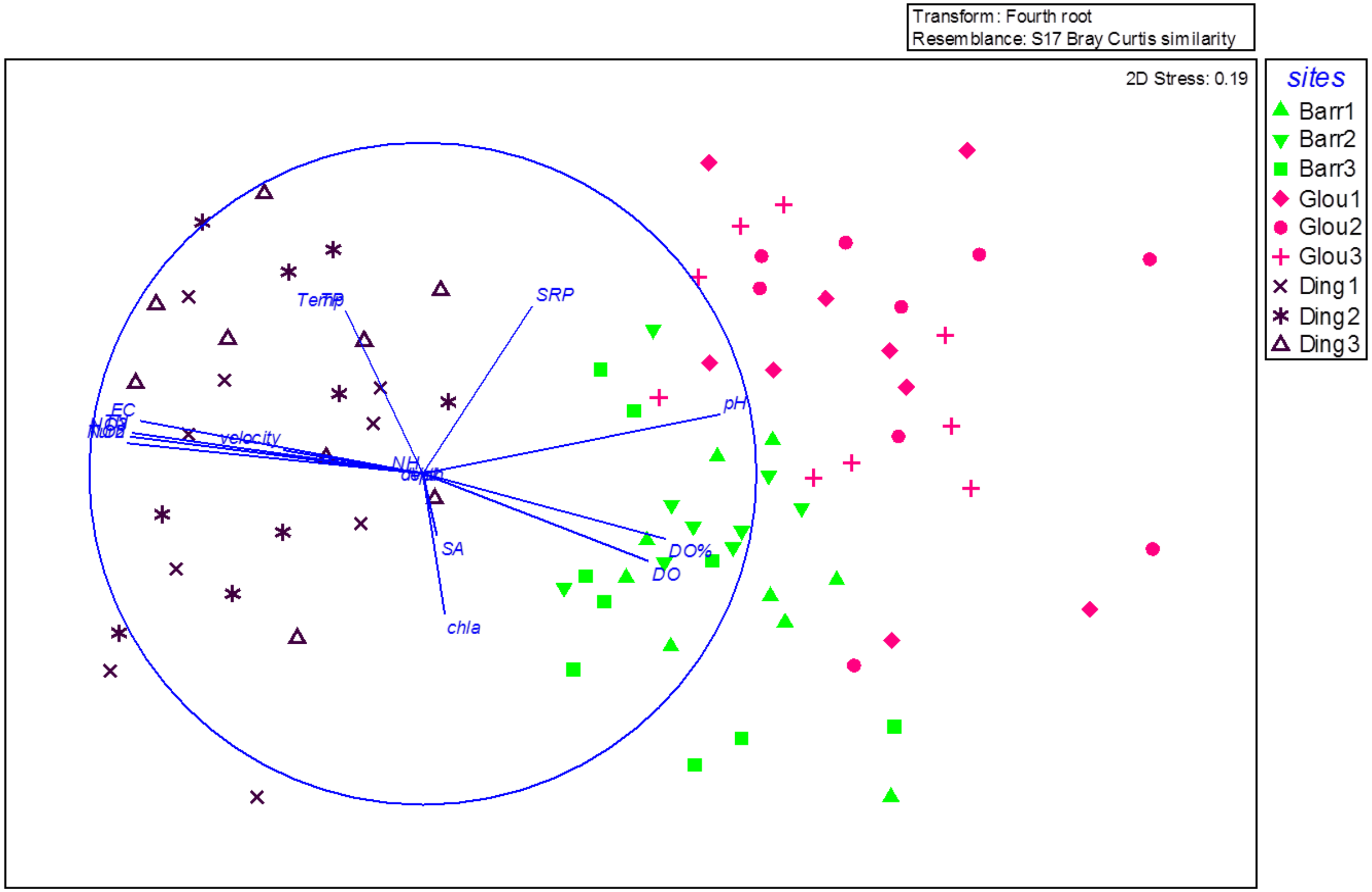


Figure 5.34 MDS plots comparing macroinvertebrate communities at reference sites and associations with environmental variables

Reference Site Key: Barr=Barrington River;Glou=Gloucester River; Ding=Dingo Creek

Environmental Variable Key:Depth=Water Depth; SA=Surface Area;Vel=Water Velocity; Turb=Turbidity; DO=Dissolved Oxygen; DO%=Saturated Dissolved Oxygen;pH; Temp=Water temperature; Chla=Chlorophyll-a; TN=Total Nitrogen, NO2=Nitrite; NO3=Nitrate; NH=Ammonium; TP=Total Phosphorus; SRP=Soluble Reactive Phosphorus.

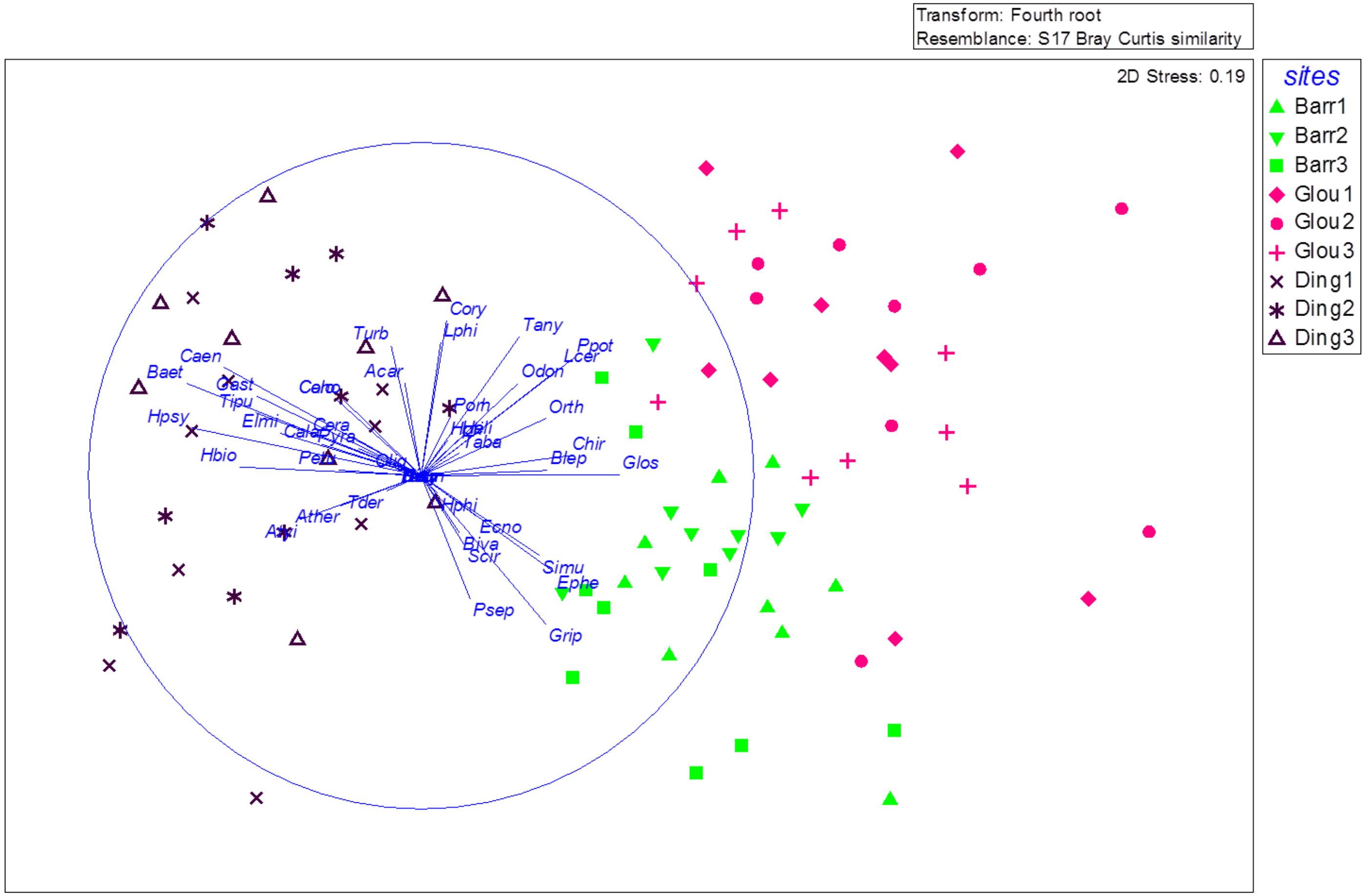


Figure 5.35 MDS plots comparing macroinvertebrate communities at reference sites

Reference Site Key: Barr=Barrington River;Glou=Gloucester River; Ding-Dingo Creek

