Chapter 1 General introduction

Stream ecosystems are greatly influenced by their catchments (Hynes 1975). Along with the obvious important contribution of water, the other great influence from a catchment is nutrients (Johnes et al. 1996, Moss 1998, de Crispen de Billy et al. 2000). While nutrients are an essential component in driving biological stream functions and processes, the continuing impact of changing land use has altered nutrient loads within many systems through diffuse inputs (McKee et al. 2000). Nutrient loads in most aquatic systems around the world are increasing and this has been reflected by impaired water quality and aquatic ecosystem function (Lake and Marchant 1990, Barmuta et al. 1992, Davis and Koop 2006).

The trend of increasing nutrient concentrations in surface waters frequently causes eutrophication of rivers, lakes and estuaries (Boesch 2002). Eutrophication significantly threatens the health of coastal catchments in Australia (Taylor et al. 2004, Lee et al. 2006, Likens et al. 2009), as it affects ecological structure by changing species composition, rates of primary production and trophic dynamics (Turner et al. 1999, Dodds 2006). As nutrients are processed through a variety of biotic and abiotic pathways through aquatic systems, understanding processes and reactions between physical, chemical, geological and biological components (known as biogeochemistry) is an essential step in assessing and managing increasing nutrients.

Stream biogeochemistry relies on an understanding of how elements are altered along hydrologic pathways (Likens 1984). Flow paths, which connect the catchment to streams and rivers are complex and important (Fisher et al. 2004). The influence of key nutrients and materials on ecological processes is dependent on antecedent conditions, changing flow paths and the balance of point and diffuse sources (Bowes et al. 2010, Jarvie et al. 2008). As a result of the importance of flow paths, major changes in how water moves from land to water will affect the strength, timing and type of connections that transport nutrients and other materials to streams (Stanley et al. 2012). Agricultural activities have altered how flows connect land to water usually resulting in large increases in discharge as a result of reduced terrestrial evapotranspiration (Scanlon et al. 2007).

Catchment flow paths are not the only influence on elemental transport and behaviour in streams. Significant hydrologic exchange within aquatic systems occurs between surface and subsurface water, which is thought to have a strong effect on nutrient dynamics and transport (Duff and Triska 2000, Hendricks and White 2000). The contact between water and the chemically and biologically active surfaces found in hyporheic zones create varying oxidation-reduction states (Findlay 1995, Valett et al. 1997, Thomas et al. 2003). Processes such as oxidation and reduction reactions (redox) influence the form and abundance of dissolved constituents which significantly impact on the processes occurring within stream benthos (Hendricks and White 1991, Mulholland et al. 1997, Thomas et al. 2003).

Interactions among nutrients, sediment and biota in surface and sub-surface waters are governed by factors such as sediment properties, discharge, extent of contact between water and sediment, stream bed geomorphology, redox environment and plant uptake (Findlay 1995, Hendricks and White 2000). Numerous processes such as chemical sorption and desorption, erosion and deposition, biological cycling, and biochemical transformation, occurring at the

sediment-water interface influence nutrient forms (Meyer et al. 1988). Conditions at these interfaces, such as dissolved oxygen concentrations, play an important part in determining nutrient speciation through processes such as denitrification/nitrification and Fe(III) reduction/Fe(II) oxidation (Baker et al. 2000). These factors result in sediments and the associated resident benthic communities being one of the most reactive parts of the aquatic environment for regulating changes in processes in the water column (Viaroli et al. 2004). The capacity of benthos to store and renew organic matter influences nutrient fluxes and the balance between oxic and anoxic processes (Santschi et al. 1990, Viaroli et al. 2004). Benthic uptake capabilities are dependent on interactions with biological components, water temperature and flow (Anderson et al. 2002) (Fig. 1.1).

The interactions between surface and sub-surface water that influence biological patterns in streams at catchment scales are relatively well-understood (Pringle and Triska 2000). The influence of nutrient flux on sensitive benthic organisms reliant on the sediment-water interface, such as biofilms, periphyton and macrophytes, is less well known for flowing waters at smaller spatial and temporal scales (Hendricks and White 2000). However, it is known that macrophytes play a major role in influencing flow, sediment dynamics and nutrient cycling (Clarke 2002), and have been found to influence nitrogen cycling through controlling sediment-water exchange rates (Carpenter and Lodge 1986). The decomposition of aquatic plants can govern nutrient cycling and energy flow in aquatic ecosystems as both organic and inorganic components are released during macrophyte decay (Carpenter and Adams 1979). Shilla et al. (2006) found that during macrophyte decay, higher nutrient concentrations were found in the water than during growth phases. These internal processes are important components of nutrient cycling within aquatic ecosystems (Fig. 1.1).

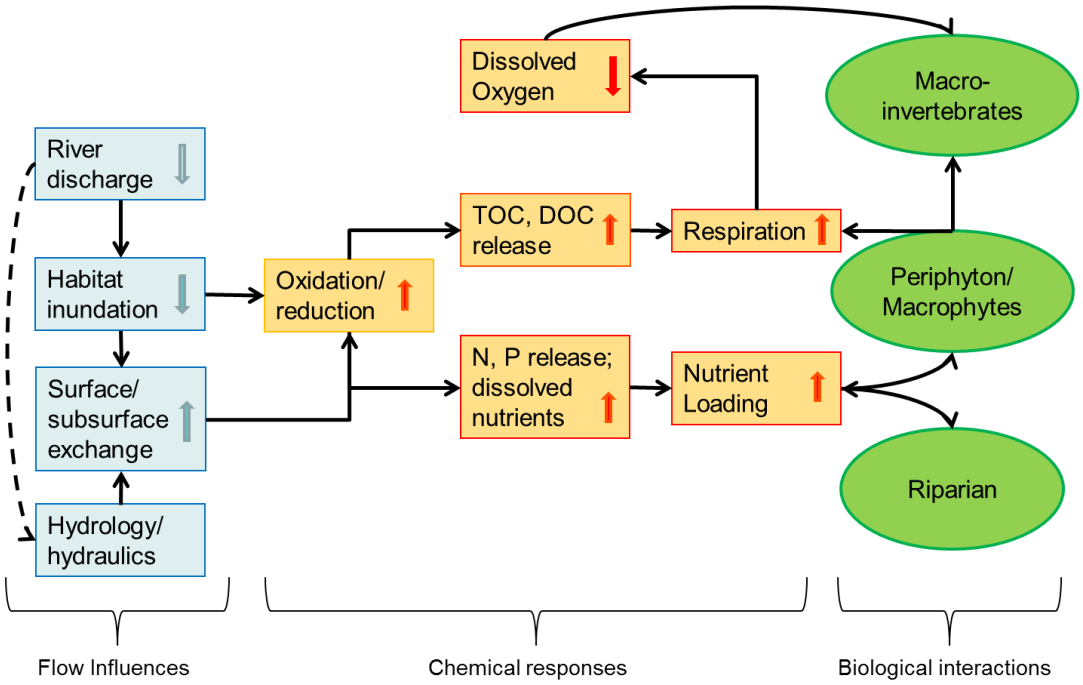


Figure 1.1 Conceptual model of stream biogeochemical processes and interactions with stream biota under low flows (Arrows indicate direction of interactions – i.e. net increase or decrease; blue rectangles indicate flow-related components; orange rectangles indicate chemical components; green ovals represent biological components)

Riparian zones are another essential compartment in which nutrient cycling occurs. These zones provide buffers, protecting waterways from non-point sources of pollution (Thorp and Delong 2002, Woodward et al. 2009, Hadwen et al. 2010a). Nutrient concentrations in most streams are often closely related to runoff events, with the inflow of suspended sediment and terrestrial organic material increasing during first flushes (Deletic 1998, Lee and Bang 2000). The magnitude of this runoff impact is directly related to the extent of the adjacent riparian zone (Baker et al. 2006, Mayer et al. 2007).

The biogeochemical properties prevailing in riparian areas dictate the flux of water, nutrients and other exogenous substances to streams (Vought et al. 1994). Biological uptake and denitrification are two of the main processes that occur in riparian zones known to provide a sink for nitrate (Hill et al. 2000, Gold et al. 2001, Hefting et al. 2005). Nitrogen taken up by riparian vegetation, aquatic plants and microbes returns to the system through decomposition, while microbial denitrification results in the permanent removal of nitrate through the conversion of nitrate to nitrous oxide or di-nitrogen gases (Woodward et al. 2009) (Figure 1.1).

Recent studies have indicated that the contribution of organic matter by riparian vegetation and algae in Australian streams varies greatly both spatially and temporally (Gawne et al. 2007, Leigh et al. 2010). Leigh et al. (2010) found that food webs within floodplain rivers that were characterised by variability in the sources of organic matter showed greater resilience against disturbances. In other systems, benthic algal sources of carbon represent the dominant contribution of carbon to all consumer organisms (Douglas et al. 2005). The importance of autotrophic carbon and, in particular algal carbon, in driving productivity is influenced by factors such as temperature, light and the variability of flow (Finlay et al. 1999, Finlay 2003). The implications of this for Australian rivers and how these elements are processed relate to the functioning of heterotrophic microbial communities and metazoan food webs (Hadwen et al. 2010b).

Understanding how and when nutrients are transported within rivers helps to determine impacts upon, and influence of, aquatic plants and other aquatic species (Selig and Schlungbaum 2002). The nutrient spiralling concept infers that upstream processes will affect downstream communities by altering the form and concentration of nutrients and organic matter during transport (Webster and Patten 1979, Newbold et al. 1982ab, Newbold et al. 1983, Elwood et al. 1983). The spiralling conceptual model describes how a nutrient atom within the water column is transported downstream in dissolved form and is later removed from the water column by biotic processes (e.g., uptake by periphyton). The nutrient spiral is completed when the nutrient atom returns to the water column after release from the biotic compartment (Webster and Patten 1979, Newbold et al. 1983).

Fisher et al. (1998) added to previous nutrient spiralling concepts by introducing the telescoping ecosystem model. This model uses the analogy of the stream-corridor ecosystem consisting of several nested cylindrical elements that extend and retract, similarly to a telescope, in response to disturbance regime. Stream ecosystems are described as several subsystems that consist of vertically and laterally positioned saturated sediments, with the outmost element being the riparian zone. As these zones are hydrologically connected, water and its associated nutrient load move through all of these subsystems as it flows downstream. In any given subsystem, chemical

transformations result in a change in the quantity of materials in transport. If the length of subsystem required to process a given amount of transported material is long, this reflects low rates of cycling which may be the result of disturbance. The telescoping ecosystem model can improve understanding of material retention or nutrient filtering capacity in hydrologically dynamic running water systems.

The river wave concept, developed by Humphries et al. (2014), proposes that river flow can be described as a series of waves varying in shape, amplitude and frequency, traveling both longitudinally and laterally. The position on the wave determines the source of organic matter and the storage, transformation, and transport of material and energy within the river and across the associated floodplain. Local autochthonous and allochthonous inputs vary on the ascending or descending limbs of waves, with upstream allochthonous inputs and longitudinal transport of material and energy dominating in troughs. Conversely, allochthonous inputs of material and energy and autochthonous production from the floodplain increase on the wave crest (Humphries et al. 2014).

