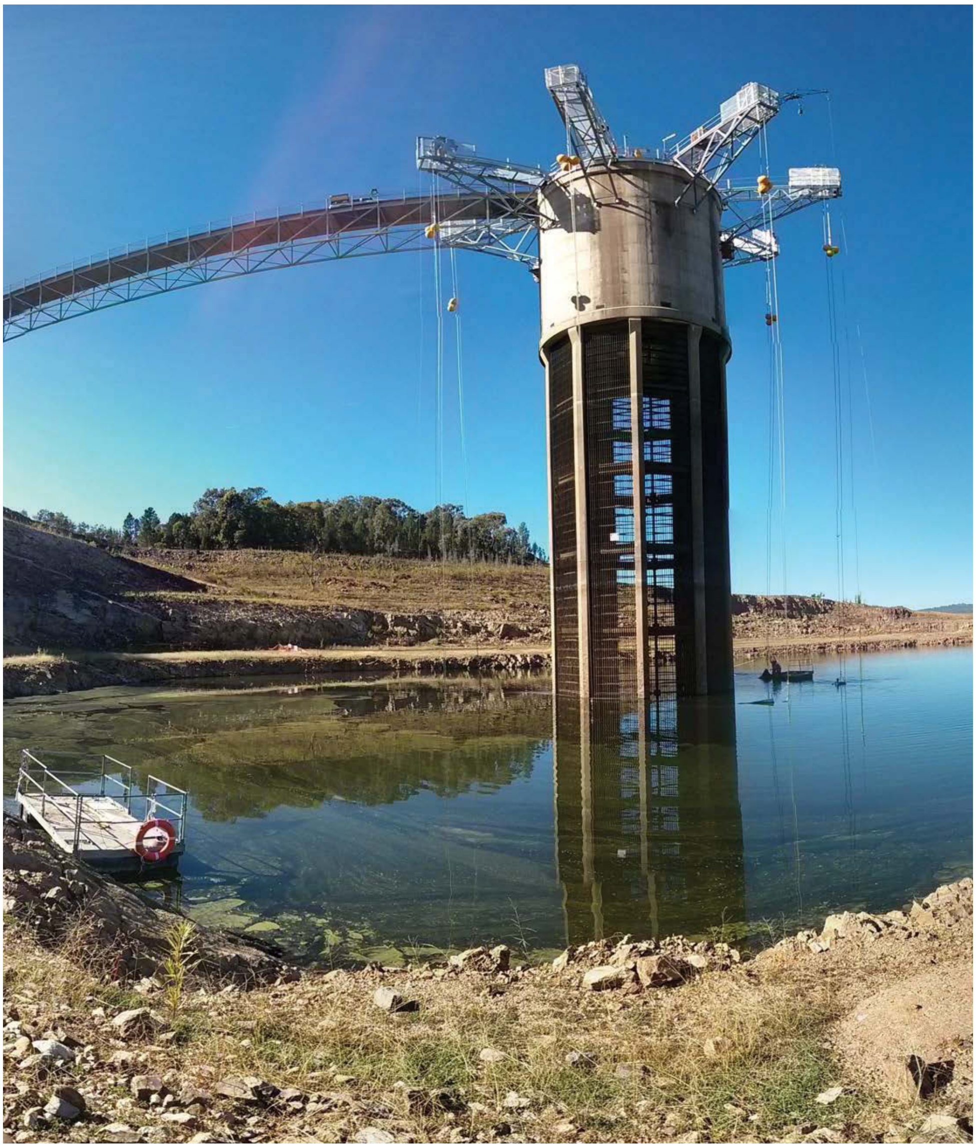
Effectiveness of cold water pollution mitigation at Burrendong Dam using an innovative thermal curtain



Master of Science by Research
2016

Rachel Gray

School of Life Sciences
Faculty of Science
University of Technology, Sydney

This thesis has been written in the style of a journal. Each chapter has been written as stand-alone chapters, in the style of journal articles so some repetition unavoidably occurs.

Certificate of Original Authorship

I certify that the work in this thesis has not previously been submitted for a degree nor has it been submitted as part of requirements for a degree except as fully acknowledged within the text.

I also certify that the thesis has been written by me. Any help that I have received in my research work and the preparation of the thesis itself has been acknowledged. In addition, I certify that all information sources and literature used are indicated in the thesis.

Rachel Gray

Acknowledgements

This research was funded by Water NSW, DPI Water and NSW Fisheries. Thank you to the Cold Water Pollution Inter-Agency Group (CWPIAG) for pioneering this project and for their continuous support.

Sincerest thanks to my primary supervisor, Dr Simon Mitrovic from the School of Life Sciences at the University of Technology Sydney and DPI Water for his ongoing support, expertise and many reviews of this thesis. Also to my industry supervisor, Lorraine Hardwick from DPI Water, for her guidance in field work and reviews of this thesis.

Special thanks are due to Dr Hugh Jones from the NSW department of Environment and Heritage, for his assistance and expertise in the analysis of data, modelling, and many reviews of this thesis. Thanks also to Tony and Anne Kelly, for accommodating me over the course of the field work portion of this research, and for the many dinners and invaluable support and enthusiasm through the duration of the project.

Thanks to the staff at Burrendong Dam, particularly Rodney Wilson and Roger Ryan for allowing me access to downstream sites and the use of their boat for regular sampling runs as well as thermistor chain data. Thanks also to the Wellington Historical Society, who granted me copies of historical photos of Burrendong Dam from the Oxley Museum in Wellington.

I would also like to express my thanks to the various volunteers who assisted me in the field work for the project, as well as my family and colleagues at the University of Technology Sydney for their unwavering support, and for making this project truly unforgettable.

Contents

List of Figures

List of Tables

Abstract

Many dams in Australia are known to create thermal pollution in rivers, often for hundreds of kilometres downstream of the dam wall. Low-level releases from a reservoir during periods of thermal stratification disrupt the downstream thermal regime by supressing the water temperature and reducing the diel variation. Low-level releases have also been linked to elevated nutrient concentrations and altered phytoplankton density and community structure downstream from the dam. To reduce the problem, Burrendong Dam has been fitted with an innovative thermal curtain that directs warmer surface water to the low-level off-take.