Macroinvertebrate Key:Turb=Turbellaria; Olig=Oligochaeta; Hphi=Hydrophilidae; Elmi=Elmidae; Psep=Psephenidae; Cera=Ceratopogonidae; Simu=Simuliidae; Athe=Athericidae; Tany=Tanypodinae; Orth=Orthocladiinae; Chir=Chironominae; Taba=Tabanidae; Blep= Blephariceridae; Tipu=Tipuliidae; Baet=Baetidae; Lphi=Leptophlebiidae; Ephe=Ephemerellidae; Caen=Caenidae; Cory=Corydalidae; Neur= Neurorthidae; Sisy=Sisyridae;Grip=Gripopterygidae; Hpti=Hydroptilidae; Ppot=Philopotamidae;Hbio=Hydrobiosidae; Hpsy=Hydropsychidae; Poly= Polycentropodidae; Ecno=Ecnomidae; Porh=Philorheithridae; Cono=Conoesucidae; Odon= Odontoceridae; Calo=Calocidae; Lcer=Leptoceridae; Tasm=Tasmiidae; Glos=Glossosomatidae; Heli=Helicopsychidae; Scir=Scirtidae; Pyra=Pyralidae; Acar=Acarina; Petu= Petularidae; Biva=Bivalve; Atyi=Atyidae; Cala=Calamoceratidae; Tder= Tanyderidae;Gast=Gastropoda

5.4 Discussion

5.4.1 Macroinvertebrate riffle community response to low flows

Under summer low flow conditions, there were fewer observed macroinvertebrate taxa when compared to the predicted number of approximately 23, calculated using AusRivAS (Hose and Turak 2004). The number of taxa collected from the Manning River sites, as part of this study, was between 15 to 18, with some predicted taxa not found at any sites. The Manning River AusRivAS assessment, undertaken in 1998, in close proximity to sites sampled as part of this study, indicated varying degrees of impairment to the macroinvertebrate fauna when comparing observed to expected taxa (Hose and Turak 2004). The greatest decline in macroinvertebrate taxa was found within edge fauna, with less impairment found within the riffle fauna community. Riffle sites were generally classified as AusRivAS Band B, described as significantly impaired with approximately 16 percent to 45 percent of macroinvertebrate biodiversity lost (Gray 2004).

All tributary reference sites had greater numbers of taxa than the Manning River sites, with the 1998 assessment of nearby sites in the Gloucester and Barrington Rivers indicating both sites were within AusRivAS Band A (Hose and Turak 2004). Band A indicates that the macroinvertebrate population is near reference condition with similar levels of biodiversity to reference sites (Gray 2004). During this investigation the reference sites of Barrington and Gloucester had consistently greater numbers of taxa than that predicted by AusRivAS, with around 23 taxa predicted for riffles in these rivers and between 31 and 35 taxa found at each site, with most of the predicted taxa collected from each site. This indicates that the reference sites were not impaired under low flow and were close to reference condition.

Dingo Creek was not included in the Manning catchment AusRivAS assessment. The 33 taxa found over the three sites sampled indicate that the riffle macroinvertebrate community of Dingo Creek is not impaired during the sampling period. The 23 taxa predicted by AusRivAS results for other sites throughout the Manning would likely to also occur in Dingo Creek, which demonstrated similar richness as the upper tributary sites. This indicates that the sites selected for Dingo Creek, as those selected for the Barrington and Gloucester Rivers, would also be considered within reference condition.

5.4.2 Macroinvertebrate response to hydraulics, nutrients and periphyton under low flows

It was predicted that spatial differences between macroinvertebrate community structure would be explained by differences in flow characteristics and nutrient concentrations. Differences in community structure among sites of varying nutrient conditions were assessed using functional feeding groups across the study sites. Results showed that the Manning sites were dominated by collector/filterers. The dominant taxa at these sites were simullids which have been found to prefer suspended particulate organic matter as opposed to fine benthic organic matter (Mulholland et al. 2000).

The domination of filter feeders that prefer suspended organic matter implies that in-situ food resources, such as leaf litter, may be limited (Hall et al. 2000) or flow conditions better suited filter-feeding strategies. Reduced longitudinal delivery of fine-particulate organic matter has been attributed to short term variability in the density and occurrence of filter-feeding macroinvertebrates in drift. In flow-diversion experiments representative of low flow water abstraction in Canada and New Zealand, filter-feeding macroinvertebrates increased significantly immediately after rapid drawdown, then returned to predisturbance levels after a period of less than two months (James et al. 2008, Death et al. 2009).

In comparison, the reference sites had a greater diversity of feeding functional groups with a larger percentage of gatherers and scrapers, with a small percentage of shredders also present. This increased diversity indicates a greater variety of food resources. Other studies also showed that functional feeding groups responded to instream impacts resulting from land use changes (Rabeni et al. 2005, Compin and Cereghino 2007, Buzby and Viadero 2007, Smith and Lamp 2008). Shifts from a variety of functional feeding groups to gatherer/collector dominated groups, with the loss of scrapers and filterers, have been associated with a variety of land use impacts that increase nutrients and sedimentation (Rabeni et al. 2005, Compin and Cereghino 2007, Smith and Lamp 2008). Rabeni et al. (2005) found that a higher percentage of feeding groups was present in lower sediment covered surfaces, with reductions in abundance and richness in most of the functional feeding groups. Findings in this study supports other studies which found that all functional feeding groups were present in undisturbed rivers, whereas disturbed rivers will lose functional feeding groups depending on the type of disturbance (Rabeni et al. 2005, Compin and Cereghino 2007). The low representation of predators at all sites may relate to human-impacts. The relative percentage of predator abundance has been found to decline when periphyton biomass increases in response to increased nutrients (Kerans and Karr 1994, Ortiz et al. 2005). The dense algal mats that developed over summer within riffle areas of the Manning may have impacted on predator numbers.

The dominance of simullids at the lower Manning River sites may be an indication of the influence of increased nutrients on the macroinvertebrate community, particularly under low flow conditions. Zhang et al. (1998) found that simuliid domination is a feature of disturbed sites as they were highly resistant to flow disturbances, while predators and competitors, particularly hydropsychid's, were suppressed. Morin (1991) also found a negative correlation between simullids and hydropsychids biomass as the two taxa compete for similar substrates in fast-flowing streams. Both simullids and hydropsychids made up very small percentages of total macroinvertebrate numbers in Barrington and Gloucester Rivers (1-3 per cent), however in Dingo Creek total numbers composed of 0.03 per cent simuliids and 20 per cent hydropsychids. In the Manning River, simuliids and hydropsychids made up an average of 70.5 per cent and 13 per cent of total macroinvertebrate numbers, respectively. The presence of a greater proportion of hdropsychids (percent of total abundance) in Dingo Creek in comparison to those found in the Manning River, may provide further explanation regarding the dominance of simuliids. The proportional reduction of hydropsychids in the Manning is likely to be related to deterioration in food and habitat quality through increasing siltation from agricultural pursuits.

The distribution pattern of simuliids and hydropsychids may also be influenced by hydraulic differences (Morin 1991, Zhang et al. 1998). The relationship between macroinvertebrate community structure and hydraulic variables within riffles has been explored by Brooks et al. (2005). Brooks et al. (2005) found that small-scale differences in hydraulic conditions created by velocity, depth and roughness are important in spatial distribution within riffle habitats. While this study has not been designed to detect these associations, the reduction of high flow areas and variability in riffles observed under low flow conditions, when compared to those under moderate flow conditions, indicates that community composition is likely to be influenced by these changes and reduction in hydrologic variability.

High N and P concentrations were associated with reduced taxa richness at all Manning River sites. The changes in taxa richness often associated with high algal biomass has been supported by Wang et al. (2007), who also found a strong relationship between nutrient concentrations and macroinvertebrate assemblages in wadeable streams across Wisconsin. High nutrient levels have been found to increase algal growth, particularly filamentous algae, often reducing aquatic biodiversity and habitat heterogeneity within sites (Dodds and Oakes 2004, Wang et al. 2007, Stevenson et al. 2012). The impact on macroinvertebrates from this increased algal growth may be the result of biotic and abiotic changes within a site, as sensitive or specialised macroinvertebrates may not be suited to habitat changes relating to increased periphyton biomass (Buss et al. 2002).

The chemical changes, frequently found in association with high algal biomass, include DO depletion via respiration and increased pH through photosynthesis (Wetzel 2001, Stevenson et al. 2012). These changes in pH and DO are likely to affect sensitive macroinvertebrate species including Plecoptera, Elmidae and Ephemeroptera, which have demonstrated a preference for high dissolved oxygen and low siltation (McCelland and Brusven 1980, Lemly 1982, Brown 1987, Buss 2002, Li et al. 2012, Stevenson et al. 2012). The structure of freshwater macroinvertebrate communities has been found to be strongly influenced by pH, or its related measure of alkalinity (Feldman and Connor 1992, Jurado et al. 2009). More tolerant species such as simuliids and chironomids generally show higher abundance with increasing organic pollution and have shown preference for moderately degraded sites (Buss et al. 2002, Li et al. 2012).

Given that the reference sites were sampled under low flow conditions, as was the case with Manning River sites, it appears low flow alone may not impair the macroinvertebrate community. Other studies have found that the combination of declining water quality has a greater influence on benthic communities than low flows alone. While not always reducing abundance, these conditions appear to greatly reduce taxa richness (Scherman et al. 2003, Lancaster et al. 2009, Rolls et al. 2012). Lancaster et al. (2009) found that macroinvertebrate communities are influenced by food resources and DO gradients to a greater extent than flow. The lower nutrient concentrations are likely to reduce the dominance of filamentous algae at these sites, allowing greater availability of a variety of food resources, reduced substrate smothering and less impact on DO gradients, thereby creating a more variable habitat for macroinvertebrates (Lancaster et al. 2009, Stevenson et al. 2012).