Hydrologic pathways also influence macroinvertebrate distribution, which in turn can also influence nutrient pathways through altering decomposition rates and nutrient cycling rates (Fig. 1.1). Hydrology of a riverine system is determined by climate and catchment characteristics, with the intrinsic hydrological variability influencing stream habitats and ecological processes (Poff and Ward 1989, Puckridge et al. 1998, Lake 2000). Flow variability is considered to underpin river ecosystem function through variations in magnitude, frequency, timing duration and rates of change (Puckridge et al. 1998, Bunn and Arthington 2002). Flow increases provide overbank pulses that transport energy to floodplains and increase connectivity to facilitate the dispersal of water-dependent species (Sheldon et al. 2002). Low flows provide a mechanism for deposition and drying phases to occur (McMahon and Finlayson 2003).

Stream hydraulics, as influenced by channel morphology and hydrology, is also considered a major determinant of stream community organisation (Davis and Barmuta 1989, Brooks et al. 2005). Stream biota, such as macroinvertebrates, are significantly influenced by factors such as velocity, water depth and substrate type (Brooks et al. 2005). Changes in these factors will influence the number and type of taxa found in affected areas as hydraulic variables influence the availability of in-stream habitat, with decreasing habitat resulting from a loss of wetted area (Brasher 2003, Dewson et al. 2007a).

Investigations that have used specific relationships between flow velocity and discharge to assess the impact of flow, particularly reduction, on macroinvertebrate fauna have found that species preferences for various flows could be used to develop minimum flow requirements that would mitigate the effects of flow reduction (Gore 1978, Brunke et al. 2001). While some work has been performed 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 (Growns 1998).

More recently a framework, known as the ecological limits of hydrologic alteration (ELOHA), has been used to assist in the development and implementation of environmental flow standards for rivers at a regional scale (Poff et al 2010). This framework is based on existing hydrologic techniques and environmental flow methods to support flow management at a regional scale and

utilises available scientific information to develop ecologically based and socially acceptable goals and standards for the improved management of environmental flows (Poff et al. 2010).

Macroinvertebrate response to flow decline is thought to vary depending on the timing and persistence of the decline and the characteristics of the affected system. Short-term discharge reductions have resulted in the aggregation of invertebrates in the decreased available habitats of small streams in New Zealand, thereby increasing local invertebrate density (Dewson et al. 2007a). Elsewhere it was found that increasing flow expanded the habitable area within a riffle and therefore the production of lotic invertebrates (Englund and Malmqvist 1996). Other studies found that persistent drought and increased extraction have, in some cases, reduced invertebrate densities and recruitment (Wood and Petts 1994, Boulton 2003).

Changes in water quality and biotic responses within a river should also be taken into account when assessing minimum flow requirements. Periphyton growth, in particular an increase in filamentous algae, can be a significant factor in how flow reduction alters invertebrate community structure (Suren et al. 2003). Populations of more sensitive species such as mayflies and caddisflies declined, while aquatic worms, snails and chironomids tended to dominate (Suren et al. 2003). 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).

Along with water quality, micro-habitat and hydraulic characteristics influence macroinvertebrate community structure in riffle areas, through influencing their metabolism, feeding and behaviour. The distribution of invertebrates correlates with hydraulic variables, and hydraulic conditions provide a major physical gradient along which the benthic community is organized (Evans and Norris 1997, Rempel et al. 2000).

To understand basic functioning of rivers, the source and form of nutrients entering the system needs to be identified. Despite many studies indicating that diffuse sources usually contribute a greater overall load to receiving waters, there has been considerable focus on point-source pollution (Cullen 1991, Jordon et al. 1991). While point sources such as Sewage Treatment Plants have localised impacts, diffuse sources, particularly agriculture, have a more widespread influence. Geology may also be a major phosphorus contributor particularly where basalt is found, with natural phosphorus concentrations higher in catchment areas underlain by soils derived from volcanics, such as basalt (McKee et al 2001). It is therefore essential to establish the source of nutrients to inform on the most appropriate management strategies to reduce nutrients entering the river and appropriate flow regimes to minimise the impact of nutrients already in the system (Davis and Koop 2006).

While reducing nutrient input is a vital component of any management approach, even immediate implementation of appropriate land use changes is unlikely to reduce overall nutrient concentrations within the system for many years. The prolific rates of primary production obvious in many coastal rivers and estuaries in particular, indicate that transported nutrients are not a limiting factor and there is likely to be a large store of nutrients associated with riverine sediment. To best inform long-term planning and management, increased knowledge is required on how

biogeochemical processes are influenced, and respond to both reduced flow due to greater extraction and an increase in nutrients as a result of inappropriate land management.

1.1 Nutrient origins and pathways in riverine systems

Nutrients utilise both terrestrial and aquatic pathways when being transported to and through the aquatic environment (Hart and McKelvie 1986). The investigation of these pathways requires integration of the terrestrial and aquatic disciplines. Though these have developed independently, both disciplines have tended to focus on nutrient limitation, ecosystem nutrient retention and transformations (Grimm et al. 2003). However, investigation of connectivity between the two system types relies on both hydrologic and ecological models, and is required to address environmental problems associated with altered biogeochemical cycles (Grimm et al. 2003).

The interaction between riparian vegetation and instream processes is one such area where the integration of terrestrial and aquatic knowledge is required. The removal of native vegetation and disturbance from cropping and grazing are known to be major impacts on water quality (Johnes and Heathwaite 1996, Harris 2001, Jones et al. 2009, Weijters et al. 2009). It is this predominance of cropping and grazing and low population densities throughout much of Australia that results in nutrients to rivers generally originating from diffuse or non-point sources (Davis and Koop 2006, Carpenter et al. 1998). Non-point source pollution poses a significant problem to managers as it is difficult to measure and regulate (Howarth et al. 2000). It can adversely impact on water quality at a larger spatial scale but generally at lower concentrations than point source pollution, making it difficult to identify and isolate. Although non-point source pollutants generally occur in low concentrations, the environmental impact on freshwater ecosystems can be just as severe over the long-term (Arbuckle and Downing 2001, Abrantes et al. 2008).

While point sources generally contribute less to overall nutrient loads in Australian riverine systems, they may still significantly impact water quality. These impacts may be greatest under low flow conditions, as point sources can contribute significantly to the volume of discharge in receiving streams and rivers (Bukaveckas et al. 2005). Unlike the usually continuous, long-term impacts resulting from non-point source pollution, point source pollution may result in the release of concentrated toxicants with impacted biota demonstrating more acute responses (Wakelin et al. 2008, Brooks et al. 2006).

Catchment geology influences nutrient concentrations of both surface and groundwaters, particularly in areas containing underlying basalt (Valett et al. 1997, Sharma and Subramanian 2008). However, the contribution of the underlying geology to nutrients in water courses is generally much less than that of anthropogenic sources such agriculture (Brodie and Mitchell 2006). The potential of land use impact extends to subsurface components of the water cycle through altering recharge characteristics and water quality impacts due to activities such as effluent application (Ray and Schock 1996). The potential influence of groundwater on surface water chemistry, whether it is negatively through contamination, or positively via dilution effects is also an area of biogeochemical interest.

Physical conditions within river systems influence the transport and transformation of nutrients. Deep river pools can extend nutrient residence in a river system through the accumulation of

sediment or increase nutrient availability by providing anaerobic conditions for nutrient release to the water column (Selig and Schlungbaum 2002) (Fig. 1.2). Factors operating within deep pools that may influence nutrient form and pathways include water turbulence, velocity, depth, presence of stratification, water chemistry and light availability (Fig. 1.2). Sediment composition, particularly particle size, will also influence nutrient adsorption rates (Selig and Schlungbaum 2002). Particles, which accumulate in deep pools and slower flowing sections of rivers are often organically rich, especially in phosphorus, resulting in the highest phosphorus concentrations being located in slow flowing or impounded sections. These areas play an important role in nutrient retention in low flow periods (Selig and Schlungbaum 2002).

Biological factors, such as macrophytes, also influence nutrient and flow behaviour within some rivers. While the distribution and growth of macrophytes are influenced by water turbulence, velocity, depth, temperature, nutrients, sediment composition, light availability and other biota, macrophytes can, in turn, influence these same variables (Clarke 2002). Once macrophytes are established, they can reduce turbidity through increasing sedimentation rates and reduce the localised current velocity, possibly increasing autotrophic response and up-take of nutrients (Petticrew and Kalff 1992).

The extent to which a macrophyte bed can do this will depend on plant life phase, density and structure (Fig. 1.2). The life phase of aquatic plants are known to determine their extent of influence on flow and nutrient pathways as macrophytes go through periods of senescence, rapid growth, maturity and decline. The reduction in macrophyte bed vigour is also related to excessive epiphytic algal growth and seasonal changes. In the case of submerged macrophytes, those that form meadows (i.e. evenly distributed biomass throughout the water column or at the base of the plant) will have a different impact on flow compared to macrophytes that form canopies (i.e. biomass is concentrated near the top of the plant) (Madsen et al. 2001). The influence of macrophyte form on flow will depend on water turbulence and velocity (Madsen et al. 2001).

The identification of factors contributing to nutrient cycling and poor water quality through assessing the influence of macrophyte beds will allow further associations to be made regarding crucial temperatures and flow. Critical flow thresholds that promote the decay of the macrophyte beds should be identified, along with times of greatest impact, with the aim of reducing the frequency and altering the timing of unfavourable flow conditions.

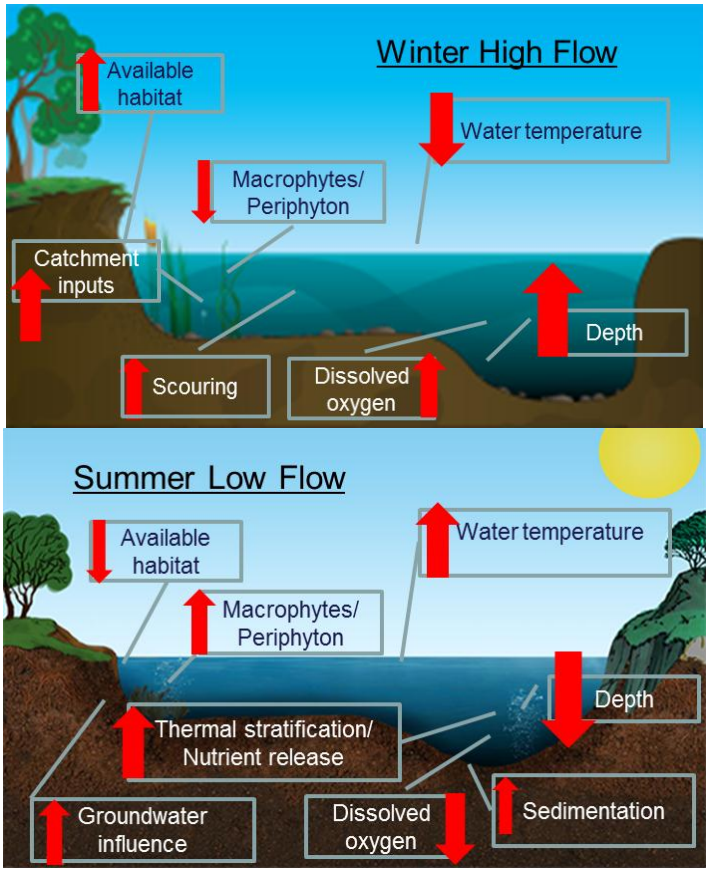


Figure 1.2 Conceptual model of winter high flow, summer low flow influences on the pools of the Manning River, New South Wales (NSW) (Size of arrows indicate level and direction of influence on water quality and aquatic community)

1.2 Short-term temporal changes in nutrient pathways

Natural disturbances vary temporally in terms of frequency, predictability and duration. Such disturbances can impact biota, disrupt ecological processes and redistribute resources by altering the inputs and flux rates of nutrients (Lake 2000). In aquatic systems, short hydrologic residence times can result in replenishment of aquatic systems with oxygen-rich water, while longer residence times can allow for depletion of oxygen by microbial respiration (Helton et al. 2013). Changes in land use and increasing regulation invariably alters hydrologic residence times through modifications to seasonal flows, further disrupting riverine communities (Lake 2000).