This study set out to quantify the magnitude and extent of cold water pollution along the Macquarie River downstream of Burrendong Dam before and after the curtain was implemented. It also aimed to quantify the effect of the dam on nutrient concentrations and phytoplankton density (indicated by chlorophyll *a* concentrations) and community structure before and after thermal curtain operation. This was achieved through analysis of time-series data from temperature loggers installed within the impoundment, upstream and downstream in the Macquarie River, prior to the installation of the thermal curtain. Water samples for analysis of nutrients, phytoplankton concentrations (chlorophyll *a*) and community structure were collected within the reservoir and upstream and downstream of the dam.

CWP was shown to be a long-term problem in downstream river reaches, caused by the release of hypolimnial waters from the dam. Epilimnial releases with use of the curtain improved the thermal regime (mean daily and mean monthly temperature, and

diel temperature range) below the dam so that it more closely resembled the upstream thermal regime with an improvement of approximately 2°C. Fluctuations in nutrients occurred in the hypolimnion of Burrendong Dam during thermal stratification, probably due to the development of an oxycline and subsequently low oxygen concentrations in the hypolimnion. Nutrient concentrations increased at the outlet as a result of hypolimnial releases, with the concentrations breaching the trigger values outlined by ANZECC which indicate ecological disturbance. This study found a substantial increase in the cell count of cyanobacteria at the dam outlet, which may lead to water quality issues in the Macquarie River downstream of the dam. The results of this study will be useful to assist in the management routine of the thermal curtain at Burrendong Dam, to maximise the efficiency of CWP mitigation, whilst not compromising the downstream ecological health in terms of nutrient and cyanobacteria concentrations.

Chapter 1: Introduction

1.1 Cold water pollution

Surface waters of large impoundments are heated by the sun and ambient air temperatures and during warmer months this warm surface layer (epilimnion) can overlay a cold, dense bottom layer (hypolimnion) after thermal stratification occurs. Cold water pollution (CWP) is commonly observed in rivers below large, thermally stratified reservoirs when water is drawn from the unnaturally cold hypolimnion. Hypolimnial releases can result in temperature depression by up to 12°C in rivers (Lugg 1999, Burton 2000b). Depressed temperatures may persist for hundreds of kilometres, with the magnitude of depression gradually lessening with distance from the dam (Lugg 1999, Sherman 2000, Preece and Jones 2002). During periods of thermal stratification, an oxycline may develop leading to anoxic conditions in the hypolimnion. This can cause the release of sediment-bound nutrients into the water column (Baldwin and Williams 2007, Baldwin et al. 2010), resulting in altered downstream water chemistry such as increased nutrient and metal levels. Bottom releases may negatively impact downstream river health including reduced fish spawning and growth, eutrophication from high levels of nutrients, and high levels of dissolved metals (Selig and Schlungbaum 2002, Astles et al. 2003, Todd et al. 2005, Baldwin and Williams 2007, Dent et al. 2014).

1.2 Burrendong Dam and cold water pollution

Burrendong Dam, completed in 1967, was built on the Macquarie River and receives inflows from the upper Macquarie River and the Cudgegong River. The dam was commissioned to secure water to support irrigated agriculture, supply town water to inland towns and cities in western NSW and provide environmental flows to the Macquarie Marshes (Australian National Committee on Large Dams 1982, Green et al. 2011). Discharge occurs through a low-level off-take, resulting in hypolimnial releases during periods of thermal stratification. Periods of stratification often coincide with the largest discharge volumes from the dam which occur in summer to support the agricultural requirements. As a result of these factors, cold water pollution of up to 10°C is a recognised problem affecting the Macquarie River downstream of the dam (Harris 1997, Burton 2000b). Cold water pollution may affect the Macquarie River as far downstream as Gin Gin (250km) or Warren (312km) (Harris 1997). As such, a solution was needed to address the problem of CWP negatively influencing the water quality, affecting anthropogenic use and riverine ecological health.

Preece and Wales (2004) identified Burrendong Dam as a 'high priority' reservoir likely to cause CWP. Dam and weir structures in NSW were assessed and ranked according to criteria including discharge volume and off-take depth. Burrendong was ranked 5th out of the 304 structures in NSW.

1.3 Options for CWP amelioration: The Thermal Curtain

Most of the current options available for the mitigation of CWP are costly. One of the most effective options for mitigating cold water pollution are multiple level off-takes (MLO's) which can be used to selectively withdraw water from desired depths within the reservoir (Department of Public Works and Services 1996, Lugg 1999) and have been reported to cost approximately $25 million in 2000 (Sherman 2000), equivalent to approximately $37 million in 2014. There are drawbacks with this system as changing the withdrawal level can take considerable time (days) and effort (1-2 operators). The thermal curtain installed into Burrendong Dam is an innovative approach that is cheaper (approximately $4 million) than of a multi-level off-take (MLO). The thermal curtain has a cylindrical shape, encircling the off-take tower and extending up towards the water surface. The top of the curtain is designed to sit approximately 3-10 m below the water surface to channel warmer water from the surface waters (epilimnion) to the off-take tower and through the dam outlet downstream. A chain and pulley system attached to the off-take tower has mechanised the curtain to allow the depth below the surface water to be adjusted as required. This allows some control of the water to be drawn into the curtain from the epilimnion, though there is less control and selectivity compared to a MLO structure.

1.4 Possible downstream effects of thermal curtain use

By allowing the selective withdrawal from the epilimnion of the stratified reservoir, the downstream nutrient levels may be affected. Whilst nutrient concentrations are often high in the hypolimnion, they are often depleted in the surface layer of the water column (Petts 1985). By drawing water from the epilimnion, nutrient concentrations in the downstream stretches of the Macquarie River are likely to decrease compared to prior years, when water from the hypolimnion was released downstream through the low-level off-take.