Macroinvertebrate community structure in the Manning River may also be impaired by the smothering of habitat and reduction in the variety of food resources and filamentous algae shading substrate under low flow conditions. The morphology of filamentous algae is likely to influence grazers, with some avoidance demonstrated to unbranched, filamentous species by snails (Steinman at al 1992). Other studies have found that filamentous algae was not commonly an important source of carbon in freshwater food webs (Delong et al. 2001, Reid et al. 2008b). However, filamentous algae can constitute a major carbon component in lowland streams, particularly under low flow conditions (Bunn et al. 2003, Hladyz et al. 2012). Species documented as consumers of filamentous algae include Chironominae and crustaceans, while oligochaetes may use it as habitat (Bunn et al. 2003, Hladyz et al. 2012). It is likely that increase of tolerant species will be to the detriment of more sensitive species that do not rely on filamentous algae and have varied habitat requirements.

While the Manning River sites contained thick mats of filamentous algae, the Barrington River presented the highest benthic chlorophyll-a concentration across all study sites. The Barrington and other reference sites appeared to have a greater diversity of periphytic species, including diatoms, with limited patches of filamentous algae in comparison to the Manning River sites. In experimental streams, reduced nutrient input was found to have little impact on algal biomass but did result in changes in algal taxonomy and increased carbon fixation (Mulholland et al. 1991, Pan et al. 2000). While the large surface area to volume associated with filamentous algae can increase exposure to flowing water and therefore nutrient uptake (Mulholland et al. 1991, Steinman et al. 1992), densely aggregrated algal forms, such as found in the Manning River, may have reduced photosynthetic rates as a result of self-shading and low effective surface area to volume ratios (Dawes et al. 1978, Stevenson 1990, Steinman et al. 1992). Nather Kahn and Firuza (2012) found that increased biomass does not always result in increased chlorophyll-a values, with some dependence on the type and rate of pollution and other physical factors such as river flow and substrate collectively determining biomass and chlorophyll-a content. This highlights the limitations of using chlorophyll-a as a measure of periphyton biomass as it does not address cell concentration, physiological status or taxonomic composition (Steinberg et al. 2012).

5.4.3 Impacts of extraction on macroinvertebrate communities

Environmental flow studies conducted on the Manning River to date, have primarily focussed on salt water intrusion on macrophyte beds and fish passage (Bishop 2012). These studies have indicated that the current WSP conditions have high ecological risks as a result of the paucity of ecological information used to determine an appropriate cease-to-pump-threshold (Bishop 2012). While studies have indicated extraction alone can be detrimental to macroinvertebrate taxa such as Baetidae, Caenidae, Glossosomatidae, Leptoceridae and Elmidae compared to more tolerant taxa such as chironomids and Odonata (Rabeni et al. 2002, Brooks et al. 2011a). Other factors such as water quality and stresses placed on benthic macroinvertebrate communities under low flow conditions should be taken into consideration to improve our understanding and ability to predict the consequences of increased low flows and water temperature on riverine ecosystems. Ecologically-relevant hydrological attributes such as habitat extent, condition, connectivity and water quality, and the major human-induced threats to low flow hydrology (Rolls et al. 2012), need to be assessed to better inform cease-to-pump thresholds for the Manning River.

As the Manning River supplies irrigation users and town water via the off-river storage of Bootawa Dam, extraction may further stress the system, as it is likely to increase the duration and extent of low flows. Current WSP rules protect Manning River low flows by imposing cease-to-pump-thresholds when flows are equal to or less than 1.5 m^3s^{-1} (97th percentile) on a rising river, or equal to or less than 1 m^3s^{-1} on a falling river, which is the estimated 98th percentile (NSW Government 2009). The adequacy of these rules for protecting the aquatic ecological community is yet to be determined, however this study indicates that under warm, summer conditions, benthic macroinvertebrate communities may be altered at 3.5 to 4 $m^3 s^{-1}$ (or around the 90^{th} percentile) discharge at Killawarra.

Climate change needs to be considered as a risk to achieving desired ecological outcomes from current cease-to-pump rules. Temperature, hydrological and chemical changes of river ecosystems have been linked to large-scale climatic patterns such as El Nino and possible climate change impacts (Durance and Ormerod 2007, Durance and Ormerod 2009). Under current climate change predictions, the increase in frequency and duration of low flow events is likely to cause impairment to benthic macroinvertebrate communities (Durance and Ormerod 2009). Coupled with a likely increase in water temperature and nutrients from agricultural inputs, the resulting increased productivity in periphyton, particularly filamentous algae, would further impact on community composition within the lower Manning River.

The eco-thermal nature of macroinvertebrates further increases the influence temperature has on growth, productivity and distribution (Hawkins et al. 1997). However, from their studies in the UK, Durance and Ormerod (2009) have found that by improving water quality, the impact of increased water temperature appear to be reduced. The importance of improving water quality in the face of apparent climate change impacts is an essential step in maintaining benthic communities in the Manning River and its tributaries. The maintenance and enhancement of riparian zones across the Manning catchment would reduce nutrient and sediment inputs to streams and, in the small to mid-sized streams, moderate temperatures (Osborne and Kovacic 1993).

5.4.4 Conclusion

Riverine productivity and the capacity of an aquatic system to recycle nutrients is primarily driven by the magnitude, duration, frequency, timing, and rate of change of flows, and changes in temperature. Understanding the interaction of these variables and the resulting impacts on ecological communities in coastal stream ecosystems is crucial in informing relevant water quality guidelines and cease-to-pump thresholds.

Within the moderately disturbed Manning River sites, periphyton biomass increases under low flow conditions, and the chemical and physical environment is altered for macroinvertebrates through reductions in habitat availability and variability, and changes in food resources. Under these conditions macroinvertebrate taxa richness is reduced, particularly predators. Macroinvertebrate communities did not display the same reduction in taxa numbers within the tributary sites, likely due to lower nutrient concentrations and the absence of dense algal beds. The greater taxa richness found at all tributary sites may indicate a more resilient macroinvertebrate community.

The differences in functional feeding groups between Manning River and tributary sites was highlighted by the dominance of collector/filterers at Manning River sites compared to the dominance gatherers and scrapers at tributary sites. This difference in dominant functional feeding groups is determined by factors such as the greater variety of food resources and reduced substrate smothering by filamentous algae providing habitat variability.

The Manning River results indicate a loss of habitat heterogeneity under low flow conditions. For the management of this system which is relied upon for a number of extractive purposes including town water supply, the reduction of nutrients entering this system is likely to improve ecological outcomes under low flow conditions. Given the scenarios presented as a result of climate change, it is likely that periods of low flow may become more frequent and over greater durations then experienced in the past (CSIRO 2007b). As water temperatures are also likely to increase under climate change, the impact of high nutrients, low flows and increased temperatures will impair benthic macroinvertebrate communities (Durance and Ormerod 2009). These low flow events, if more frequent and of greater duration, may result in the permanent loss of species that are unable to resist chronic impacts.

The vulnerability of aquatic ecosystems to climate change impacts makes these systems useful indicators of environmental change at local and global scales (Williamson et al. 2008). Macroinvertebrates as poikilotherms are particularly vulnerable as their growth, survival and reproduction is temperature dependent (Woodward et al. 2010). Their dominance in stream biodiversity and their central role in ecosystem functioning, make them key sentinels and integrators of climate change. As indicators of “ecosystem health”, with health based on pressure-state-impact-response, macroinvertebrates can inform the systems overall response to climate change impacts. Improved understanding of the resistance, resilience and directional responses of streams to climate change will better inform effective natural resource management (Williamson et al. 2008).

Chapter 6 Synthesis

This thesis aims to describe interactions between low flows and biogeochemical pathways, and the resulting influences on ecological communities in a coastal stream ecosystem to inform the development of sustainable extraction limits. This was undertaken by measuring discharge, longitudinal position, seasonal and diurnal temperature, nutrient changes and ecological responses to those changes (Table 6.1). The feasibility of water extraction thresholds were assessed under current water quality conditions, as influenced by land use, and the likely impacts on dependent biota given the biophysical characteristics of the system.

While concentrations of total nutrients were highest under high flow conditions, it was under summer low flows that nutrients shifted to inorganic forms, becoming bio-available and leading to the greatest ecological response. The decline in dissolved oxygen concentration and release of nutrients to the hypolimnion within thermally-stratified deep pools not only indicated a reduction in available aquatic habitat for aquatic biota such as macroinvertebrates and fish, but also transformed nutrients from organic to inorganic forms. This has been found to occur elsewhere within Australian waterbodies, with increased flows or large decreases in air temperature required to break down persistently stratified pools (Condie and Webster 2002, Turner and Erskine 2005, Reinfelds and Williams 2012).

The physical, chemical and biological responses of the Manning River and its dependent aquatic ecosystem demonstrated the degree to which catchment properties will impact on system resilience under summer low flows. Using water quality information from reference sites to determine regionally-relevant thresholds, non-reference sites reflected land use influences through the degree to which they deviated from the thresholds. The response of biota to summer low flows is strongly influenced by nutrient concentrations at the subcatchment level.