Season and flow conditions are important determining factors of the nature and strength of the relationship between land use and in-stream nutrient concentrations (Yates et al. 2014). Streamwater nutrient concentrations have been shown to vary seasonally, with highest concentrations in summer and lowest in late autumn and winter (Mulholland 1992, Mulholland and Hill 1997). Mulholland and Hill (1997) concluded that nutrient concentrations were higher in summer as a greater proportion of stream discharge was generated by the bedrock zone during periods of lower flow. The increased influence of sub-surface flows and reduced within-channel connectivity under summer low flows results in the greatest influence on nutrient concentrations being confined to the immediate area (Yates et al. 2014).

Studies looking at thermal stratification, and impacts of changes in temperature and dissolved oxygen have mainly focused on large storages or lakes (Szumiec 1985, Boland and Padovan 2002, Wilhelm and Adrian 2008), with very few studies looking at stratification in rivers (Western et al. 1996, Turner and Erskine 2005, Becker et al. 2010). When looking at stratification within deep pools in rivers, Turner and Erskine (2005) found that pools with a depth greater than 5m stratified under low flow conditions resulting in anoxia in the bottom layers. This lack of oxygen resulted in reducing conditions which contributed to an increase in phosphorus and a general decline in water quality. However, diel changes were not discussed as part of this study.

The few stratification studies that have assessed diel changes within depth profiles (Frempong 1983, Talling 2004, Cavalcante et al. 2007) have found that diel cycles are largely repetitive and usually relate to weather episodes. These cycles may also extend seasonal stratification by vertical heat transfer through the water column (Talling 2004). Along with changes in temperature through the depth profile, there may also an associated change in other variables such as dissolved oxygen, pH and conductivity (Cavalcante et al. 2007). These differences often result in changes in nutrient concentrations over a short temporal period (Cavalcante et al. 2007).

Prolific plant and algal growth, particularly under warm, low flow conditions, can result in nutrient increases at certain times through sediment release and decomposition (Fig. 1.2). Changes in pH, dissolved oxygen and redox potential resulting from photosynthesis and respiration influences are likely to contribute to altering the nutrient speciation of freshwater systems. While studies have investigated the interactions between macrophytes and water quality (Grimm and Petrone 1997, Park et al. 2003), particularly the changes in diel oxygen cycles associated with plant respiration and photosynthesis and the influences on chemical transformations (Brick and Moore 1996), there has been relatively little work on how short-term variations in oxygen concentrations and its effect on chemical transport through streams (Harrison et al. 2005).

Land use and hydrologic changes frequently alter the sources and relative importance of different types of organic matter in-stream, with higher nutrient concentrations increasing the amount of primary production in the channel resulting in a reduction in available habitat and periodic depletion of oxygen (Wilcock et al. 1995, Young and Huryn 1999). Vink et al. (2005) found that these hydrologic changes, particularly flow regulation and climatic effects, impact on carbon supply and biogeochemical cycling in Australian riverine systems. These impacts include changes in aquatic thermal regimes influencing rates of microbially mediated biogeochemical transformations, increased occurrence of dissolved oxygen depletion and reduced floodplain inundation, resulting in reduced carbon inputs (Talling and Lemoalle 1998, Neill et al. 2006).

Diel changes are frequently used to measure stream metabolism and the influences of land use and hydrology (Marzolf et al. 1994, Bott et al. 1997, Young and Huryn 1999). Research has inferred a link between diel fluctuations in oxygen and denitrification rates in a rivers (Venterink et al. 2003), especially those supporting macrophytes and sediments heavily colonized by benthic microalgae (Christensen et al. 1990, Nielsen et al. 1990). The link between diel patterns in oxygen consumption to patterns of nitrate and nitrite production was also shown by Laursen and Seitzinger (2004). However, few studies have examined the role or implications of night-onset anoxia for nitrogen cycling and transport in streams or rivers. These rapid in-stream fluctuations

can affect a broad spectrum of biologically important nutrients and other factors such as alkalinity and conductivity over very short timescales (Harrison et al. 2005, Wilcock and Chapra 2005).

Despite the potential for day-night shifts in oxygen conditions to alter stream chemistry, many models of river biogeochemistry ignore diel changes in stream chemical dynamics (Harrison et al. 2005). These models incorporate the ability of rivers to transform nutrients (Fisher et al. 1998, Wollheim et al. 1999) rather than treating them as simple conduits, but generally ignore short-term environmental variations despite the fact that diel oxygen fluctuations are relatively common in biologically productive rivers and streams (Harrison et al. 2005). This lack of consideration of short-term variability within these models is a factor of the ecosystem-level approach taken.

While low flows have a substantial influence on instream nutrient behaviour through extending residence time, disconnection and facilitating stratification, overbank flows support ecologically critical exchanges between sediments. Overbank flows allow nutrients and detritus from sediments to become available to biota, facilitating vital exchanges between the floodplain and water column (Boulton and Brock 1999). Floods and spates also scour aquatic environments, resetting biofilms and modifying geomorphic features to alter habitats (Sheldon and Walker 1997, Ryder 2004). The scouring of substrates results in a high turnover of biofilms contributing to productivity and energy flow through aquatic systems (Shaver et al. 1997, Ryder et al. 2004).

High flow influences are altered through changes in land use, particularly riparian vegetation, while geomorphic factors such as light availability and carbon supply, are fundamental factors controlling organic matter production, respiration, and transport in streams (Young and Huryn 1999). These terrestrial influences have significant impacts on stream metabolism and can create allochthonous or autochthonous dominated systems (Young and Huryn 1999). The influence of short temporal changes on biogeochemistry and the response to these changes by primary producers influences how nutrients are transported and used in-stream.

1.3 Nutrient behaviour in lotic and lentic environments

The form of nutrient transfer varies depending on river discharge and biophysical features. Flow has a significant effect on ecosystem functioning and characteristics of flow such as hydrologic residence time determines the extent to which the chemistry of flowing water is altered (Grimm et al. 2003). Other features likely to influence nutrient uptake and release include stratification, primary producers and consumers. The way in which these components influence nutrient form and transport are discussed below.

1.3.1 Thermal stratification

The alteration of water temperature within pools and, as a consequence, the modification of biogeochemical cycles, such as changing oxygen budget, cycling of organic carbon and nutrient balances is of particular concern for water quality and ecological responses (Elliott 2000, Becker et al. 2010). Thermal heterogeneity in the water column is often complex and controlled by factors such as shading, groundwater, inflows and hyporheic exchange (Danehy et al. 2005, Moore et al. 2005, Webb et al. 2008). The modelled effects of thermal stratification on oxygen concentrations within weirs and locks in a large German river, showed that the high temporal and spatial variability of oxygen concentrations, as result of increasing algal populations and water

temperatures at the pool surface, was of major ecological significance (Becker et al. 2010), and this can result in the loss of aquatic refugia during low flows (Elliott 2000).

Persistent thermal stratification is also a common phenomenon within Australian weirs, and is often a precursor to the development of extensive algal blooms concentrated in the epilimnion (Sherman et al. 1998, Webster et al. 2000, Mitrovic et al. 2006). Deeper pools stratify both thermally and chemically, particularly in summer and under low flows (Turner and Erskine 2005). Stratification tends to occur when flows decrease over the summer period as mixing is reduced and surface waters become warmer than the water underneath (Fig. 1.2). The resulting stratification causes water quality problems including hypolimnetic deoxygenation and increases in ammonia, hydrogen sulphide, iron, manganese and phosphorus from the reducing conditions (Petts 1986). Despite acceptance that thermal stratification occurs under low flows in the deeper sections and impoundments of some rivers, it is generally ignored in monitoring and modelling programs (Erskine and Turner 2005).

Most studies looking at nutrient and trace metal release from sediment have been conducted in lakes (Mortimer 1942, Jenson and Andersen 1992, Driscoll et al. 1993), and while longitudinal transport cannot be examined for these systems, there are similarities in processes between lakes and river pools. Release from sediment within lakes can be a major fraction of the total phosphorus load during summer (Andersen 1982), as a result of raised water temperatures increasing release of organically bound phosphorus into surrounding sediment pore water (Kamp-Nielsen 1975). Other factors influencing the supply of phosphorus from sediments include redox conditions, adsorption capacity of sediments, mineralisation of organic matter and turbulence (Driscoll et al. 1993).

The capacity of deep river pools to store and release nutrients is determined by factors such as location of the pool on the river, residence time, changes in temperature, dissolved oxygen levels or changes to pH. Therefore, several questions arise concerning the influence of pools on nutrients in rivers. If nutrients are released from deep pool sediments, how do these released nutrients impact on downstream water quality and associated aquatic communities? Equally important is how upstream nutrient sources behave once deposited within the deep pools.

It is thought that, following periods of high flow as water velocities decrease, large amounts of detritus and sediment settle in deep pools (Fig. 1.2). The large influx of decaying material and its subsequent processing may result in hypoxic conditions and adversely affect water quality within the pools, especially if stratification occurs. This impact could be transferred further downstream once stratification breaks down.

1.3.2 Primary production

Aquatic macrophytes play an important role in biogeochemical cycles, particularly surface and subsurface exchange processes (White and Hendricks 2000), influencing nutrient availability and pathways (Madsen et al. 2001, Clarke 2002). Nutrients can take several paths through lotic macrophyte beds, not only affecting the macrophyte bed but also being affected by processes within the bed. Biological processes, changes in velocity and dissolved oxygen may alter also nutrient forms (Fig. 1.2).

Greater aquatic macrophyte production has been found to occur in areas of advective surface water or groundwater movement through bottom sediments, particularly if nutrients are limiting.

The association of increased macrophyte production with advective water movement is related to the subsurface processing of dissolved organic carbon, phosphorus, nitrogen and metals. Microbial activity and plant-microbial interactions at the rhizosphere drive many of these subsurface processes, with the microzones (or plant root biofilms) associated with large lotic plant beds playing a significant role in nutrient dynamics of stream ecosystems (White and Hendricks 2000). Furthermore, a study on sediments taken from inside and outside plant beds found aquatic plants significantly increase phosphorus release from the sediments (Stephen et al. 1997). Macrophytes have also been associated with denitrifying bacteria, which influence nitrogen dynamics within the aquatic environment (Howard-Williams 1985). Conversely, Forshay and Stanley (2005) found that, in the absence of macrophytes and phytoplankton, denitrification on the floodplain was the primary driver for nitrate removal during overbank flooding.

However, it is not only the ability of macrophytes to influence nutrient release and uptake that is important, but the large biomass of epiphytes and bacterial biofilms often associated with macrophyte beds that also provide considerable surface areas for microbial metabolism (Weisner et al. 1994). Not only are macrophytes a major influence on nutrient uptake and release, but the sands and silt found predominantly in the pools also contribute significantly as a store and source of nutrients. Channel cross section and velocity determines whether particulate matter in rivers is transported, deposited or resuspended (Selig and Schlungbaum 2002). The location of pools, runs and riffles establishes where sediments accumulate, where nutrients are released and where aquatic plants are able to take advantage of favourable environments.