A potential drawback with the thermal curtain is that it is likely to increase levels of phytoplankton downstream, including toxic blue-green algae (cyanobacteria). Under pre-curtain operation, there were low concentrations of phytoplankton in bottom-release waters from the dam. Once the curtain is in place, drawing water from the surface, greater concentrations of phytoplankton are expected to be released downstream. This should be especially the case when the reservoir is stratified because phytoplankton biomass is usually greatest at this time in the epilimnion, and toxic cyanobacteria often dominate (Sommer et al. 1986). If released, these high phytoplankton concentrations may persist for a considerable distance downstream, and if toxic, pose a risk to water users and the ecological health of downstream ecosystems (Ingleton et al. 2008).

Research is needed to quantify the magnitude and extent of CWP on the Macquarie River as a result of Burrendong Dam, and to assess the effectiveness of the curtain in reducing the impacts of cold water pollution as well as its other effects such as on downstream nutrient concentrations and algae (including cyanobacteria). If effective, it is likely that the curtain design will be utilised for addressing cold water pollution problems at other large dams affected by CWP (Water NSW *pers. comm*.). Across NSW there are approximately 55 dams that produce CWP (Ryan and Preece 2003).

1.5 Research aims and hypotheses

This project aims to quantify the magnitude of cold water pollution (temperature) below Burrendong Dam under pre-curtain management arrangements. This will form a basis for comparison to assess the effectiveness of the thermal curtain structure in reducing the detected magnitude of CWP immediately downstream of the dam wall. Further, the distance and extent of the downstream cold water pollution will be assessed before and after the use of the thermal curtain. The results of the study are expected to assist in the effective management and operation of the thermal curtain. The study also aims to assess the pre- and post-curtain impacts of the dam on nutrient concentrations, and phytoplankton and cyanobacterial concentrations in the Macquarie River. These three broad aims were developed into a set of specific hypotheses:

- Aim 1: Determine the effect of the dam on the thermal regime of the Macquarie River prior to and after the operation of the thermal curtain.

  Hypotheses:

  a. Historical analysis of temperature data will indicate the incidence of CWP immediately below the dam during the warmest months.
  b. Use of the thermal curtain will show increases in temperature which better reflect predicted natural temperatures.
  c. There will be an improvement in thermal recovery distance along the length of the Macquarie River with the use of the thermal curtain.

- Aim 2: Determine the effect of the dam on the nutrient concentrations of the Macquarie River immediately downstream of the wall prior to and after the operation of the thermal curtain.

  Hypotheses:

a. During periods of stratification, the hypolimnion will have greater nutrient concentrations and lower dissolved oxygen (DO) concentrations than the epilimnion.
b. During periods of stratification, nutrient concentrations in hypolimnial releases will be greater downstream than upstream of the reservoir.
c. After the curtain is implemented, downstream nutrient concentrations will be lower in epilimnial releases compared to concentrations in hypolimnial releases.

- Aim 3: To determine the effect of the dam on the phytoplankton concentration (chlorophyll *a*) and cyanobacterial concentrations in the Macquarie River prior to and after the operation of the thermal curtain.

  Hypotheses:

  a. Prior to thermal curtain operation, chlorophyll *a* concentrations in hypolimnial releases downstream of the dam will be low.
  b. Chlorophyll *a* concentrations and cyanobacterial densities will increase downstream of the dam following the use of the thermal curtain and epilimnial releases.

Hypotheses 1a, b and c will be answered by:

Monitoring surface water temperatures in the Macquarie River at sites located from 108 km upstream to 312 km downstream of the dam. Thermal profiles within the reservoir were measured to establish the periods of thermal stratification. Upstream sites will be used as reference sites to indicate the natural, unregulated thermal regime of the river.

Hypotheses 2a, b and c will be answered by:

Monitoring the nutrients (total phosphorus, total filterable phosphorus, nitrogen oxidised as N, total nitrogen, total filterable nitrogen, ammonia, silica, and filterable reactive phosphorus) and iron concentrations with depth profiles within the reservoir, upstream and immediately downstream of the dam wall during periods of mixing and stratification. The monitoring will be conducted from one year preceding, until one year after the operation of the thermal curtain.

Hypotheses 3a and b will be answered by:

Monitoring the chlorophyll *a* concentration with depth profiles within the reservoir, and longitudinally along the river from 104 km upstream to 152 km downstream of the dam wall. Cyanobacterial concentrations will also be monitored immediately below the dam wall. The monitoring will be conducted one year prior to and after the operation of the thermal curtain.

Chapter 2: Literature Review

2.1 Freshwater reservoirs

2.1.1 Freshwater resources on a global scale

Once considered a renewable and plentiful resource, freshwater availability is now being threatened by over-use by humans on a global scale (Hanjra and Qureshi 2010, Srinivasan et al. 2012). Water is a precious, finite resource depended upon by humans for urban, industrial and agricultural processes and is vital for human health, economic growth and ecological health and functioning (Gordon et al. 2004, Gleick and Palaniappan 2010, Hanjra and Qureshi 2010, Jowsey 2012).

With the global population predicted to reach 9 billion by 2050 (United Nations 2014), the already high demand for water use in the urban and agricultural context is due to increase. Since the early 1950's, global water demand has increased threefold, whilst water supply has been progressively decreasing (Gleick 2003). Modelling by Shiklomanov (2000) showed irrigation to be the dominant sector to withdraw water, being responsible for 66% of total water withdrawals globally. The need to expand irrigation coverage in countries including Africa, Ethiopia and the Arab region to meet food demands and gain food security is well documented (Lobell et al. 2008, Hanjra et al. 2009, Sulser et al. 2011). Additionally, anthropogenic climate change is predicted to magnify these water demands (Lobell et al. 2008) with altered spatial and temporal distribution of rainfall (Hanjra and Qureshi 2010). Therefore, there is an expected increase in water demand to support continued global population growth.