Table 6.1 Summary of findings for the Manning River system

Hypothesis	Design	Outcome
Catchment properties (geology and agricultural land use) increase total nutrient concentrations	• Water quality determined for reference and non-reference sites using historical and recent data • Reference sites used to determine regionally-relevant thresholds • Non-reference sites compared to thresholds • NDS deployed to assess nutrient limitation in upland streams	• Geology had a greater influence on nutrient concentrations in reference site compared to non-reference sites • Regionally-derived thresholds were influenced by catchment properties. TP and upland NOx thresholds increased from guideline thresholds • Threshold exceedances by non-reference sites were influenced by the magnitude and extent of disturbance within sub-catchments • N may be limiting and C may suppress periphyton growth in upland streams
Dissolved nutrient concentrations increase under summer low flows	• Physical and chemical behaviour in pools in relation to discharge and other environmental variables assessed • Molar ratios determined under low flows	• Biogeochemical interactions, stratification and nocturnal processes under low flows increased dissolved nutrients within Manning River pools

Reduced flow and increases in nutrient concentrations has ecological impacts by increasing primary productivity and biomass	• Observations of aquatic plant responses under different flow conditions • Periphyton samples from reference and non-reference riffle sites compared	• Extensive macrophyte beds were identified in the lower Manning River under low flows • Filamentous algae dominated lower Manning sites under low flow conditions
Biomass increases alter the chemical and physical environment for macroinvertebrates, reducing taxa richness.	• Relationships between nutrients, chl-a and macroinvertebrate community structure at reference and non-reference sites assessed	• Dense algal mats within riffle areas under summer low flows impacted on predator numbers • Macroinvertebrate taxa richness was reduced in Manning River riffles under low flow conditions when compared to tributary sites
Low flows and increased nutrients reduce temporal habitat availability and heterogeneity, thereby reducing the ability of aquatic communities to resist or recover from disturbance.	• Physical changes at sites under moderate and low flows determined and compared • Macroinvertebrate community response at tributary sites compared to modified Manning River sites compared	• Habitat heterogeneity was reduced under low flows • Manning River riffle sites were dominated by collector/filterers. • Tributary reference sites had greater diversity of feeding functional groups and a larger percentage of gatherers and scrapers • Increased diversity at tributary sites indicates greater resilience

6.1 Physical influences and chemical responses

The Manning River is typical of many coastal rivers located in Eastern Australia, most of which have received limited scientific attention. The physical setting of the Manning catchment varies considerably; encompassing relatively pristine upland areas, to the downstream sections of its floodplain where landuse is dominated by agricultural impacts. With varying geology, discharge, altitude and land use, it is important to determine how catchment characteristics influence ecological responses to low flows. Once an understanding of spatial variability and patterns in these processes have been determined across the catchment, this information can more closely reflect relevant spatial scales for both water quality thresholds and minimum flow requirements to protect or enhance dependent ecological communities.

With declining flows and reductions in velocities, persistent stratification in reaches containing deep pools is a common occurrence within the Manning River and throughout Australian rivers (Western et al. 1996, Turner and Erskine 2005, Reinfelds and Williams 2012). Turner and Erskine (2005) found that under stratified conditions, P concentrations increased substantially below the oxycline in NSW rivers. While the contribution of stratification to algal blooms is well recognised for inland regulated rivers (Webster et al. 1997, Sherman et al. 1998, Davis and Koop 2006), this contribution remains unidentified and under-estimated for most unregulated rivers in Australia (Turner and Erskine 2005).

Spatial variability and the transport of nutrients appear to be strongly influenced by the location and size of pools throughout the Manning. These findings correspond with other studies that have documented the major influence geomorphic features have on the availability of nutrients throughout river systems (Western et al. 1996, Turner and Erskine 2005, Reinfelds and Williams

2012). Impacts to the immediate area are greatest under low-flow conditions, with localised effects evident within both deep and macrophyte-dominated, shallow pools as a result of hydrological and biological interactions. It is under high-flow conditions that downstream environments are impacted by processes that take place within the pools under low flows.

The increased influence of local processes in pools under summer low flows when compared to conditions under moderate to high-flow conditions was also found by Sheldon and Fellows (2010), in their study on dryland rivers. They determined that spatial variability in water quality was greatest under no-flow conditions when local processes were increasingly important (Sheldon and fellows 2010). Local spatial variation in nutrients was shown to be driven by catchment characteristics such as local geology and land use, while larger spatial differences were driven by landscape-scale processes, reflecting the interaction of geology, water and atmosphere through which rivers flow (Sheldon and Fellows 2010).

Large pool areas in the lower Manning River are major abiotic sinks for nutrients as they travel downstream. The deep pools reduce flow velocity resulting in attached nutrients and particulate matter settling out. Once thermal stratification occurs under warm, low-flow conditions, microbial respiration releases nutrients from sediments, these nutrients then become available for primary producers when the photic depth extends to the hypolimnion or once turn-over occurs. Inorganic nutrients would become available to downstream primary producers or for uptake by sediment following turn-over of the hypolimnion. The contribution of these temporary sinks to nutrient loads within the Manning has not been quantified and is likely to be highly variable, with deep pools contributing the greatest concentration of available nutrients to downstream environs when stratification breaks down.

The location of the deepest pool, Ida Lake, immediately downstream of the confluence of the Manning and Gloucester Rivers, one of the largest tributaries to the Manning, is likely to increase the deposition of sediment and nutrients, in comparison to upstream environments. The role of fluvial processes has relevance for riverine ecology as biota and ecosystem functions adjust to the downstream changes of hydraulic and geomorphic processes (Benda et al. 2004). Tributaries alter the flux of water and sediment into the main river channel through increased frequency and magnitude of sediment inputs (Benda et al. 2004, Svendsen et al. 2009). Ida Lake would receive more frequent and larger volumes of sediment and attached nutrients as a result of its position in the stream network.

The second deepest pool, Bungay, which also became strongly stratified under low flows, is located immediately upstream of the MCW off-take for Bootawa Dam. The proximity of the deep pool may impact on downstream water quality when the breakdown of stratification occurs, influencing the quality of water extracted for use in the Dam. It has been found elsewhere in Australia that when seasonal mixing of these pools occurred, episodic pulses of P and other elements, such as hydrogen sulphide and iron, were released downstream (Turner and Erskine 2005). It was this periodic contribution of nutrients that was thought to drive cyanobacterial blooms in the Nepean River, NSW (Turner and Erskine 2005).

Under reduced flows, low-velocity (0.1 m s^{-1}) river environments have been closely associated with patches of low TN, high TP, and low TN:TP (De Jager and Houser 2012). Longitudinal profiles measuring the impacts on water quality released from stratified pools to downstream environments have also been associated with increasing nitrification in comparison with carbon-based oxygen consumption in a downstream direction (Becker et al. 2010). This supports the idea that local geomorphic and hydraulic conditions influence spatial patterns of nutrient delivery and biochemical transformation and create a mosaic of nutrient distributions within large floodplain rivers (De Jager and Houser 2012). These are important components of nutrient cycling (Fig. 6.1) and it is through the examination of the origin and pathways of nutrients that catchment and water management can be better informed.

The macrophyte-filled shallow pools located throughout the lower Manning River also influence nutrient pathways. The dense macrophyte beds reduce water velocity, settling out particulate matter, the macrophytes and associated epiphytic algae also facilitate nutrient uptake, further aiding the temporary retention of nutrients. Once the macrophytes and associated algae begin to breakdown, which was observed to accelerate under low flows and warm conditions, nutrients are again available to travel downstream (Fig. 6.1). The large biomass of plant material within the numerous shallow-pools located throughout the lower Manning reduces dissolved oxygen when decaying and contribute significant amounts of available nutrients during this process. As occurs in deep pools, this break down of macrophytes and increased nutrient contribution tends to occur throughout all the pools at the same time, thus generating a pulse of available nutrients and increasing respiration, possibly resulting in deoxygenation, further contributing to poor water quality (Fisher and Grimm 1988).