Periphyton communities are important in nutrient uptake and cycling in stream ecosystems (Grimm 1987, Triska et al. 1989). High-biomass stream periphyton communities can be nutrient-limited even when nutrient concentrations in stream water are well above those required for periphyton growth (Bothwell 1989). As a result of this, it is assumed that nutrient cycling by stream periphyton communities may be closely linked to stream hydraulics. Past studies on nutrient dynamics in streams have focused on the influence of water velocity on nutrient uptake (Horner and Welch 1981), and particularly on phosphorus uptake (D'Angelo et al. 1991). A study of the influence of differences in streambed hydraulic gradient on periphyton biomass found a strong relationship between these two factors (Valett et al. 1994). Where the frequency of floods are high relative to the rate of biomass production, periphyton assemblages have been found to be dominated by smaller species such as adnate diatoms that have evolved to resist scouring (Cardinale et al. 2006). Where flood frequencies are low relative to biomass productivity, assemblages were dominated by stalked and filamentous algae prone to scour impacts (Cardinale et al. 2006).Mulholland et al. (1994) also demonstrated that accumulations of periphyton biomass can alter the hydraulic characteristics of streams, particularly by increasing transient storage zones and increasing internal nutrient cycling. This indicates the importance of periphyton on nutrient transport and cycling, particularly under low flow conditions (Figs 1.2 and 1.3).

Increasing human population and changes relating to climate change may add to any nutrient and extraction impacts. Given that flows are usually reduced in summer in response to lower rainfall and higher extraction, it is likely that macrophyte beds and periphyton within riffles may become exposed (Figs. 1.2 and 1.3). This exposure, combined with high summer temperatures, would most likely result in very unfavourable conditions for the community structure of periphyton, macrophytes and associated biota.

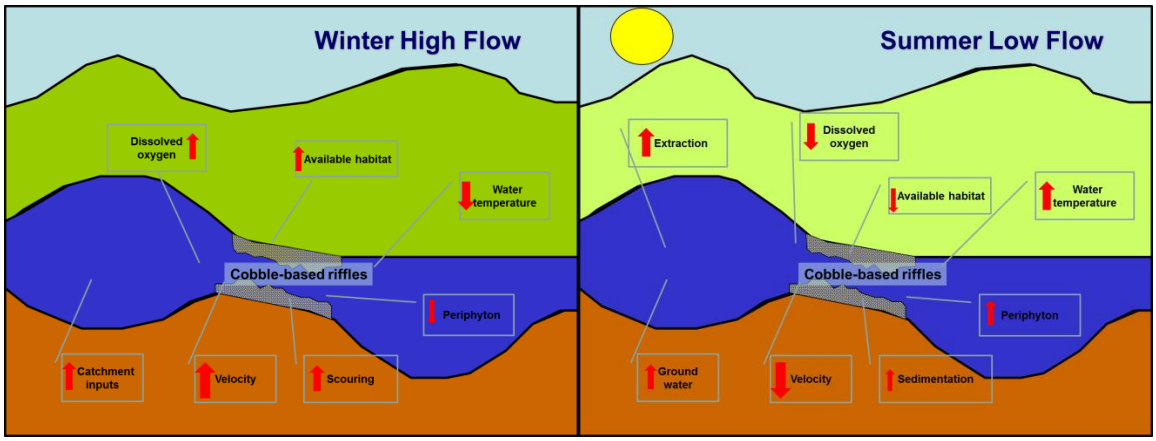


Figure 1.3 Conceptual model of winter high flow, summer low flow influences on riffles of the Manning River, NSW (Size of arrows indicate level and direction of influence on water quality and aquatic community)

1.4 Macroinvertebrates and flow relationships

Closely associated with macrophytes and periphyton are macroinvertebrate fauna. Macroinvertebrates are not only major contributors to biodiversity within riverine systems but are important processors of plant and algal material (Chessman 1986) and food for higher trophic levels such as fish and platypus (Grant and Temple-Smith 2003). For these reasons, macroinvertebrates have long been regarded as useful indicators of ecosystem health (Williams 1980, Lenat 1988, Resh et al. 1995) and may therefore provide further information regarding the state of the system by reflecting the decline in conditions within habitats over time.

Studies have found that periphyton growth, in particular an increase in filamentous algae, can be a significant factor in how flow reduction alters invertebrate community structure (Suren et al. 2003). Populations of more sensitive species such as mayflies and caddisflies declined, while aquatic worms, snails and chironomids tended to dominate (Suren et al. 2003). Seasonal variations within macroinvertebrate populations are also impacted by changes in the natural flow regime. 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 number of studies have indicated that decreases in discharge cause decreased water velocity, water depth, and wetted channel width; increased sedimentation; and changes in thermal regime and water chemistry (Dewson et al. 2007a, Dewson et al. 2007b) (Fig. 1.3). Though macroinvertebrate abundance may vary in response to decreased flow, macroinvertebrate richness usually decreases in response to the loss in habitat. 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 (Brooks et al. 2005). The distribution of invertebrates has been found to significantly

correlate with hydraulic variables, and hydraulic conditions are thought to represent a major physical gradient along which the benthic community is organized (Rempel et al. 2000).

1.5 The Manning River system and resource management issues

From 1981 to 2014, the Australian population has risen from 15 million to over 23 million, increasing pressure on all natural resources but particularly water (ABS 2014). This pressure on water resources is related to both the volume and quality of water available for consumptive use. The NSW coast remains the focus of economic development and growth for the State, with over 80% of the NSW's population living in a local government area adjacent to the coast. The fastest growth rates between June 2001 and June 2006 occurred in the Richmond-Tweed and Mid-North Coast regions, both recording an increase of 1.2% per year (*www.innersydney.org.au).* As population growth increases in NSW coastal catchments, extraction requirements place added pressure on stream ecosystems. This increased pressure raises a number of questions regarding the system functions and the possible localized and downstream impacts of reduced flow and declining water quality in modifying system functions and associated aquatic communities.

As development pressures and climate change impacts increase within NSW coastal zones, a long-term view of water and land management is a crucial component to maintaining and improving the health of the river and its catchment. This long-term view entails the formulation and enforcement of environmentally-sustainable water extractions and changes in the way the whole catchment is managed, particularly in terms of maintaining and/or improving water quality. As a result of these pressures the Manning River was chosen as a case study for assessing the adequacy of water sharing plans and catchment management.

The freshwater systems of the NSW mid-north coast encompass the river valleys of the Macleay, Hastings and Manning Rivers. The Manning Valley, the most southerly of these catchments, is within the sub-tropical climatic zone extending along the north coast of New South Wales. It is characterised by warm temperatures throughout the year with seasonal and daily extremes restricted to western upland sections of the valley. Maximum rainfall occurs during late summer and early winter with lower rainfall occurring during late winter and early spring (Birrell 1987). Rainfall distribution is dominated by the topography, with the highest rainfall occurring along the plateau escarpments on the northern extent of the valley and the ridges. Catchment characteristics for the Manning are detailed in Chapter 2.

Several deep pools are located in the lower Manning River, and their impact on water quality and nutrient cycling is not currently well understood. Processing large quantities of decaying vegetation within deeper pool areas may result in hypoxia, particularly under stratified conditions. If stratification and hypoxia does occur within these deep pools, the flows and temperatures required to induce such conditions need to be investigated. Once more is known regarding the link between hydrodynamics and biogeochemistry of these deep pools, the influence of these deep pools on water quality downstream may be better assessed.

The lower Manning also supports many large shallow pool areas, dominated by dense macrophyte beds containing species of *Myriophyllum*, *Potamogeton*, *Vallisneria* and *Hydrilla* (Thurtell and Bishop 2006). The wide, shallow pools located in the lower Manning have high sun-exposure and therefore considerable potential for heating over summer. These pool areas provide ideal environments for the prolific growth of aquatic plants. The increase in water

temperature under summer low flow conditions appears to result in aquatic plant death and rapid decay, promoting poor water quality (Thurtell and Bishop 2006) (Fig. 1.2). This is of particular concern given that major areas of macrophytes are located upstream of the water supply off-take for major regional centres.

The relationship of low flows and nutrient impacts on riffles areas can be determined through the examination of benthic communities within riffles downstream of the most impacted pools (Fig. 1.3). The examination of hydraulic and hydrologic data in association with changes in the benthic communities would assist in determining critical flow thresholds and how of extraction activities impact on these thresholds.

1.6 Aims and hypotheses

This investigation describes the interaction of flow and temperature on nutrient pathways and the resulting impacts on ecological communities in coastal stream ecosystems. An aim of this thesis is to understand the contribution of nutrients and season in determining the degree to which ecological systems are affected by low flows. These factors need to be taken into account when assessing suitable thresholds for water extraction. This includes determining the importance of maintaining connectivity between pools and safeguarding moderate to high flows that improve water quality and, ultimately, maintain dependent biota.

I hypothesise that catchment properties and agricultural land use increase total nutrient concentrations within the Manning River system. Dissolved nutrient concentrations further increase when median summer flows are reduced, through extraction, to below the 90th percentile of summer flows. These increases in nutrients occur through biogeochemical interactions, influenced by stratification and diurnal processes. Reduced flow, coupled with changes in nutrient concentrations, will have ecological impacts through the increase of primary productivity and biomass. This increase in biomass will alter the chemical and physical environment for dependent species such macroinvertebrates, reducing taxa richness. Furthermore, low flows and increased nutrients will reduce temporal habitat availability and heterogeneity, thereby reducing the ability of the riverine community to resist or recover from disturbance.

1.6.1 Thesis structure

Chapter 2 describes the physical characteristics of the study area. The chapter also details the impacts associated with the catchment and current water management.

Chapter 3 examines the origin and pathways of nutrients in the Manning system to increase understanding of where nutrients originate and how they travel through the system under various flows and temperature conditions. To identify the role of C, N and P as possible limiting factors with respect to periphyton growth, an in situ experiment was undertaken to identify limiting nutrients to autotrophic production. Water chemistry reference values on which to compare overall catchment water quality performance and appropriate threshold values for ecological responses are also assessed spatially and temporally, to further inform catchment and water management within the Manning system.

Chapter 4 investigates nutrient response to flow, water depth, temperature and macrophytes in Manning River pools. The chapter includes flow influences on nutrient behaviour within both shallow and deep pools. An investigation of water chemistry within a deep river pool and extensive macrophyte beds in a large shallow pool was used to ascertain changes on water chemistry and possible biological influences.

Chapter 5 examines ecological responses to flow and temperature through assessing periphyton and macroinvertebrate responses to low flow conditions within riffles and under various nutrient regimes.

Chapter 6 synthesises the results from the previous chapters and examines the management implications of land use and water extraction and how management actions influence ecological responses within the Manning River to low flows.

Chapter 2 Study area

The Manning River (31°53'54"S, 151°28'04"E), is an east-flowing unregulated coastal river on the mid-north coast of NSW in south-eastern Australia (Fig. 2.1). The river rises in the Barrington Tops and flows 250 km to its estuary located in the vicinity of the township of Taree. Along with the township of Taree, other major population centres which rely on the Manning and its associated tributaries include Wingham, Gloucester, Tuncurry and Forster. Major tributaries include the Gloucester River, Nowendoc River, Barrington River and the Barnard River.