2.1.2 Freshwater resources in Australia

Australia is considered a dry continent, with one third of the country classed as arid (receiving an annual rainfall of less than 250mm) and another third as semi-arid (receiving 250-500mm of annual rainfall) (Raupach et al. 2001). The Australian economy is heavily dependent upon the agricultural industry, with the sector generating up to $4.3 billion in gross value annually (Wells 2015).

Much of the farming land is located in semi-arid and arid environments (Chapman et al., 1996, Winter et al., 1996), thus requiring water for irrigation in consistently dry circumstances to support vegetative growth. Irrigation flows are released from dams (lentic systems) to supply farmers with water necessary for farming purposes (Maheshwari et al. 1995, Lemly et al. 2000, Sattari et al. 2009). Approximately 75% of stored water is used for irrigation (Tisdell et al. 2002). Rice and cotton crops require an average of 1300 and 600mm irrigation water applied per crop, respectively (Hochman et al. 2013). Furthermore, cotton is the third most water intensive agricultural activity in the industry, using 15% of all irrigation water allocations (Hochman et al. 2013).

With such a high economic dependence on the agriculture industry, it is essential it is preserved. Water resources are facing growing pressure with climate change, agricultural practices and a growing global population making dams a necessity to maintain water and food security.

2.1.3 Dams in Australia

There are at least 564 large dams in Australia (ANCOLD 2012). A large dam is internationally recognised as having a height of greater than 15 metres, measured from the lowest point of foundations to the crest of the wall, or greater than 10 metres in height under the conditions that (1) the crest is not less than 500 metres in length, (2) the capacity of the dam is over 1 million cubic metres, (3) the maximum flood discharge of the dam is at least 2000 cubic metres per second or (4) the dam is of unusual design (ANCOLD 2012).

2.2 Thermal regime

2.2.1 Thermal stratification of a water body

Thermal stratification is the distinct layering of a water body, usually between spring and autumn, whereby intense solar radiation heats the surface waters faster than deep layers (Fig. 1) (Sherman 2000, Jorgensen et al. 2005). The phenomenon typically occurs in static or slow moving water bodies (Bormans and Webster 1997, Sherman et al. 1998, Mitrovic et al. 2003) and can occur in almost any water body with a depth greater than 5 m (Harleman 1982). The upper layer (epilimnion) is of low density, and remains warm from the continuous impact of solar radiation throughout the day. A steep transitional zone of temperature decline with increasing depth (thermocline, or metalimnion) exists between the epilimnion and the cold, dense bottom layer (hypolimnion). The differing densities of the epilimnion and hypolimnion further establish the divide between layers (Turner and Erskine 1996, Turner and Erskine 2005, Gassama et al. 2012).

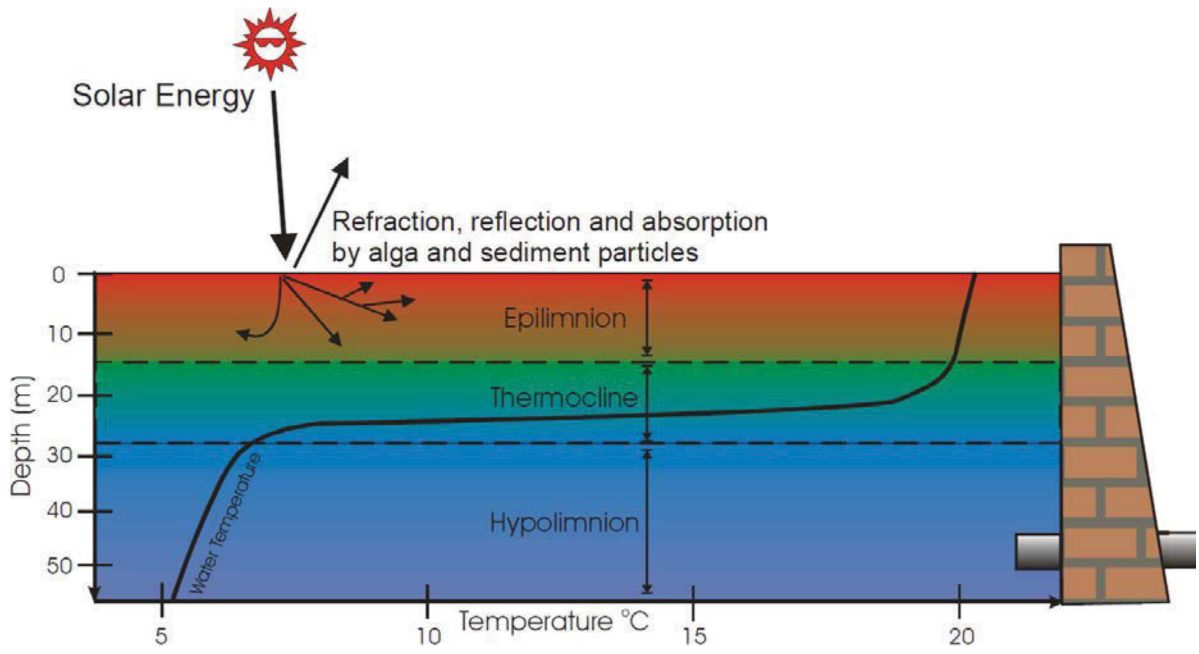


Figure 1: Illustration of thermal stratification within a reservoir. Image sourced from EPA Victoria (2004).

Commonly, an oxygen gradient develops in conjunction with thermal stratification (Turner and Erskine 2005). A steep transitional zone of decreasing oxygen concentration with depth exists, where the surface waters are well oxygenated and the bottom waters are often anoxic. This occurs due to the lack of mixing of water density layers and the consumption of dissolved oxygen by respiring organisms below the oxycline (Beutel 2001, Matzinger et al. 2010). A reducing environment ensues, resulting in the release of sediment bound compounds including nitrogen, phosphorus and iron (Mortimer 1941, Bormans and Webster 1997, Turner and Erskine 2005). Thus the hypolimnion is typically characterised by unnaturally cold, oxygen-depleted and nutrient-rich water.