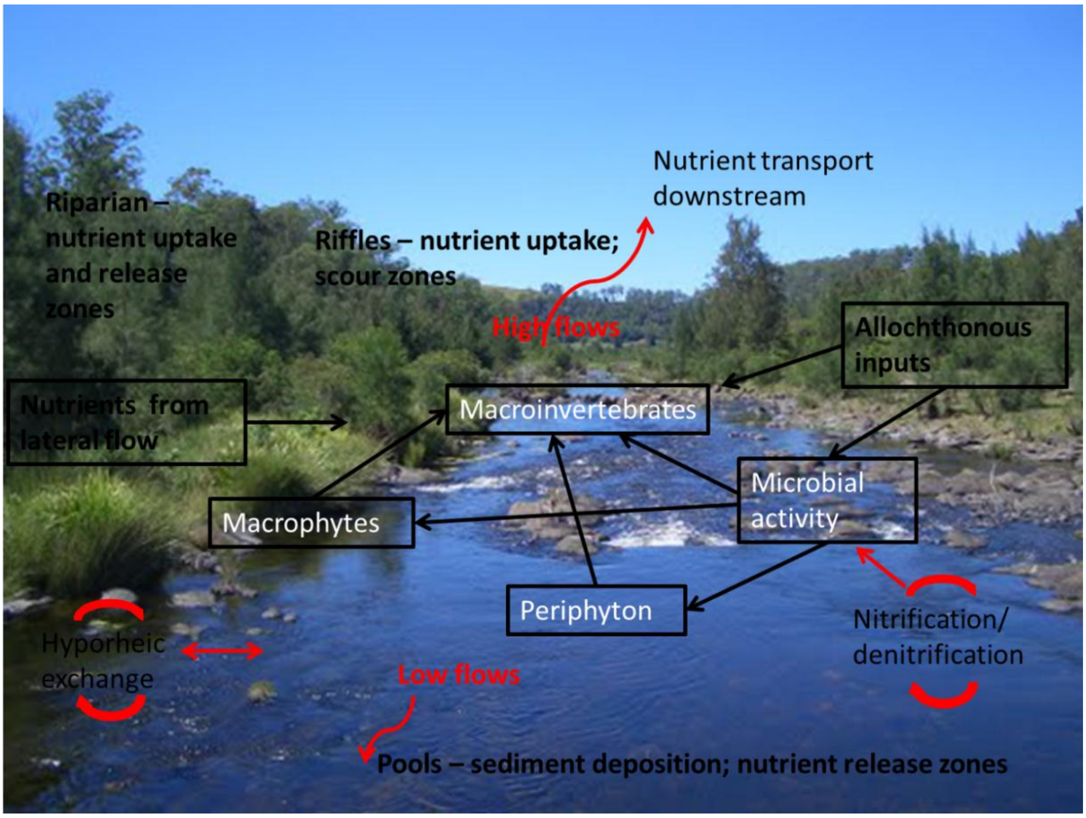


Figure 6.1 Interactions of biophysical components and nutrients in an upland stream
(Black arrows indicate direction of nutrient transport; red arrows indicate influences on type and distribution of nutrients)

Kleeberg et al. (2010) found that, rather than acting as important sites for the retention of nutrients, the key role of submerged macrophytes in lowland rivers is more directly related to seasonally modifying the equilibria which exists between vegetation trapping sediment and resuspension. Further to this, O'Brien et al. (2014) asserted that while macrophytes drive ecosystem metabolism and had a large impact on trophic state, they had limited influence on water column nutrient concentrations, therefore offering little influence on nutrient removal. This assertion provokes reconsideration of the role these areas play as nutrient sinks within the lower Manning River, and their long-term influence on water quality.

The regionally-relevant thresholds derived to assess water quality in relation to the protection of aquatic ecosystems were frequently exceeded throughout the Manning catchment. The extent to which these thresholds were exceeded depended on the magnitude and extent of land-use change and hydrological disturbances within sub-catchments. It was found that nutrient concentrations in less modified upper catchments more closely reflected local geology, soil type and composition, while areas which have a greater extent of land use change reflected the increased anthropogenic influences. Increased nutrient concentrations found under low flow conditions at some reference sites in the upper catchment areas was explained by geological or groundwater influences which are more evident under low-flow conditions at less disturbed sites, particularly sites with underlying basalt geology which is associated with increased levels of phosphorus

Apart from landscape influences, interactions between flow, sediments, autotrophic and heterotrophic processes greatly influence nutrient availability. These interactions highlight the need to consider instream processes as regulators of nutrient concentrations, particularly under low flow conditions when runoff and other external influences play less of a role (Mulholland 1992). The differences in nutrient dynamics between upland and lowland sites in the Manning catchment centre around the influence of the mostly intact riparian zone in upland reaches compared with the much greater influence from the instream autotrophic component in the lower Manning. Nutrient dynamics within these two zones varies significantly over time, particularly in relation to discharge and temperature.

Non-reference upland tributaries located in the upper catchment, particularly the Avon and Barnard Rivers, were also found to have nutrient concentrations that exceeded regionally-derived thresholds. The Avon and Barnard rivers, modified through mining and agriculture respectively, contributed greater nutrient concentrations under high discharge compared with less modified sub-catchments. The degree to which this increased nutrient input would impact on the Manning River is dependent on flow conditions both within the tributaries and the Manning River itself.

Increases in discharge are linked to higher concentrations of nutrients, and likely transport of nutrients through the system and deposition to pools following flow recession. Water quality can be strongly influenced by sediment resuspension which is associated with increased turbidity, the enhancement of nutrient recycling and the promotion of algal growth in the photic zone. Resuspension of sediment contributes not only to light attenuation by increasing particle content within the water column, but also increased phytoplankton growth due to enhanced nutrients in the water column (Madsen et al. 2001).

While resuspension reduces light penetration, it increases the available nutrients in the water column, including the photic zone, which promotes phytoplankton growth. This indicates that resuspension is likely to be an important driver of primary production in lowland rivers, where the transport of sediment-associated P often constitutes a high percentage of the total annual P flux (Svendsen et al. 1995). Erosion and deposition are other processes likely to influence primary production within streams and also relate strongly to geomorphic features and interaction with hydrology. Pool areas, subjected to deposition, may be more prone to thermal stratification and nutrient release and availability to downstream environments.

6.2 Nocturnal patterns

When the nocturnal behaviour of nutrients was examined in both the deep and macrophyte-dominated shallow pools, changes were noted in the concentrations of available nutrients and nutrient ratios. The Manning River nocturnal study indicated that changes in biological activity and flow can result in an alteration in water quality and nutrient behaviour, in deep sections of stratified pools and the substrates of shallow macrophyte-dominated pools. Both study sites exhibited temporal changes in nutrients over the sampling period, driven by changes in oxygen and pH, and increased respiration. SRP was most concentrated within the thermocline of the deep pool, while TP doubled in concentration during the 12-hour sampling period within the epilimnion. There was also an overall increase in TP within the shallow macrophyte sites during the nocturnal study. Similarly to TP, NH_4^+ and TN increased within the surface layers of the deep pool, while TN decreased and NO_x increased during the night within the shallow-macrophyte-dominated pool.

These changes in nutrient concentrations resulted in variations in nutrient ratios that can result in the alteration of biological uptake rates, thereby impacting on periphyton communities and nutrient availability within streams (Sardans et al. 2012). The relatively low N:P ratio (i.e. <16:1) within the shallow pool and throughout all zones of the stratified deep pool, indicated N may be a limiting nutrient within these pools (Bothwell 1985, Neill et al. 2001). Measured N:P ratios were greatly reduced in the epibenthic zone, with ratios of around 2:1, indicating that N may have been limiting despite high nutrient concentrations. Chessman et al. (2006) also found that, despite streams on the NSW south coast being nutrient enriched, N limitation occurred across a range of streams including those impacted by agriculture and urbanisation.

Changes in nutrient concentration, speciation and ratios may impact on the flow and balance of energy in ecological interactions within features such as deep pools and in downstream environments (Frost et al. 2002, Giling et al. 2012). Alterations to elemental ratios can affect stoichiometric interactions between primary producers and their consumers, favouring some taxa, resulting in changes to community structure, and impacting on nutrient fluxes (Stelzer and Lamberti 2001, Singer and Battin 2007). Stelzer and Lamberti (2001) found that the community structure of benthic algae, particularly diatoms, was substantially altered by changes in N:P ratios. This suggests that periphyton community structure is sensitive to the relative proportions of nutrients within streams, and changes to this balance that occur through physical, chemical and biological interactions, can impact on this structure and dependent consumers. The consequences of elemental imbalances between instream producers and their consumers include changes in consumer growth rates, influences on food-web structure and impacts to nutrient recycling (Stelzer and Lamberti 2001, Frost et al. 2002, Giling et al. 2012).

Nocturnal patterns determined as part of this study found that changes in ratios can occur over relatively short-time frames over 12-hour periods, or may relate to longer time-frames through seasonal influences. These changes may in turn influence the ecological structure and nutrient patterns within the Manning.

6.3 Biota and summer low flows

6.3.1 Biotic interactions

The dense macrophyte beds and thick algal mats observed throughout the lower Manning River indicate that nutrients are in sufficient concentrations to support extensive primary production. While instream nutrient concentrations are a major influence on the distribution and community structure of the primary producers located in the Manning River, algae and macrophytes in turn influence nutrient cycling and hydraulic characteristics (Clarke 2002). The extensive macrophyte beds located in the lower Manning would play a major role in the chemistry of the fluvial sediments through the oxidisation of the surrounding substrates (Sand-Jensen et al. 1982). This influence on sediment chemistry and the utilisation of nutrients by the plants themselves results in significant impacts on instream nutrient dynamics (Clarke 2002), as nutrients are transformed within these pool areas and transported downstream.

The extent different groups of primary producers such as macrophytes, benthic algae and phytoplankton have on nutrient uptake and hence water quality and other biota is influenced by flow and season. Nutrient concentrations within plant tissues vary between seasons, usually declining as plants age, altering the plants status as a source or sink for nutrients (Bernard and Hankison 1979). Studies on streams in New Zealand found within-stream variation was large and different for each nutrient and may be useful in understanding factors that drive nutrient uptake in streams (Simon et al. 2005). Low flows, even short-term, combined with high summer temperatures, generally results in unfavourable conditions for macrophytes and associated epiphytes resulting in decay and limited nutrient uptake. However, in terms of providing aquatic habitat, Hearne and Armitage (1993) suggest that macrophytes have the ability to supplement water resource availability by maintaining wetted perimeter under low flow conditions, through impounding effects. If this is the case in the Manning, the presence of large areas of macrophytes may assist in maintaining deeper habitats under low flows as refuge areas for aquatic biota.