The total catchment area of the Manning is approximately 8,190 km^2 which is bound to the north by the Bulga and Comboyne Plateaus and to the west and north-west by the Mount Royal ranges. Rainfall in these upland ranges of around 1,200 mm $year^{-1}$ results in average runoff to the catchment of 2,230,000,000 $m^3 year^{-1}$ (MCW 2006). Approximately 35 per cent of the catchment is cleared for agricultural pursuits and 15 per cent for forestry (Table 2.1). The lower Manning floodplain is extensively modified with almost 40 per cent cleared for grazing (carrying approximately 150,000 head of stock) and cropping, with 1.5 per cent cleared for urban areas. However, there still remain extensive areas of native forest and a variety of vegetative communities including rainforest and dry woodland complexes (Table 2.2).

The rivers and creeks in the Manning valley are unregulated by large structures but small weirs may influence discharge patterns, water quality and fish passage. Within the Manning region water is extracted from the Manning River system for a range of uses that include irrigation and stock water, rural and domestic use and urban reticulated supply. Though it is one of Australia's few large river systems not to be dammed for water supply purposes, water is pumped from the River to Bootawa Dam, an off-river storage for local urban water supply by Mid Coast Water (MCW). The Bootawa storage has a capacity of 2,275,000 m^3 and diverts 24,822,000 m^3 $year^{-1}$ from the lower Manning River. This Dam supplies Taree, Wingham, Forster and Tuncurry. The river system is the sole contributor of water supply to Gloucester and Barrington. Further to this, the Barnard Scheme was developed in 1986 to divert water from the Barnard River to the Hunter River (the adjacent southern catchment) in periods of low flow (ERM 1996). To date this scheme has not been used and water has not been diverted.

The lower Manning River contains many large (>100m^2) shallow (<2m deep) pool areas, dominated by extremely dense emergent and submerged macrophyte beds. A number of deep pools have also been identified, with one of the deepest (Bungay pool: 8m) found immediately upstream of MCW offtake. Riparian vegetation consists of river oaks (*Casuarina cunninghamiana*), weeping bottlebrush (*Callistemon viminalis*), water gums (*Tristaniopsis laurina*) and Lomandra species which are often impacted by farming activities, particularly cattle. The figures below illustrate riverine features found throughout the Manning catchment (Figs. 2.2 to 2.6). The main features associated with the mid to lower Manning River include deeper pool areas (Fig. 2.2), pastoral activities on the slopes (Fig. 2.3), shallow macrophyte-dominated pool areas (Fig. 2.4) and riffle habitats (Fig. 2.5). Figure 2.6 illustrates the nature of many upper tributaries, which generally have intact riparian zones.

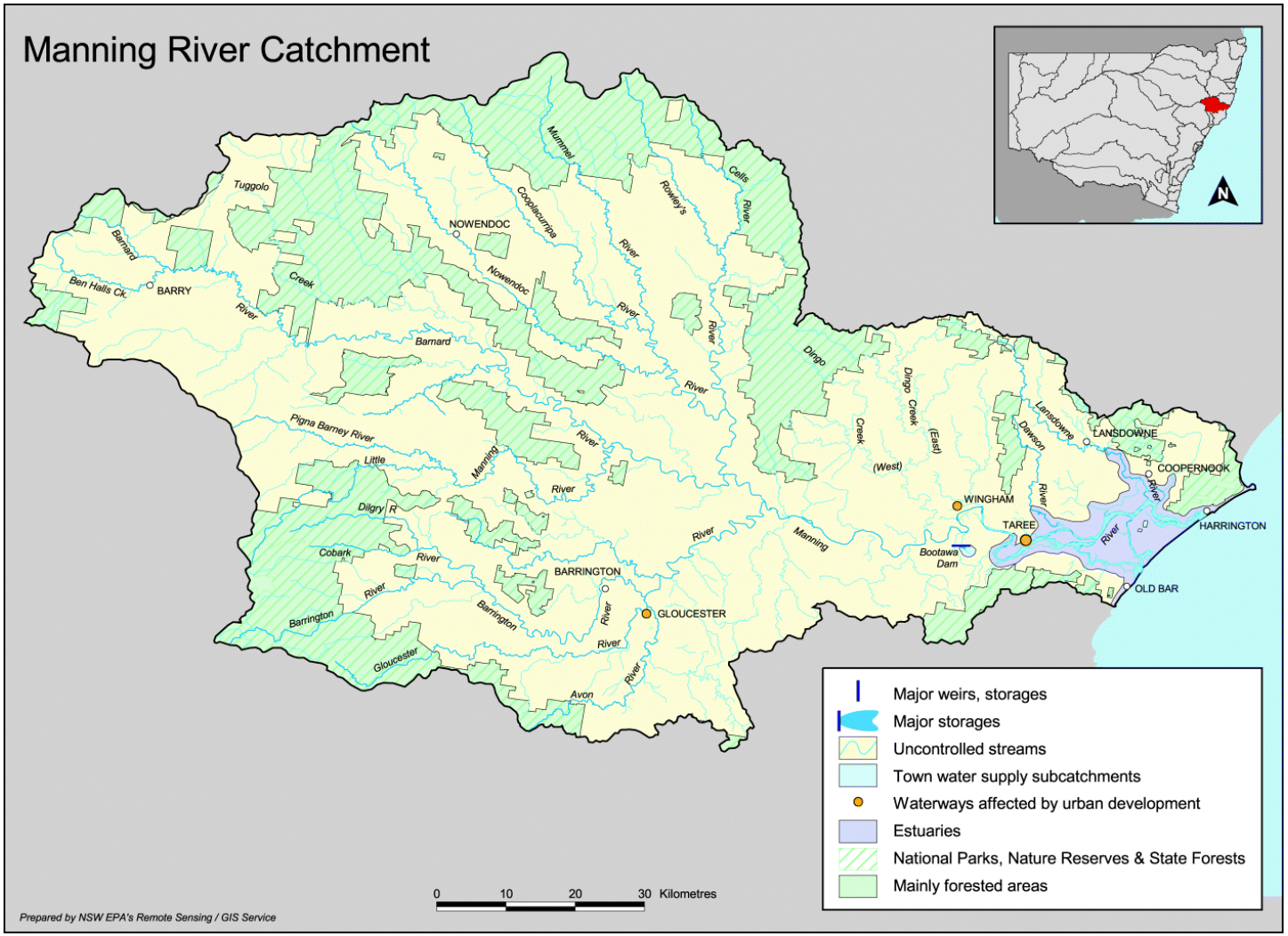


Figure 2.1 Manning catchment (Source: NSW Office of Environment and Heritage)

Figure 2.2 The lower Manning River downstream of Mount George

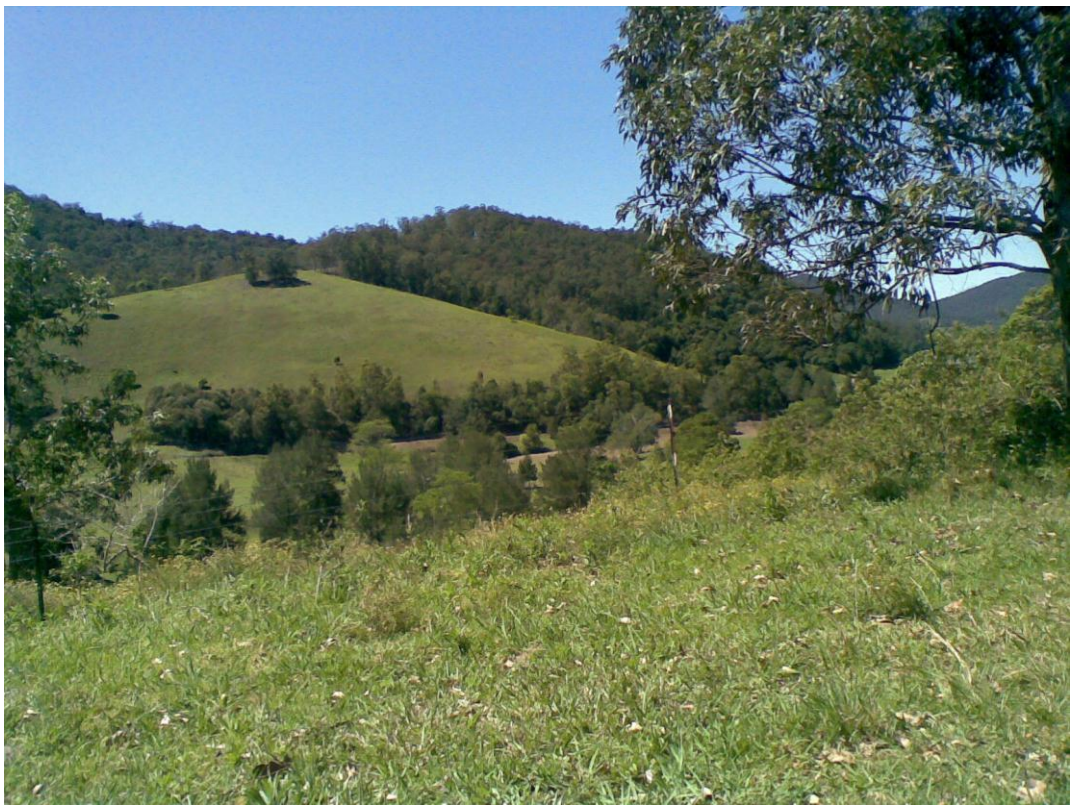


Figure 2.3 Gloucester catchment, cleared grazing areas on slopes.

Figure 2.4 Lower Manning River downstream of Ida Lake, indicating grazed banks and substantial macrophyte growth

Figure 2.5 Lower Manning riffle site indicating cobble and gravel beds and riparian vegetation

Figure 2.6 The Little Manning River, an upstream tributary of the Manning River

Table 2.1 Area of land uses in Manning River Basin 1996-1997 (source: Australian Natural Resource Atlas – website accessed 11 January 2013)

Land Use Description	**Total Extent ('000 ha)**	**Total Extent (%)**
Nature Conservation	102.1	12.5
Minimal Use	273.4	33.5
Livestock Grazing	291.3	35.6
Forestry	128.9	15.8
Dryland Agriculture	14.3	1.7
Built Environment	1.5	0.2
Waterbodies not classified elsewhere	1.1	0.1

Table 2.2 Vegetative communities of the Manning catchment (Australian Natural Resource Atlas – website accessed 11 January 2013)

Vegetative Community	Total Extent ('000 ha)	Total Extent (%)
Cleared	301.7	36.6
Coastal Complex	4.43	0.5
Disturbed Bushland	35.5	4.3
Dry Forest System	168.4	20.4
Exotic Forest System	4.1	0.5
Moist Forest System	266.9	32.4
Plantation	3.1	0.4
Rainforest System	27.0	3.3
Sub-Alpine Woodland	0.6	0.1
Woodland System	12.7	1.5

2.1 Geology

The Manning catchment consists of three main physiographic zones which are mainly rugged or mountainous with only ten per cent of flat terrain. These zones are plateau areas, dissected uplands, and alluvial valleys and coastal riverine plains (ERM 1996).

The Manning is encircled to the north and west by extensive areas of dissected uplands and plateaus, with the highest peaks reaching over 1500 m above sea level (Birrell 1987). Amongst these plateaus are the Barrington and Gloucester Tops, Nowendoc, Bulga and Comboyne Plateaus. The rivers flow from these dissected uplands and plateaus into broad valleys with low gradients, dropping 1500 m along its 250 km length. At the eastern edge of these zones the streams descend into foothills through deep gorges and steeply graded valleys.

Zones of ridges and valleys exist in the central part of the Manning joining the coastal plain with plateaus. The valleys are wide and flat-floored with steeply rising valley walls merging into ridges. The valleys of smaller tributary streams are narrow and steeply graded with deep gullies dissecting the slopes of ridges (Birrell 1987). Wide basins are extensive along the Barrington, Manning, Barnard, Nowendoc and Rowleys Rivers. Small alluvial flats border the meanders on larger tributary streams.