In winter, the decrease of air temperature and solar radiation and the increase of seasonal wind action leads to the mixing of the water body and subsequent destratification (Bormans et al. 1997). As the surface temperature of a stratified reservoir drops, the water density of this layer increases and sinks due to the pull of gravity (Jorgensen et al. 2005, Jung 2009). Furthermore, in rivers (lotic systems) with a maximum annual rainfall in winter, mixing is further encouraged by fluvial turbulence (Turner and Erskine 2005). The upwelling of nutrient-rich hypolimnial water to the warm epilimnial water can cause eutrophication within the photic zone, often creating optimal environmental conditions for excessive phytoplankton proliferation (Reynolds and Walsby 1975). When blooms do develop, they are often dominated by potentially toxic cyanobacteria, posing a risk to the water quality of the reservoir, and to downstream river systems if released through the outlet (Sherman et al. 1998).

2.2.2 Cold water pollution

Cold water pollution is the degradation of water quality in a river downstream from a reservoir by means of artificially lowering the temperature below what is considered the natural thermal range. It is a common problem worldwide in many rivers downstream of large dams (Ryan 2001, Preece and Jones 2002, Rutherford et al. 2009). The phenomenon typically occurs below large dams with low level off-takes during periods of thermal stratification (Lehmkuhl 1972, Ward 1974, Preece and Jones 2002, Rutherford et al. 2009). Releases from the hypolimnion of a stratified lake can be 8-12°C lower than expected temperatures in summer, resulting in the reduction of water temperature downstream from the pre-dam thermal range (Lugg 1999, Burton 2000b, Sherman 2000, Preece and Jones 2002, Astles et al. 2003). The natural thermal regime of the downstream river is altered by hypolimnial releases in a variety of ways including the suppression of summer water temperatures, a reduction in annual and diel variation and range, and delayed seasonal peaks in temperature (Ryan 2001, Preece and Jones 2002).

The extent and magnitude of cold water pollution is influenced by the size of the reservoir and depth of release from the water column (Ryan 2001). Larger dam size and residence level often facilitate stronger thermal stratification events, and raise the thermocline further from the outlet (Preece and Jones 2002, Sherman 2005). Lower positioning of the release valve further increases the chance of water being drawn from the hypolimnion of a stratified reservoir (Ryan 2001). Many of the major dams in Australia have been constructed with low level or bottom off-takes (Camargo and De Jalon 1995, Sherman 2000). As a result, large dams with low-level off-takes are likely to release water from well below the thermocline, in the cold depths of the hypolimnion (Ryan 2001, Ryan and Preece 2003).

The significance of cold water pollution is also influenced by the magnitude of releases to the downstream river channel. A larger release volume requires more time to respond to ambient air temperatures in order to resemble natural water temperatures (Edinger et al. 1968).

2.2.3 Thermal recovery

The distance that the thermal effects of a dam persist downstream is termed *thermal recovery*, and is dependent on a range of factors including the volume and temperature of discharge from a dam, size of the reservoir, and relative contribution of unregulated tributaries (Preece and Jones 2002, Rutherford et al. 2009). The distance required for the water temperature of a river to return to naturally expected temperatures can be affected by discharge volume from a dam. Large waterbodies are exceptionally stable to thermal variation due to the high specific heat capacity of water (Gordon et al. 2004, Sherman 2005). The water temperature of high discharges is less responsive than low discharges to the surrounding climate because the greater volume of water (Gu et al. 1998). Furthermore, greater release volume results in a higher velocity, thus increasing distance travelled per unit time (Preece and Jones 2002). Therefore, a longer distance is required for the in-stream thermal properties to recover to within natural ranges. A combination of these factors results in low temperatures observed at the dam outlet persisting further downstream.

The contribution of tributaries downstream of a dam can reduce the distance of thermal recovery. The volume and temperature of an inflow from a tributary into a river influences the rate of recovery by contributing warmer water to the stream, raising temperatures closer to the natural range (Rutherford et al. 2009). Preece and Jones (2002) found that thermal recovery below Hume Dam (a large dam located on the Mitta Mitta River) was greatly influenced by the relative contribution of two unregulated tributaries. The effect was greatest in spring and early summer when tributary discharge was 50% that of Hume Dam compared to summer when the tributary discharge dropped to 10%.

The thermal effects of dams, specifically CWP, are well documented in the Murray-Darling Basin, NSW. Harris (1997) developed a longitudinal thermal profile of the Macquarie River, concluding that over 300 km of river was seriously affected by CWP below Burrendong Dam (defined as 5°C below predicated natural temperatures). Preece and Jones (2002) found temperature modification associated with Keepit Dam on the Namoi River to persist up to 100 km downstream, and temperature suppression on the Murray River downstream of Hume Dam has been found to persist for 200 km (Walker 1980).

2.3 Nutrient Dynamics

2.3.1 Internal reservoir dynamics

Nutrients are introduced to reservoirs from upstream river stretches through inflows, a process often referred to as external nutrient loading (Burger et al. 2008). The nutrient content and water quality of a river is dependent upon stream sediment transport, parent soil type, land use, stream velocity and rate of erosion, and vary across regions (Threlkeld 1990). Nutrients are transported into lakes from inflowing rivers, where the rapid decrease in velocity of water movement leads to the sedimentation of particles (Kunz et al. 2011a, Kunz et al. 2011b). Sedimentation of biotic and abiotic particles in a mixed lake results in the retention of phosphorus and nitrogen bound to sediments (Kunz et al. 2011b). Subsequently, aquatic environments downstream of a reservoir experience reduced nutrient concentrations during periods of reservoir mixing (Kunz et al. 2011a, Kunz et al. 2011b).