Short-term dry periods not only alter the influence macrophytes may have on nutrients, but also regulates ecosystem process rates (Hart and Finelli 1999, Poff and Zimmerman 2010). Rates of release of solutes such as nutrients can be sensitive to hydrologic changes, as sinks such as riparian soils shift to sources under drier conditions (Freeman et al. 1993). The capacity of riparian areas to supply essential ecosystem services can be diminished under dry conditions as a result of reduced hydrological connectivity (Capon et al. 2013). This makes it particularly important to ensure water is retained in riparian areas through the protection of subsurface flows which can be achieved through planning for riparian ecosystems in a landscape context, considering both catchment processes and connectivity (Capon et al. 2013).

Riparian zones tend to have a greater influence in upland streams, with intersecting hydrologic flow paths producing dynamic moisture and biogeochemical conditions that can facilitate the retention of nutrients (Fig. 6.1) (Vidon et al. 2010). Spatial and temporal variability in soil moisture conditions, redox potential, vegetation and temperature has a substantial influence on the fate and transport of solutes through riparian zones (Vidon and Hill 2004). This variability in biogeochemical conditions in riparian ecosystems has important implications for the speciation and availability of nutrients (Dahm et al. 1998, Vidon et al. 2010). The importance of the upland riparian zones in regulating nutrient supply and retention to the lower Manning River emphasises the importance of managing these zones to ensure riparian vegetation is retained. While fixed-width buffers of native vegetation have been a primary tool for the protection of aquatic ecosystems, a need to vary widths according to ecosystem requirements has been identified (Richardson et al. 2012). Landscape-level considerations need to be included in site-specific guidelines to ensure buffers are of appropriate size to maintain ecological functions (Richardson et al. 2012).

Large proportions of total annual nutrient loads from riparian areas occur during major storm events. Lewis (2002) found that runoff was the dominant control on nutrient flux from minimally disturbed catchments, such as those located in the upper Manning. Differences in seasonal flow regimes and storm-driven variation in discharge have been found to be of greater importance in determining nutrient concentrations than the longitudinal variation in nutrient processes (McNamara et al. 2008). Intense storms have been found to produce almost-instantaneous shifts in nutrient cycling pathways; these rapid changes in nutrient dynamics have a major impact on rates of biogeochemical transport, retention and transformation (Dahm et al. 1998). Increased flows provided by intense runoff events and floods deconstruct stratified layers, resulting in mixing within pools, scouring of periphyton, transport of macrophytes and sediment (Biggs and Close 1989, Robinson et al. 2004, Turner and Erskine 2005). These disturbances, while of short duration, have a major influence on nutrients and reliant biota within the Manning River.

Unlike most upland sites with intact riparian zones, the lower Manning River exhibited signs of eutrophication, with sites exceeding regionally-derived thresholds for TP and TN. Increased nutrient concentrations during periods of high discharge indicated that diffuse pollution from land use such as agriculture was the major contributor of nutrients to the lower Manning River, rather than other catchment factors such as geology. The influence of altered land use and the significance of its impact on water quality within catchments is well-documented (Young 1996, Prosser et al. 2001, Beckert et al. 2011), with diffuse pollution known to degrade aquatic ecosystems through excessive nutrient inputs resulting in trophic shifts towards eutrophication (Carpenter et al. 1998, Dodds and Oakes 2008).

The impact of short-term increases in discharge on the input of nutrients to the system and the response by aquatic biota was not explored in this thesis. Disturbances such as flooding have been shown to increase the processing length of materials within affected streams (Fisher at al 1998). This increase in processing length, though dependent on the form of nutrients and retention capacity of the stream, acts as an impediment to efficient cycling of nutrients (Fisher et al 1998). The ability of a system to recover from these changes is thought to be dependent on its status prior to the disturbance, including the degree of regulation and nutrient-status (Dodds et al. 2010, Cortez et al. 2012). Future research on coastal systems needs to address how nutrient status prior to disturbance influences its ability to recover from short-term disturbances.

6.3.2 Biotic responses

Water level and water temperature appear to have the greatest impact on physical and biological processes. During 2007, low-flow conditions resulted in prolific algal and plant growth throughout the Manning River (Fig. 6.2). High flows, as occurred in December 2007, delivered large amounts of sediment and debris to the lower Manning River, which was likely to have been deposited within the pools once flow reduced. Once deposited within the pool habitats, deoxygenation through increased respiration results in nutrient release from sediment for redistribution within the system and uptake by aquatic plants (Fisher and Grimm 1998). This response was evident in the deep pools of the Manning River, with deoxygenation under stratified conditions resulting in increased concentrations of nutrients.

Figure 6.2 Macrophytes and algal responses to flows in the Manning River, 2007

My study of Manning River pools was undertaken in summer, when biological activity and nutrient uptake is at its peak. It is during this period of high biological activity that nutrients may be limited, and this is supported by the indication of possible N-limitation in upland tributaries during the summer. Other studies have identified large internal nutrient sources accumulate and recycle on a seasonal basis, driven by catchment inputs, hydrodynamics and biological production (Lowrance et al. 1984, Flynn 2008). It is important to understand this seasonal variation in nutrient sources and sinks to help determine possible impacts of climate change.

Short-term low flow periods, lasting several weeks, occur on a regular basis in the Manning. Under these conditions, water quality often deteriorates and habitat availability is reduced through stratification within pools, reduction in inundated area and algal proliferation. While many Australian aquatic species are adapted to seasonal droughts (Boulton 2003, Humphries and Baldwin 2003), ecological impacts can be exacerbated by water extraction and poor water quality

(Puckridge 1999, Sheldon et al. 2002). Cooper et al. (2013), when looking at Mediterranean-climate streams, found that land use changes increase the length of dry season flows, through introducing more rapid runoff and decreased groundwater recharge. The extension of dry periods from land use change may also increase contaminant concentrations, foster higher temperatures, lower dissolved oxygen and facilitate the accumulation of detritus, algae and plants (Cooper et al. 2013). These influences are likely to adversely impact on sensitive native species and observations from the Manning indicates similar responses to low flows as those described above.

To persist under such conditions, aquatic species have acquired matched resistance and resilience adaptations (Hershkovitz and Gasith 2013). To ensure species persistence, heterogeneity is required to provide a range of suitable physical habitats even under extreme low flow or extended low flow periods (Chester and Robson 2011, Hauer et al. 2013). While community persistence depends on the integrity of these available refuges, it also depends on the restoration of connectivity to allow migration to occur from refuges to new habitat patches (Robson et al. 2013). Therefore the focus on conserving low flow or drought refuges needs to be expanded to protecting connectivity between these refuges through reducing impacts such as water extraction (Robson et al. 2013). In unregulated systems such as the Manning, this can be achieved by the protection of low flows through cease-to-pump rules.

The overall impact of nutrients and low flows on riffle macroinvertebrate communities indicates that higher nutrient levels reduced community richness, possibly magnifying the impact of low flows. The negative relationship encountered between macroinvertebrate assemblages and nutrients in previous studies shows that nutrients are a key factor in determining biological health (Miltner and Rankin 1998, Wang et al. 2007). As a result of this relationship, consideration of catchment health generally, or water quality specifically, needs to be taken into account when considering minimum flows for biotic communities. This relationship is just one of the complex links that exist between low flows, the physical aquatic habitat, habitat conditions, connectivity, sources and exchange of energy (Dewson et al. 2007b, Dunbar et al. 2010, Rolls et al. 2012).

The natural capacity of many aquatic species to recover from drought has been impaired as a result of catchment and hydrologic disturbances. A better response to drought is to ensure its likely occurrence is incorporated into long-term planning strategies (Reid and Ogden 2006). This includes ensuring an appropriate distribution between consumptive and environmental needs during drought and non-drought periods and improving catchment conditions to improve resistance and resilience of macroinvertebrate communities (Boulton 2003, Suren et al. 2003).

The intact riparian vegetation associated with the tributaries used as reference sites for the development of regionally-derived water quality thresholds appeared to affect tributary water quality through its influence on nutrients. Riparian land use has been found to be a better predictor of water quality than catchment land cover as a whole (Osborne and Wiley 1988, Gregory et al. 1991). Dodds and Oakes (2008) found that, though the influence of riparian condition on water quality diminished as stream order increased, the impact on water quality from upper tributaries was still measurable on downstream receiving waters. This influence remained evident under low flows, indicating the importance of connectivity in catchment-wide water quality.

Fellows et al. (2006) confirmed the importance of the influence of small streams on nutrients, particularly dissolved N being transported downstream. It was found that the status of catchment vegetation controlled nutrient retention in forest streams, with this process supported by allochthonous carbon (Fellows et al. 2006). Similarly, studies comparing forested and logged streams found changes in stream nutrient retention efficiency, with an increase in NO_3-N when riparian vegetation was removed (Sabater et al. 2000). Disturbance to riparian vegetation appeared to affect stream nutrient fluxes and retention efficiency which demonstrates the considerable influence riparian zones have on stream ecosystem functioning (Meyer et al. 1998, Sabater et al. 2000). This riparian influence on ecosystem functioning was reflected in water quality responses measured in the upper Manning catchment.