Geology within the Manning catchment consists of a range of Devonian sedimentary and metamorphic rocks which are overlain by lower Permian and Carboniferous sediments and a small area of Triassic sediments with an area of Permian granite overlain by Tertiary basalt in the

plateau areas (Garrard [n.d.1]). Soils include basaltic and alpine humus soils in the upper catchment areas of dissected uplands and plateaus, intensely weathered yellow podzolic soils throughout large areas of the uplands and alluvial valleys, highly fertile alluvial floodplain soils in the valleys and coastal riverine plains and hydromorphic and acid swamp soils in the coastal riverine plains (Garrard [n.d.]). The typically slow draining, low fertility clay soils of the Manning catchment limits irrigation potential. Streambank erosion along sections of the Manning River is common place.

2.2 Climate

The Manning catchment is within the sub-tropical climatic zone extending along the north coast of New South Wales, Australia. It is characterised by warm temperatures throughout the year with seasonal and daily extremes restricted to western upland sections of the valley. The average pattern is one of hot, wet and humid weather in summer and autumn, followed by generally drier, cooler conditions into winter and spring. Mean daily maximum temperatures for Taree vary from 28.3 to 28.9^{o}C in summer, and from 18.6 to 20.1^{o}C in winter. Likewise, mean daily minimum temperatures vary from 16.4 to 18.1^{o}C in summer and from 5.3 to 7.2^{o}C in winter (Bureau of Meteorology 2012).

Mean annual rainfall for Gloucester in the upper catchment is 984mm and for Wingham in the lower catchment the mean annual rainfall is 1100mm (Figs. 2.7 and 2.8). Despite the average trends, the variability index for monthly rainfall ranges from 1.9 – 4.5. Relative to other parts of Australia these variability index values are considered to be very high (1.5 to 2) or extreme (>2) and episodes of drought and flood are therefore considered likely within the Manning catchment (MCW 2006). Maximum rainfall occurs during late summer and early winter with lower rainfall generally occurring during late winter and early spring (Birrell 1987). Rainfall distribution is dominated by the topography with the highest rainfall occurring along the plateau escarpments on the northern extent of the valley and the ridges. As a result of the total annual rainfall experienced throughout the catchment, the need for irrigation is not substantial and irrigation is not widely used in the area.

[1] No Date

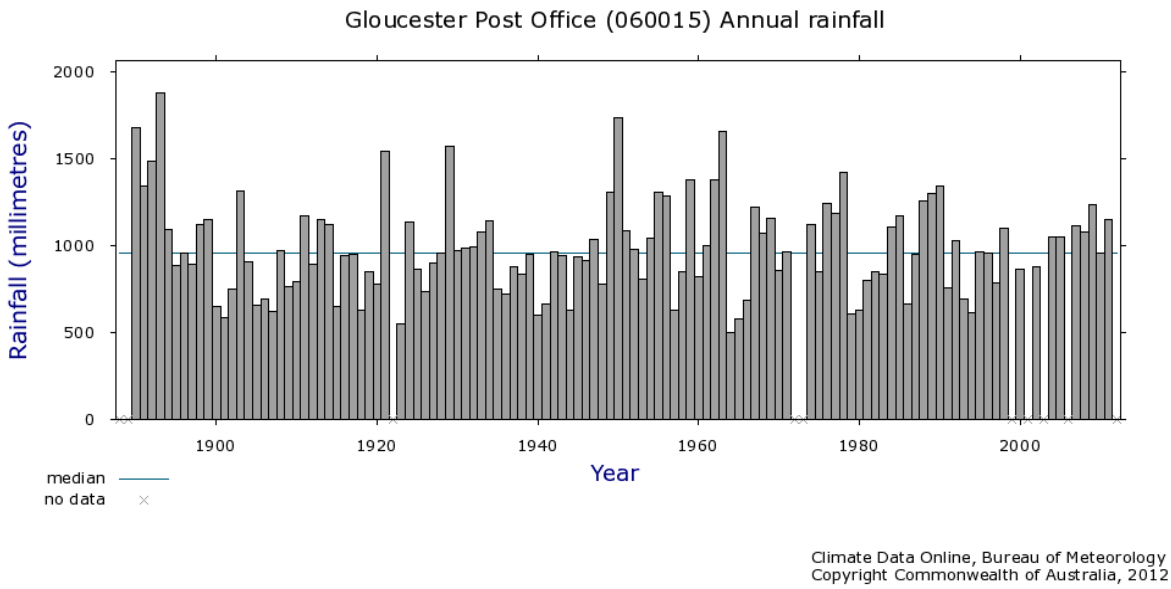


Figure 2.7 Long-term (1890-2012) annual average rainfall recorded at Gloucester, NSW
(Years with no available data are indicated by 'x' below blank columns - causes for missing data are typically due to an equipment fault or a site closure for a period of time BOM 2014.)

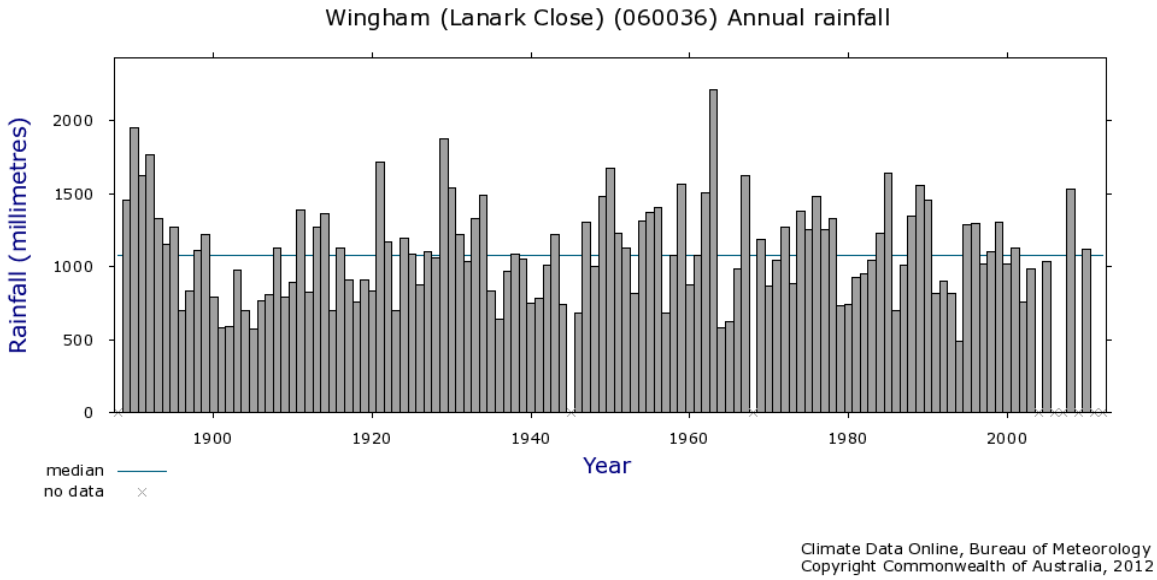


Figure 2.8 Long-term (1890-2012) annual average rainfall recorded at Wingham, NSW
(Years with no available data are indicated by 'x' below blank columns - causes for missing data are typically due to an equipment fault or a site closure for a period of time - BOM 2014).

2.3 Climate change

Climate change is projected to alter runoff and water availability by 2050, increasing extremes of dry and wet periods. Predicted changes to seasonal temperatures and hydrologic variables as a result of climate change are likely to impact on rivers of the Manning catchment (CSIRO 2007a). The future climate associated with the NSW Mid-North Coast is likely to be warmer and, given projected increases in evaporation, the catchment is likely to be drier but with potential for seasonal increases in extreme rainfall events (CSIRO 2007a). These changes in rainfall and higher evaporation rates are likely to lead to less runoff for streams and rivers located in this region.

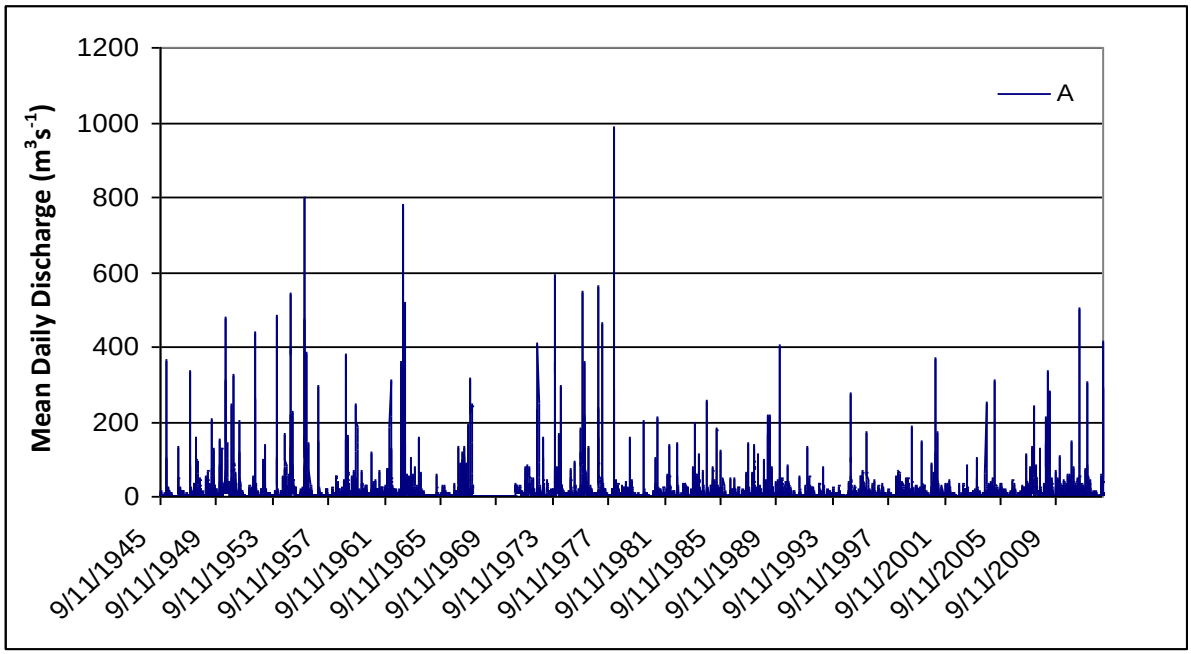
These changes may result in a decline in water quality by creating a more favourable environment for potentially harmful algal blooms or increasing aquatic plant growth to a level that results in localised deoxygenation of the water column (CSIRO 2007b). It is likely that warmer water temperatures and long periods of low flows will also result in greater thermal stratification within deep pools. In addition to effects on surface water, the quality of coastal groundwater aquifers may be adversely affected by rising sea levels and salt-water infiltration (CSIRO 2007b). The degree to which these changes influence stream health will be dependent on the condition and geomorphology of the receiving stream and the level of extraction or extraction patterns.

Although climate change is predicted to affect the ecological communities of regulated river basins to a greater degree than more free-flowing river basins such as the Manning (Palmer et al. 2008), the projected increase in extreme conditions is likely to exacerbate existing impacts from extraction, land use and pest species in unregulated systems (Durance and Ormerod 2009, Palmer et al. 2009). While environmental flows have been the focus of conservation strategies for regulated rivers (Pittock and Finlayson 2011), this approach is not appropriate for unregulated systems which require a different approach. Measures, such as freshwater-protected areas, can build resilience of freshwater ecosystems, both regulated and unregulated, to climate change impacts (Pittock and Finlayson 2011). This approach is a crucial step in the conservation of free-flowing river ecosystems where maintenance of ecological processes can enhance their capacity to resist climate change impacts, and where adaptation may be maximised. The need to employ good governance and rigorous adaptive management is essential for effective conservation of riverine environments (Kingsford 2011).