Internal nutrient loads contribute extensively to the nutrient content of the water column within a stratified reservoir, and is influenced by external nutrient loads (Burger et al. 2008). Internal loading occurs through the release of nutrients from sediments, and is dependent upon oxygen dynamics and the sedimentation history of an impoundment (Threlkeld 1990). The development of an oxycline commonly coincides with thermal stratification of a reservoir, resulting from a combination of factors including the isolation of the hypolimnion from the atmosphere and respiration of bacteria in the process of decomposing biotic material below the thermocline (Blumberg and Ditoro 1990). Anoxic conditions can lower the redox potential at the sediment-water interface, creating an environment favourable to the release of sediment-bound nutrients and ions, such as phosphorus, nitrogen and iron (Mortimer 1941, 1971, Amirbahman et al. 2003, Beutel 2006, Baldwin and Williams 2007, Baldwin et al. 2010). The dissolved oxygen concentration of the water column is required to reach below 2 mg/L at the sediment-water interface before a significant change in chemistry in the overlying waters is likely to be observed (Mortimer 1971). Therefore, during periods of stratification, the hypolimnion is often nutrient-rich, whilst the epilimnion generally becomes nutrient-poor. When the water body then undergoes mixing during the cooler months, the layers combine, often leading to eutrophication (Burger et al. 2008).

2.3.2 Downstream of a reservoir

Reservoirs form physical barriers to the transport of nutrients through the sedimentation of biotic and abiotic particles (Kunz et al. 2011b). Subsequently, the downstream river stretches receive less essential nutrients. During periods of thermal stratification, reservoirs with low-level outlets can release anoxic water with high concentrations of nutrients and metals and low silica (Conley et al. 2000, Marshall et al. 2006). Thus dams can alter the nutrient flux downstream, acting as a sink during cool seasons of lake mixing, and a source for nitrogen, phosphorus and iron during stratification in warm months.

2.4 Phytoplankton Dynamics

2.4.1 Within reservoir dynamics

The seasonal succession of phytoplankton is influenced by the stratification patterns of a reservoir, typically following a pattern of domination of small centric diatoms in spring, green algae and large diatoms in early summer, followed by dinoflagellates then cyanobacteria in mid-late summer, and the take-over of diatoms again in autumn (Sommer et al. 1986). The currents present during the mixing period of the water body support the growth and persistence of fast growing phytoplankton such as diatoms, which tend to sink in still environments (Pinckney et al. 1998, Ptacnik et al. 2003). Green algae dominate in early summer before cyanobacteria take over the nutrient-poor epilimnion (Lopes et al. 2007). The still and often clear waters of lentic systems during periods of stratification select for species that can actively regulate their position in the water column to receive more light (Mitrovic et al. 2001). For example, some species of cyanobacteria can control cell buoyancy through the use of gas vesicles and mucilage content while some other phytoplankton can regulate position through active motility permitted with the use of flagellum (Fraisse et al. 2013). These features can allow cyanobacteria to dominate the phytoplankton community in summer (Huisman et al. 2005), though these species often do not compete well when under conditions typical of lotic environments as they have slow growth rates (Kohler 1994).

2.4.2 Downstream of a reservoir

Lotic and lentic systems differ in environmental conditions, thus supporting differing phytoplankton species. Species with large, round or elongated shaped cells and faster growth rates, like diatoms (Kohler 1994), are favoured by the conditions within lotic systems of typically higher velocity, turbulent flows and varying light availability (Wehr and Descy 1998, Coles and Jones 2000). When phytoplankton favoured by lentic systems (usually slower growing species often capable of regulating their position in the water column) are introduced to lotic systems through dam releases, they often do not compete well during periods of high flow (Kohler 1994).

Dam releases may also dictate the phytoplankton community composition in the downstream river as lentic populations are released from the reservoir (Coutant 1963). As a result of this, the community structure and longitudinal distribution of phytoplankton within rivers is altered (Coutant 1963, Petts 1985, Zalocar de Domitrovic et al. 2007).

2.5 Downstream impacts on fish

There is very little in the literature on the effect of an altered riverine thermal regime on biota. Of the information available, there is much focus on the effects on native and invasive fish species. Suppressed water temperatures downstream of dams have negative implications for native fish species. Fish are cold blooded organisms, meaning ambient temperatures dictate their body temperature. They require a certain temperature range for successful growth, development and reproduction (Todd et al. 2005). Consequently, temperatures that are below these natural ranges result in slowed metabolism and growth of organisms as well as reduced success rate of reproduction, posing a serious threat to the viability of native species. The reduction of stream temperatures as a result of low level dam releases has been shown to directly negatively impact on native fish populations (Harris 1997, Astles et al. 2003, Sherman et al. 2007, Donaldson et al. 2008, Lyon et al. 2008). Astles et al. (2003) investigated the effects of a cold river channel on growth rate and survival of juvenile fish. Groups exposed to water within the natural range grew significantly more than those exposed to temperatures of 10°C below natural ranges. The group exposed to the cold temperatures also exhibited a lower survival rate. Todd et al. (2005) found the viability of a Murray cod population within the Mitta Mitta River to be threatened significantly by the unnaturally cold releases from Dartmouth Dam. Eggs and larva exposed to temperatures below 13°C exhibited a 0% survival rate. These studies illustrate the risks to the viability of native fish species exposed to CWP below large dams.

Cold shock in aquatic biota is also a principal concern for the health of ecosystems exposed to cold water dam releases. It occurs as a result of sudden temperature reduction often associated with the sudden change of release depth, often via multi-level off-take (Ryan and Preece 2003). The effect of cold shock on fish is well documented (Astles et al. 2003, Ryan and Preece 2003, Donaldson et al. 2008) with little information available on the effects on other aquatic organisms. Fish exhibit a range of symptoms, depending on severity of stress responses induced by temperature change ranging from uncoordinated swimming, to falling into a coma-like state (Donaldson et al. 2008). These behavioural changes make affected organisms susceptible to injury from scraping the substrate due to in stream currents or death from predation (Donaldson et al. 2008). Ryan and Preece (2003) identified 55 dams in NSW to have high potential for causing cold water shock.