It is important to identify factors that regulate periphyton nutrient status, as these attached benthic communities can be a significant sink for nutrients (Dodds et al. 2002). Despite the frequent exceedance of nutrient thresholds and the obvious proliferation of aquatic macrophytic plants throughout the lower Manning, the nutrient enrichment experiment identified that benthic autotrophic production within upper catchment sites was greatest in nutrient treatments that contained N.

Nutrient spiralling concepts suggest that an increase in biotic uptake of dissolved nutrients is likely if biota are limited by a particular nutrient (Newbold et al. 1981, Fisher et al. 1998). This can directly relate to nutrient availability for downstream transport, as limiting nutrients have been found to reduce uptake lengths (Newbold et al. 1982, Hamilton et al. 2001). Factors that alter stream nutrient ratios, such as riparian vegetation removal are likely to influence the level of retention within ecosystem compartments by causing shifts in stoichiometry (Dodds et al. 2004). As more efficient nutrient uptake can occur at lower concentrations in upland streams, particularly in forested upper catchments (Valett et al. 2002, Dodds et al. 2004), sites downstream of intact riparian areas may be subject to nutrient limitation.

In the case of the upland tributary sites where nutrient-diffusing substrates were deployed, likely nutrient sources include terrestrial inputs from the surrounding vegetation and groundwater contributions. The associated intact riparian vegetation is known to facilitate the retention of nutrients via biological and geochemical processes that occur in the upper soil profiles (Woods et al. 1991, Mulholland 1992), with the major pathway for nitrogen movement through sub-surface flow (Hill 1995).

The capacity of riparian areas to remove N is dependent on hydrological characteristics, such as retention time, and biological processes such as plant uptake (Groffman et al. 1996). Dodds et al. (2004) found that C:N ratios, which may be influenced by riparian vegetation, explained a substantial part of the variation in N-specific uptake rates, so that changes in these ratios through the removal of riparian vegetation could result in changes in productivity and N uptake. Similarly, Giling et al. (2012) found that, for streams in south-east Australia, a reduction in riparian vegetation resulted in an alteration in stoichiometric interactions, with impacts on nutrient cycling and consequences for the balance between producers and consumers. The removal of riparian vegetation alters the contribution these tributaries have as sinks and would allow greater nutrient contribution to receiving waters (Naiman and Decamps 1997). These findings demonstrate the important influence riparian vegetation has on nutrient availability and transport, and reflects responses measured in the upper Manning (Fig. 6.1).

The impacts of increased nutrients from land use change on aquatic biota are well-documented, and include reductions in biodiversity through the loss of sensitive species, instream habitat changes through the smothering of substrates and increase in periphyton growth and severe diurnal fluctuations in DO and pH (Resh et al. 1988, Dodds and Welch 2000, Buss et al. 2002, Wang et al. 2007). These impacts are likely to be exacerbated under low flow conditions (Rolls et al. 2012). The regionally-derived nutrient thresholds can be used to guide management, indicating the overall nutrient status of the catchment, possible impacts on dependent taxa and highlighting possible nutrient hotspots. By changing land management approaches, and thereby addressing issues such as nutrient hotspots to improve water quality, the aquatic community's ability to resist and recover from impacts such as climate change will increase.

This is supported by work undertaken by Dodds et al. (2010), who found that the resilience (the ability of a system to recovery from disturbance), and resistance (the capacity of a system to remain unchanged from disturbance) of an aquatic community is dependent on the degree of modification that has occurred prior to the disturbance. In the case of regulated river systems, Cortez et al. (2012) found a gradient of recovery existed, with heavily modified river systems showing no signs of recovery, whereas less modified systems showed varying degrees of recovery dependent on the extent of modification. This indicates that systems such as the Manning, with limited regulation, may have a greater capacity to for the ecological community to resist and recover from short-term droughts and flood pulses than more regulated systems.

6.4 Cease-to-pump limits

This study was initiated during the Millennium Drought which commenced in 2001 and ceased in most areas of NSW in late 2009. The drought was considered the worst since European settlement (Murphy and Timbal 2007). While the drought was not as severe or prolonged on the coastal fringe as experienced elsewhere in NSW, mean annual discharge was greatly reduced in the Manning River from 2002 to 2007 compared with the long-term record (Fig. 6.3). This extended period of lower than average flows had an observable impact on water quality (see Chapter 3). These impacts included an increase in macrophyte abundance and epiphytic algal biomass in shallow pool areas, the perseverance of thermal stratification that facilitated the release of nutrients from deep pool sediments into the hypolimnion and a build-up of large amounts of filamentous algae and biofilms in riffle areas due to the lack of scouring.

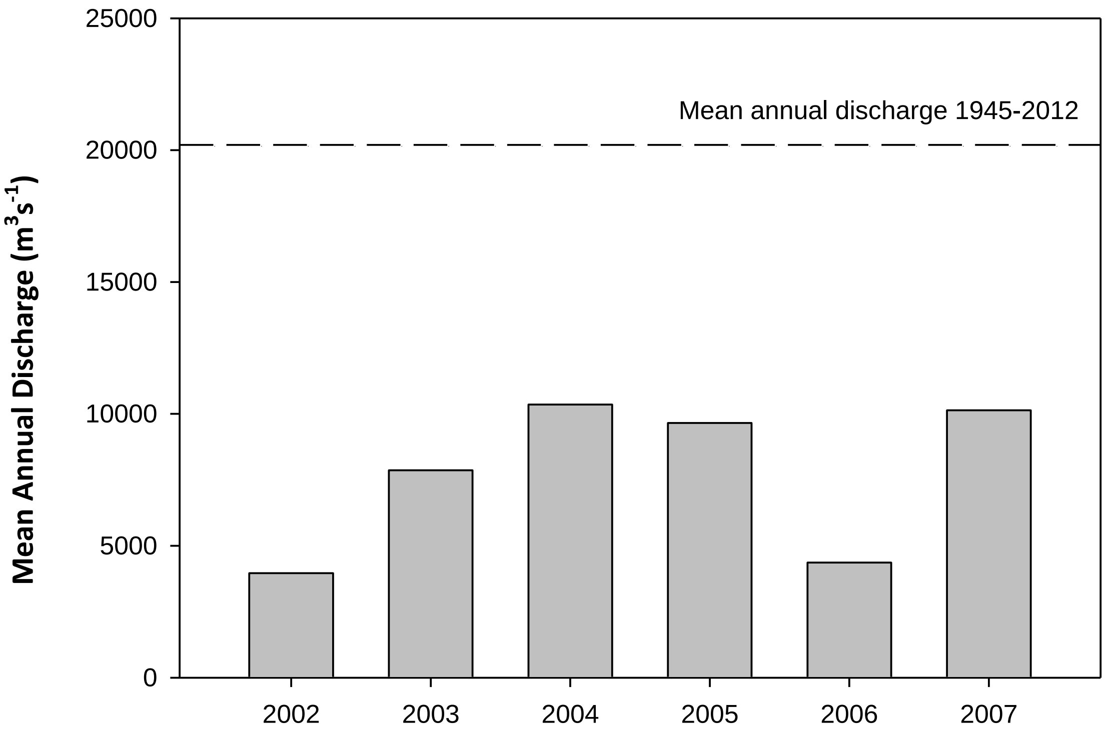


Figure 6.3 Manning River mean annual discharge at Killawarra, 2002-2007
Discharge data accessed from NSW Water Information website (www.waterinfo.nsw.gov.au)

As population increases, water resources across Australia, particularly on the coastal fringes of NSW face enormous pressure. Coupled with the increased likelihood of extreme weather events under climate change, the development of guidelines to assist in the ecologically sustainable use of coastal water resources is essential to protect aquatic ecosystems and water quality. Though drought is a relatively frequent occurrence in Australia, management strategies capable of dealing with drought-related impacts on water resources have failed to be implemented, despite current understanding that anthropogenic modifications generally increase drought impacts (Bond et al. 2008). By measuring the response of biogeochemical processes to changes in flow and developing an understanding of its influence on aquatic communities under low flows, insights can be provided into catchment influences on system resilience and recovery to drought impacts. The extent of this interaction of ecological response with land use and low flows varies with physical characteristics such as channel geomorphology (e.g. deep pools) and biological features such as macrophyte beds.

The implementation of water restrictions, persistence of instream and water storage algal blooms, and long-range forecasts predicting more extreme weather events including increases in evaporation, all raise concerns for water supply security for the catchment. While drought severely impacts aquatic ecosystems, the effects are exacerbated by anthropogenic influences on catchments and waterways that have reduced the ability of aquatic ecosystems to resist disturbance (Suren et al. 2003, Bond et al. 2008, Durance and Ormerod 2009). Despite the frequency of droughts, the consequences of long-term inter-annual droughts as an ecosystem disturbance are poorly understood (Bond et al. 2008).

Climate change is emerging as one of the most significant issues facing biodiversity conservation and, while there are attempts to predict the likely impact of future climate change on freshwater ecosystems, this task is made more difficult due to the many large-scale climate drivers that are likely to be masked by 'feedback' mechanisms (Bradley and Ormerod 2001, Clarke 2009). However, current predictions for south-east Australia highlight the increased likelihood of extreme events which will require resilient and resistant ecosystems to adapt to such dramatic changes (Hughes 2003).