2.4 Hydrology

The unregulated nature of streams within the Manning has resulted in the maintenance of the natural variability of flows. As hydrological variability is a key regulator of lotic ecology (Poff et al. 1997, Thoms 2006), the range of flows in the Manning system indicates that the associated ecological processes and functions would be dependent on a high degree of flow fluctuation. This assumption is supported by the variability index for monthly rainfall in the Manning region as discussed previously (MCW 2006).

Flooding is widespread along the Manning valley with the urban areas of Taree, Wingham, and Gloucester particularly prone to inundation from moderate floods (MCW 2006). Moderate flooding occurs in these areas when the Gloucester River exceeds 4.0m at Gloucester or the Manning River exceeds 8.9 m at Wingham. These high flows scour riverine habitats and increase habitat availability, while overbank flows provide opportunities for energy transfer and increased connectivity (Boulton 2003). The 1950s and 1960s appeared to have higher discharges under wetter conditions whereas discharges in the 1980s and 1990s indicate a period of drier conditions, with a reduced frequency in high flows (Figs. 2.9 and 2.10). The reduction in flow volume is likely to reduce habitat availability, reduce longitudinal and lateral connectivity and alter energy pathways (Junk 1999). It was during this drier period that Water Sharing Plans were developed throughout NSW, with particular focus on protecting low flows in unregulated water courses. Details of the Water Sharing Plan for Lower North Coast are discussed further in section 2.7 of this chapter.



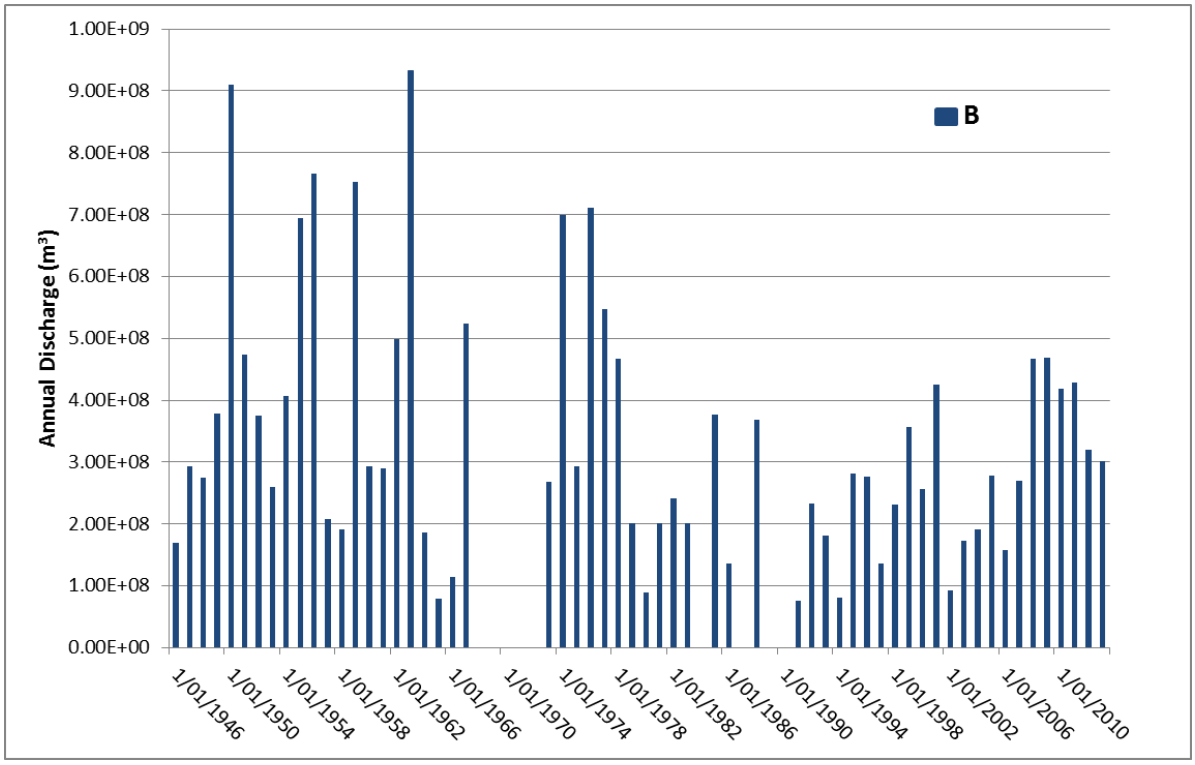
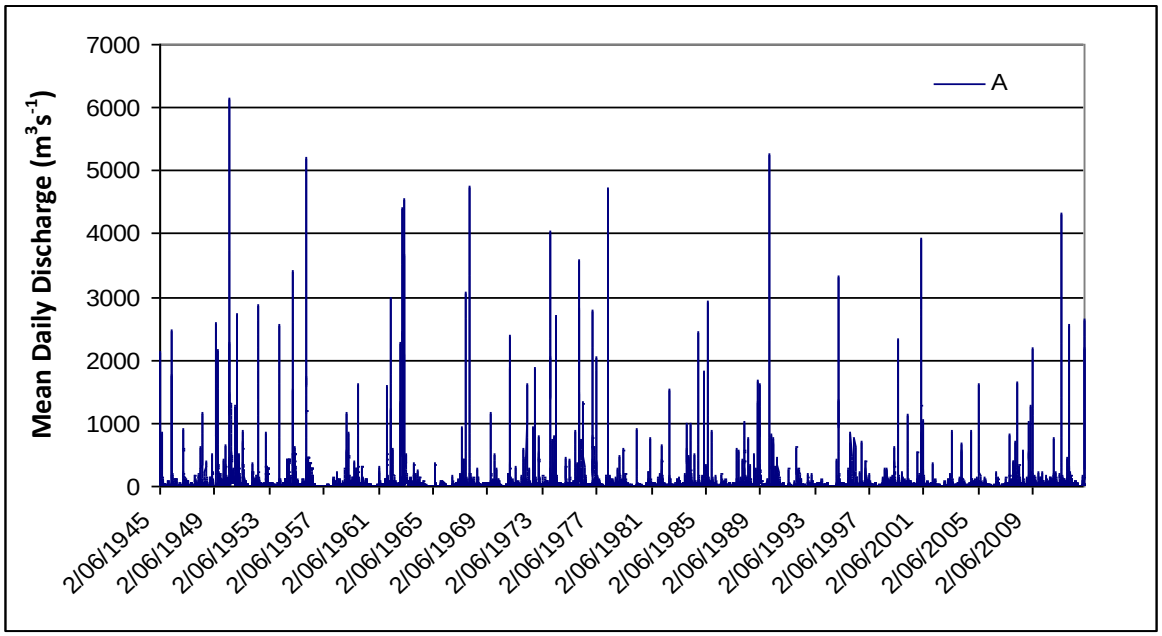


Figure 2.9 A) Daily and B) Annual discharge at Barrington River at Forbesdale



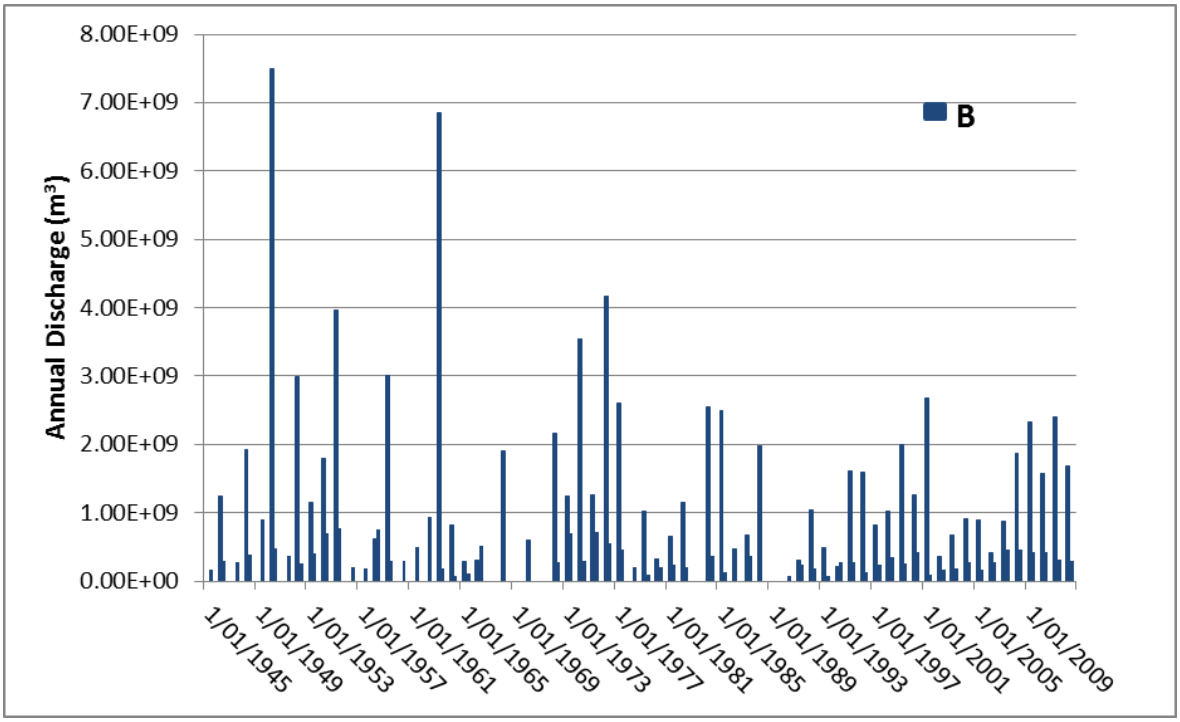


Figure 2.10 A) Daily and B) Annual discharge at Manning River at Killawarra

2.5 Low flows in the Manning catchment

Recurrence intervals using peak annual discharge can be used to indicate the frequency and size of moderate and high flows. Figures 2.11 and 2.12 illustrate the recurrence intervals for flows in the Barrington and Manning Rivers, respectively. One-in-10 year events are around 7,000 m^3s^{-1} for the Barrington River. At 35,000 m^3s^{-1} for 1-in-10 year events, the Manning River displays approximately 5 times the flood volume during these events when compared to the Barrington.