2.6 Mitigation Options for cold water pollution

2.6.1 Destratification: Bubble plume

Artificial thermal mixing via bubble plumes is a common alternative to the multi-level outlet. The idea is based on mixing a stratified lake either in full, or partially near the off-take (Fernandez et al. 2012). One or more air compressors pump air to create plumes from along the base of the dam, entraining water from the hypolimnion to the surface water (Sherman 2000). Through subsequent mixing DO levels are homogenised, with anoxic hypolimnial water mixed with oxygenated surface water. The success of air plumes is dependent upon strength of stratification and air flow rate. A lake experiencing a strong stratification period requires a more powerful air flow rate (Sahoo and Luketina 2003). In cases of entire lake mixing, this method can reduce the internal nutrient loading from sediments (Sherman 2000).

Sherman (2000) estimates the cost of installation to be approximately AUD$1.5 million for construction and installation into Burrendong Dam, with operating costs to start from around AUD$75,000 - $100,000 per annum. Unfortunately large dams may not be able to be destratified. Costs depend on the size and depth of the dam and the reservoir, and the required compressor size.

2.6.2 Partial destratification: Surface pump

Surface pumps are used to mitigate cold water pollution downstream of large dams through mechanically pumping warmer, oxygenated surface waters down to the outlet depth. They are generally only used to locally destratify reservoirs in this manner, rather than destratify the whole water body. The capital cost for the pump system in Douglas Dam cost approximately AUD$2.5 million, with mechanical equipment and floatation platforms accounting for AUD$1.5million and monthly operational costs estimated at $5000 (Sherman 2000).

2.6.3 Selective withdrawal: Multiple level off-takes

Multi-level outlet structures are considered one of the most common techniques for combating cold water pollution below large reservoirs (Sherman 2000) as they allow for the release of water from multiple strata of a water column. Multiple openings are located along the structure between the surface and the bottom of the reservoir. Outlet levels are closed with bulkheads which can be replaced with trash racks to allow water releases from desired depths. A higher number of outlets along the structure allow for more regulation of withdrawal depths and temperatures.

Installation of multi-level outlet structures has been seen to cost up to USD$80million (Shasta Dam) and estimated costs of retro-fitting Burrendong Dam reach up to AUD$25 million in 2000 (Sherman 2000). Costs of installation are substantially higher for larger dams, taking in considerations actual material costs and construction. Another factor is the number of outlets for withdrawal; a higher number of outlets results in a large increase in costs (Sherman 2000).

2.6.4 Selective withdrawal: Thermal curtain technology

The concept of submerged curtains has been exploited a number of times to solve problems related to the water quality of dams and downstream releases. Curtains have been utilised in Japan to control eutrophication and resulting algal blooms within the Terauchi Dam Reservoir (Asaeda et al. 1996, Priyantha et al. 1997, Asaeda et al. 2001). Two curtains were extended across the surface layer and to a depth of 5m to encourage the nutrient-rich inflow through bottom layers of the reservoir to the outlet, preventing these nutrients from supplying a cyanobacterial bloom. A curtain of similar design was installed in the Saidenbach Reservoir, Germany for the purpose of phosphorus elimination (Paul et al. 1998). Although in both instances, deployment of the curtains were not intended for management of discharge temperature, they were deemed successful for their purpose and provide an insight into how curtains can be used to manipulate the hydrological dynamics within a reservoir.

The concept has also been used in American dams to control the temperature of releases. Surface water releases from Lewiston and Whiskeytown Reservoirs, California were threatening the viability of downstream salmon species due to increased temperatures too warm for egg incubation and juvenile development and survival (Vermeyen 1997). Installation of the curtains was successful in allowing selective withdrawal from both reservoirs, reducing downstream temperatures by up to 15°C.

The curtain to be installed into Burrendong Dam is innovative as the design uses the concept of selectively drawing epilimnion water to the outlet with the use of a polypropylene structure; however the structure differs in its shape and size. The

Burrendong curtain has been attached to the off-take tower with the cylindrical shaped curtain extending to the surface layer, and height controlled using a chain – pulley system. Traditional thermal curtain design inhibits cold water accessing the low outlet by forming a long barrier around the outlet, with attachments to buoys to regulate height.

Chapter 3: Temperature

3.1 Introduction

Cold water pollution (CWP) is a serious problem in many Australian rivers downstream of large dams (Ryan 2001, Preece and Jones 2002, Rutherford et al. 2009) and other rivers around the world (Olden and Naiman 2010). CWP is caused by the release of cold, hypolimnial water from low level off-takes when the water body is thermally stratified. The natural thermal regime is altered by hypolimnial releases in a variety of ways including the suppression of summer water temperatures, a reduction in annual and diel variation and range, and delayed seasonal peaks in temperature (Ryan 2001, Preece and Jones 2002). Dams on rivers may also cause elevated winter temperatures (warm water pollution) (Cowx et al. 1987).

The distance downstream of a dam where river temperatures are impacted by CWP, termed the discontinuity distance (Ward and Stanford 1983) or thermal recovery distance (Preece and Jones 2002), can vary depending on the size of the reservoir, residence level in the reservoir and release volumes. Dams with a large depth and water storage capacity can produce CWP effects that extend for hundreds of kilometres downstream of the dam wall (Lugg 1999, Preece and Jones 2002). Historically, there have been varying accounts for the thermal recovery distance from CWP. Reports range from relatively short distances (e.g. 8.5 km, Lehmkuhl, 1972), to long stretches of river, as far as 300 km below the dam (Ward 1974, Harris 1997, Preece and Jones 2002).

There are risks to biota inhabiting reaches downstream of dams affected by cold water pollution. Invertebrates and fish are cold-blooded (ectothermic) organisms, and ambient temperatures dictate their body temperature. Natural thermal niches are important for the survival of these organisms, as they require a specific temperature range for successful growth, development and reproduction (Lehmkuhl 1972, Clarkson and Childs 2000, Astles et al. 2003, Todd et al. 2005). Reduced numbers of macroinvertebrates living in rivers below large reservoirs have been attributed to thermal suppression (Lehmkuhl 1972, Ward 1974). Furthermore, unnaturally low temperatures result in reduced metabolism and growth of fish as well as reduced reproductive success, posing a serious threat to the viability of native species exposed to temperatures below their niche (Clarkson and Childs 2000, Astles et al. 2003).