Northern Hemisphere studies investigating the impact of increased water temperature and reduced flows under climate change scenarios suggest that spring macroinvertebrate abundance may decline by over 20% for every 1°C increase, mainly impacting on rarer species (Durance and Ormerod 2007). In Australia, reduction in flows and increased water temperatures under climate change are predicted to result in a loss of migratory thresholds for native fish, reduced DO and increased benthic respiration (Bunn et al. 1999, Davies 2010). However, studies have also found that improving water quality reduced the impacts of warming and that this implied positive management can minimise some climate-change impacts (Murdoch et al. 2000, Durance and Ormerod 2009). The reduction of catchment pressures such as eutrophication, habitat modifications and pest species is thought to improve ecosystem resilience (Clarke 2009). Any of these positive management actions may be hindered by increased pressures on hydrology through greater water extraction under low flow conditions and declining water quality as a result of greater pollutant loads from more intense storms, as proposed by current climate change thinking (Kundzewicz and Krysanova 2010).

Incorporating hydrological extremes into the management and restoration of ecosystems and landscapes ensures that strategies compromised by drought and other extremes are identified and improved strategies are put in place to address impacts over the longer term (Bond et al. 2008). In the Manning these strategies may include maintaining refuge habitats such as deep pools, providing improved cease-to-pump rules, riparian protection and aquatic population management.

Government-led water reform has been underway in Australia since 1994, including the development of the NSW Water Management Act in 2000. These reforms were designed to address the increasing degradation of aquatic ecological values through changes in natural flow regimes (Kingsford 2000). In response to these reforms, several science programmes were put in place to test responses to environmental flows and water regime changes to better inform water management (Driver et al. 2013). The recognition that flow alteration threatens biodiversity and ecosystem function of rivers and the implementation of programmes that attempt to increase understanding of these systems, has led to growing interest in the development of quantitative understanding of aquatic ecosystem response to various types of flow alteration (Poff and Zimmerman 2010).

As flow has an over-riding influence on riverine ecosystems (Poff and Ward 1989, Walker et al. 1995, Puckridge et al. 1998), implementing environmental flows is a key measure for protecting and restoring river ecosystems (Poff et al. 1997, Arthington et al. 2010, Belmar and Velasco 2011). Constructing an environmental flow regime based on a known allocation or cease-to-pump threshold assumes that there are key components of the natural flow regime that will maintain or restore river condition and that there is a sound understanding of the flow requirements of ecosystem components. Unfortunately, this is not true for most Australian river systems, resulting in a need to improve the understanding of flow-driven geomorphological and ecological processes, and to improve the methods available to assess change in condition in response to changes in flow (Ryder et al. 2008).

Engineering or modelling considerations are frequently made before ecological science in the realm of water management (Richter et al. 2006). To be useful in a variety of water management contexts, a process for integrating scientific input must be adaptable to a variety of applications ranging from water supply planning to water extraction and must be practical under a broad range of resource availability (Richter et al. 2006). However, given the inherent complexities of ecosystem responses to variable flow regimes, water managers and other stakeholders should not expect scientists to be right about all environmental flow needs at all times (Richter et al. 2006).

To complicate matters, the ability of scientists to better inform water management decisions is hindered by the possibility that misleading information may result in system degradation or human water-use may be unnecessarily limited (Richter et al. 2006). Therefore the process of flow restoration needs to be iterative and adaptive, with water management actions viewed as experiments that require monitoring and evaluation (Richter et al. 2006). A more recent framework using the ecological limits of hydrologic alteration (ELOHA) is based on the premise that flow regime determines the structure and function of aquatic systems and this information can inform environmental flow standards at a regional scale (Poff et al. 2010). This approach, similar to Richter et al. (2006), focusses on the need for adaptability and the requirement of testing responses to environmental flows. The framework also highlights the need for stakeholders and decision-makers to evaluate acceptable risk and reach a balance between ecological goals, costs and uncertainties (Poff et al. 2010).

In the case of the Manning River, only the protection of low flows in the river has been addressed using limited hydrological and ecological information (Bishop 2006). The adequacy of setting of a cease-to-pump threshold equivalent to the 97th percentile on a rising river, or 98th percentile on a falling river in protecting aquatic communities is yet to be determined. It is apparent from this study that both water quality and benthic macroinvertebrate communities in the lower Manning may be adversely affected when flows reach 90th percentile levels under summer conditions. This indicates that there may need to be a seasonal approach to developing appropriate extraction thresholds for the protection of coastal systems.

Minimum flows represent the least amount of water required to protect the needs of aquatic biota (Silk et al. 2000, Neubauer et al. 2008). However, this single approach does not account for other critical components of flow regimes such as magnitude, return interval, duration, timing and rate of change (Poff et al. 1997, Bunn and Arthington 2002). Metrics encapsulating flow magnitude and temporal variability including mean annual flow, inter-annual coefficient of variation and duration of droughts may better reflect the hydrological and climatic characteristics of a region (Belmar and Velasco 2011). Such approaches may provide a basis for a better understanding of flow alteration-ecological response relationship which is a critical step in assessing environmental flow requirements (Belmar and Velasco 2011).

Water resource agencies need to better collaborate with scientists to base hydrological analyses on the biologically significant facets of flow regime, such as pulse timing, variability of flow magnitude and predictability of annual flows (Puckridge et al. 1998). By reinstating the natural hydrograph, ecosystem processes can be better protected (Bayly 1991, Puckridge et al. 1998).

Furthermore, adaptation to climate change will require a combination of scientific, operational, planning and policy tools. These adaptations will need to occur at a range of spatial scales, from asset level to catchment or basin scale (Saintilan et al. 2013). Scientific tools may include hydrodynamic modelling to allow pre-testing of management interventions or to improve understanding of water requirements. Planning tools, such as water sharing plans, need to incorporate predicted climate change impacts, so these impacts are considered. However, without understanding how systems respond to flows and water quality, it will be difficult to predict changes in biodiversity under conditions climate change is likely to bring. By looking at the current state of the Manning and how current management may be altered to reduce impacts from adverse climate conditions, a proactive approach to the challenges faced in the future can be taken.

6.5 Conclusion

The increasing pressure on water resources through population growth and likely unfavourable climatic conditions, and the accompanying water sector reforms, has introduced a new paradigm for water management in Australia. This has resulted in increased freshwater conservation planning and environmental flow assessments (Nel et al. 2011). The integration of freshwater conservation planning and environmental flow assessment alone will not achieve sustainable development goals without being included within integrated water resources management and supported by water governance and adaptive management.

One component of achieving sustainable and equitable natural resource management is through incorporating relevant specialist information that addresses issues identified by the community and managers, filling knowledge gaps, sharing knowledge develop management strategies and adapting management as knowledge increases (Hillman et al. 2005). While differences between disciplines may act as barriers to integration, the integration of multiple criteria assessments to develop management actions and the identification of knowledge gaps is essential to the successful use of environmental flows (Hillman et al. 2005).

Another important component in achieving improved natural resource management is the need for long-term information. Long-term water quality trends indicated that nutrient concentrations in the lower Manning River are highly variable and are likely to be a reflection of differences in discharge, runoff patterns and land use. Implications of high nutrients include declining river health and a shift from a diverse ecological community to a system dominated by organisms tolerant to organic pollution (Suren et al. 2003, Wang et al. 2007). This shift may impact on the efficiency with which nutrients are processed and may reduce opportunities in which water can be pumped from the lower Manning by MCW to Bootawa Dam for town water supply. This is particularly true under high-flow conditions, when not only runoff is contributing higher nutrient loads, but the breakdown of thermal stratification in upstream pools may also contribute higher levels of inorganic nutrients. Other consequences could be an increase in nuisance plant and algal growth followed by subsequent decay, particularly under low summer flows, adding further stress to aquatic communities.

Land use was found to influence water quality in Australian dryland rivers over the longer term, while flow influenced water quality in the shorter term (Sheldon and Fellows 2010). This finding demonstrates the importance of long-term perspectives and the influence of land use on ecosystem condition. The influence of land use is emphasised by the high climatic and hydrologic variability that exists in Australian systems (Kingsford 2000, Sheldon et al. 2000, Sheldon and Fellows 2010). However, as a result of short funding cycles, very little long-term ecological data exists for Australia's highly variable aquatic systems (Reid and Ogden 2006).

As part of this study I attempted to integrate the effects of land use on aquatic responses to low flows. While there is some understanding of how biota of lotic systems respond to low flows at a broad scale, there is still limited knowledge of how ecosystem processes such as nutrient cycling may change with drought and if and how the system recovers from prolonged drought. Further research is required to improve current understanding of the capacity of NSW coastal rivers to resist and recover from disturbances such as low flows. Research to inform improved management includes investigations into the possible impacts of nutrient-rich water from pool areas to downstream aquatic communities and quantification of the influence of water quality on the Manning River under varying flow and seasonal conditions, and how this influences resistance and recovery from disturbances.

By increasing understanding of what critical low flow levels result in adverse consequences for ecosystem processes and functions and recognising that factors other than flow influences the resilience of a system to impacts, improved management in catchment and flow management can result.

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