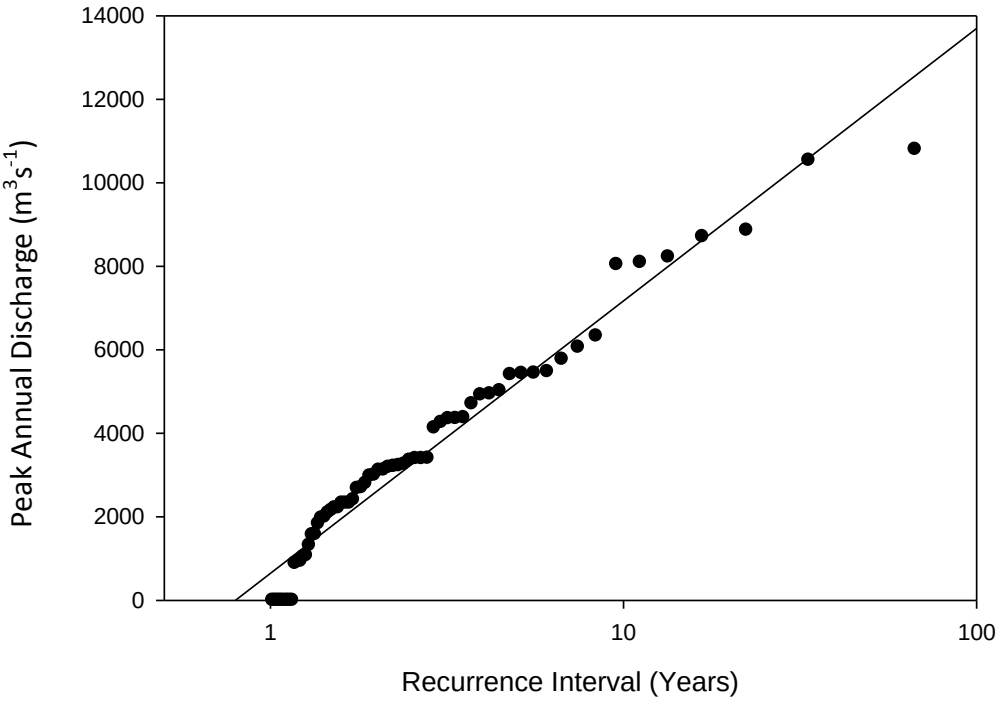


Figure 2.11 Recurrence intervals for peak annual flows in the Barrington River at Forbesdale (Source: NSW Office of Water – NSW Water Information website)

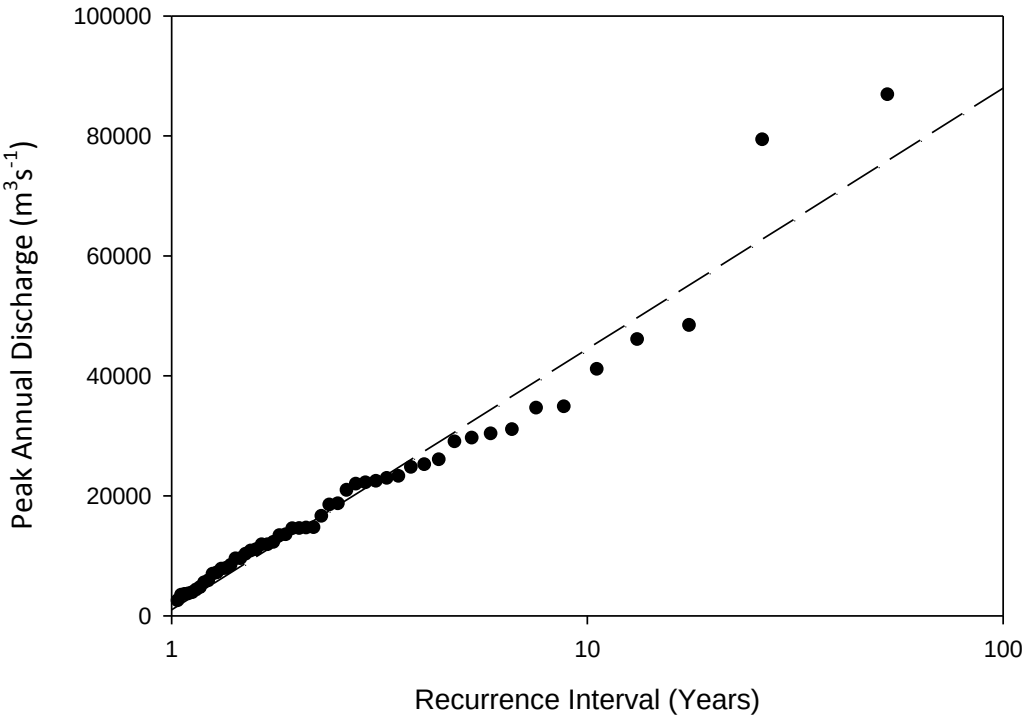


Figure 2.12 Recurrence intervals for peak annual flows in the Manning River at Killawarra (Source: NSW Office of Water – NSW Water Information website)

2.6 Water use

Water resources available within the catchment are used for agricultural, industrial and urban purposes. Water extracted from the Manning River supplies town water to Wingham, Taree, Tuncurry and Forster, while the Barrington River is used for water extraction for the Barrington and Gloucester Water Supplies. The Manning River system receives licenced treated effluent from a number of townships including Gloucester and Wingham.

The Manning River has extraction licence entitlements of approximately 55,500,000 m^3year^{-1}. Of these licence entitlements 23 per cent are allocated for urban development with 610,000 m^3year^{-1} from the Barrington River and 12,500,000 m^3year^{-1} from the lower Manning (MCW 2006). The remaining 505 licences that have access to surface water, extract water for industry and irrigation purposes, with around 5,800ha licenced for irrigation. Groundwater licence entitlements in the Manning catchment equals 2,206,000 m^3year^{-1} for 33 licences, representing about 4% of all licensed extraction within the catchment (MCW 2006).

Allocations to licences relative to river flows are shown in Figure 2.13. The figure demonstrates the greater impact extraction is likely to have under low flow conditions as demonstrated by the 95th percentile flows. The recognition of the greater stresses placed on the river system, particularly under low flows, has resulted in the development of Water Sharing Plans which set cease-to-pump and commence-to-pump thresholds for water users when flows fall below the 97th percentile and also encourage extraction during high flows. The proposed incentive through the Water Sharing Plans is to allow water users that convert to taking water during higher flows to be granted additional volumes of water (DWE 2009b)

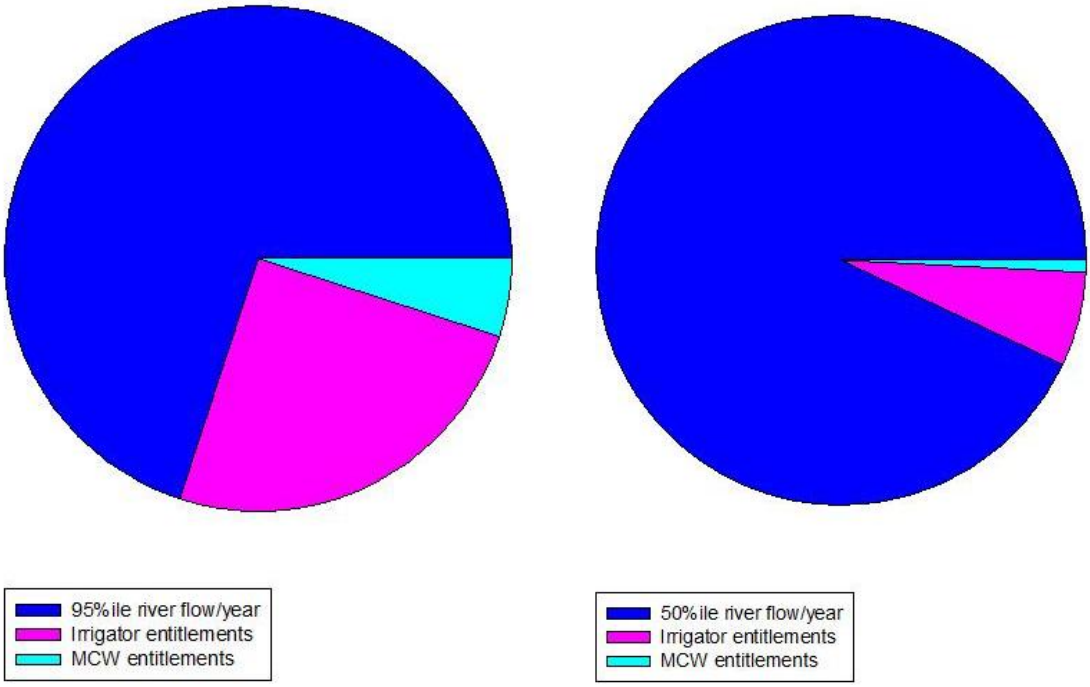


Figure 2.13 Licenced entitlements at 50th percentile and 95th percentile flows in the Manning River 2004/05 (Source: MCW 2006)

2.7 Water Sharing Plan

Water Sharing Plans have been developed across NSW to provide a legal basis for sharing water between the environment and consumptive purposes. The Water Sharing Plan (WSP) for Lower North Coast Unregulated and Alluvial Water Sources commenced on the 1st August 2009 (DWE 2009a). The Plan covers the Lower North Coast unregulated rivers, including the Manning, and the highly connected alluvial groundwater above the tidal limit. It also includes the tidal pool areas which have not been previously covered by the NSW *Water Act 1912* (DWE 2009a). The purpose of the Plan is to address the continued expansion of water use across NSW and the increasing competition between water users for access to water (DWE 2009a). The pressure of increasing water use has had a detrimental impact on the health and biological diversity of many rivers and aquifers (Nilsson et al. 2005, Poff et al. 2007, Poff and Zimmerman 2010). The Manning River also receives added environmental pressure from an expanding human population, expected to increase by more than 25,000 by 2020, and climate change (GTCC 2005).

Under the NSW *Water Management Act 2000*, the sharing of water must firstly protect the health of the water source and its dependent ecosystems, followed by basic landholder rights (DWE 2009b). Amongst licensed water users, priority is then given to water utilities and licensed stock and domestic use, ahead of commercial purposes such as irrigation and industry.

Access rules were identified through a process that classified each water source as high, medium or low on the basis of instream and economic values and the risks to these values. The classification method took into account the amount of water extracted, the impact of extraction, the associated uses from this extraction and the social and economic considerations of restricting extraction (DWE 2009a). Specifically the classification process involved assessment of factors such as instream values, the risk to these instream values from water extraction, the amount of water extracted relative to low flows, the economic value of the agriculture reliant on water extraction and community dependence on the economic value (DWE 2009a). Figure 2.14 demonstrates ecological conditions in the lower Manning River under low flow conditions.

Figure 2.14 Low flows in the lower Manning River

For the first five years in the Plan (2009 to 2014), cease-to-pump conditions exist when there is no visible flow at the Manning River at Killawarra Gauge. From year six of the Plan (2015) cease-to-pump conditions are in place when flows are at or below 98,000 m^3s^{-1} (equivalent to the 98th percentile) on a falling river at the Killawarra Gauge. Additionally, the cease-to-pump condition will also apply to aquifer access licences extracting from all alluvial aquifers within 40m of an unregulated river, except for existing Stock and Domestic access licences and Local Water Utilities Access licences (DWE 2009b).

Similarly, during the first five years of the Plan, commence-to-pump can occur when there is visible flow at a pump site and/or the Manning River at Killawarra Gauge. From year six of the Plan, commence-to-pump can occur when flows are greater than 137,000 m^3s^{-1} (equivalent to the 97th percentile) on a rising river at Manning River at Killawarra.

These rules apply to the majority of access licences including for the purposes of irrigation, farming, industrial and recreational uses. However, there are limited volumes available below this cease-to-pump threshold for some town water supplies and for licence holders that require continued access to water for hygiene and health purposes.

As there is a lack of knowledge to assist in setting the Plan rules it is important to investigate how the river and its tributaries respond to low flows and if the current and proposed rules are adequate to protect the ecological condition of the river. This thesis aims to address some of these knowledge gaps by investigating the physical, chemical and biological response to changes in flow across the Manning catchment. Increased knowledge will help inform the establishment of access rules that better protect the ecological condition of the Manning River.

Figure 2.15 Lower Manning River pool early morning