Minimum temperatures required for successful spawning have been previously determined for three species of fish present in the Murray Darling Basin, Australia: Murray Cod (*Maccullochella peelii peelii*), golden perch (*Macquaria ambigua*) and silver perch (*Bidyanis bidyanis*). Threshold temperatures required to initiate spawning for these species vary from 23°C for Murray cod (Lake 1967, Rowland 1983), 23-26°C for golden perch (Lake 1967, Merrick and Schmida 1984) and a temperature rise from 16.4°C and 20.6°C for silver perch (Lake 1967), during spring and summer months. Reabsorption of gonads, reduced growth rates in juveniles, and lowered development and survival rates of eggs and larvae have been reported when environmental conditions are inadequate (Lake 1967, Astles et al. 2003, Todd et al. 2005). Sherman et al. (2007) modelled growth in Murray Cod populations affected by CWP and predicted that the population would grow with increased water temperature, which he suggested could be achieved by using surface releases instead of hypolimnial releases. In addition, the proportion of native fish species relative to invasive species is often reduced in disturbed waters near the outlet of large reservoirs (Driver et al. 1997, Astles et al. 2003).

An affordable, effective mitigation method for CWP is desirable given the extensive ecological impacts that occur along stretches of affected rivers. Sherman (2000) suggested two options for mitigation of CWP: 1) reservoir destratification, or 2) selective withdrawal of warm water from a chosen depth. Destratification works by inducing circulation within a reservoir to stop stratification and raise the temperature of bottom waters. One method is via a bubble plume, whereby air is released from the hypolimnion of the stratified reservoir, resulting in mixing of the layers (Miles and West 2011, Fernandez et al. 2012). Destratification methods often incur excessive running costs (mainly electricity consumption) for large reservoirs and are unable to keep the water body fully mixed. Another method to reduce CWP is to selectively withdraw water via a multiple level off-take (MLO). MLOs are less energy-intensive than active destratification options like bubble plumes and work by channelling warmer surface water to the outlet. However, they can be time consuming to manage, often taking at least one working day to change the off-take depth (Sherman 2000).

Burrendong Dam is located on the Macquarie River, central-western New South Wales and has a storage capacity of 1,200 gigalitres. Thermal stratification occurs frequently in summer due to its depth. High-volume summer releases for the agricultural irrigation season are drawn from the hypolimnion via a low level off-take, producing CWP. Because of this, the Macquarie River has been targeted in studies assessing the extent and magnitude of cold water pollution. Harris (1997) focused on the presence and magnitude of CWP below Burrendong Dam, measuring the longitudinal thermal profile of the Macquarie River in western NSW. He concluded that more than 300 km of river was affected by CWP. Harris speculated an adverse effect on native fish species and the proliferation of invasive species as a result of thermal depression. Preece and Wales (2004) identified Burrendong Dam as a 'high priority' reservoir for CWP, ranking it 5th out of the 304 dam structures in NSW. In a scoping study, Sherman (2000) outlined options for CWP mitigation at Burrendong Dam including retrofitting the dam with a MLO which was an expensive undertaking and also suggested installing a submerged thermal curtain as a cheaper alternative.

Utilising these ideas, a thermal-curtain-style temperature control structure was designed and installed at Burrendong Dam in 2014. The thermal curtain began operation in May 2014. The structure is designed to direct surface water to the bottom release outlet during periods of thermal stratification, increasing downstream temperatures. The structure cost AUD$4 million to install, making it a cost-effective alternative to a MLO (approximately AUD$37 million).

It is important to monitor the effectiveness of this new technology in improving water temperature outcomes to assess its functionality and applicability for installation into other large dams that cause CWP, and to provide operational data to improve management of the curtain.

Three specific hypotheses were developed:

1. Historical analysis of temperature data will indicate the incidence of CWP immediately below the dam during the warmest months.
2. Use of the thermal curtain will show increases in temperature which better reflect predicted natural temperatures.
3. There will be an improvement in thermal recovery distance along the length of the Macquarie River with the use of the thermal curtain.

3.2 Study area: The Macquarie River and Burrendong Dam

The Macquarie River is located within the Murray-Darling Basin. The river rises in the Great Dividing Range near Oberon and flows for 560 km in a north-westerly direction to the Macquarie Marshes before merging with the Barwon River. The Macquarie-Bogan catchment covers an area of over 74,000 km^2 (Green et al. 2011). Two large reservoirs exist within the catchment: Windermere Dam and Burrendong Dam. The former reservoir is located on the Cudgegong River near Rylestone on the central slopes of NSW while Burrendong Dam receives inflows from the Cudgegong River downstream of Windermere Dam and from the Macquarie River (Green et al. 2011, Kunz et al. 2011a).

The Bell, Little and Talbragar Rivers, and Coolbaggie and Ewenmar Creeks constitute the main unregulated tributaries of the Macquarie River below Burrendong Dam. The Bell and Little Rivers enter the Macquarie River upstream of Dubbo, and the Talbragar River just downstream of Dubbo. Coolbaggie Creek joins the Macquarie River upstream of Narromine and Ewenmar Creek joins the river downstream of Warren.

Mean annual rainfall in the catchment follows a decreasing southeast-northwest gradient, ranging from 1,200 mm in the southeast to 300 mm in the northwest (Green et al. 2011).

3.2.1 Study reservoir: Lake Burrendong

Burrendong Dam was completed in 1967 at a height of 76 m and a storage capacity of 6,345 GL (Australian National Committee on Large Dams 1982). At full supply level, the reservoir reaches an elevation of 344 m above sea level (a.s.l.), while the outlet sits 31.43 m below the water surface. The depth of the water column in the reservoir reaches 57 m at full supply level (State Water Corporation 2009). The dam was constructed to secure water for irrigated agriculture, flood mitigation, to supply household water needs and to secure environmental flows to the Macquarie Marshes (Australian National Committee on Large Dams 1982, Green et al. 2011).

Burrendong Dam regulates water discharge through a fixed low-level outlet. The outlet is located at 312.57 m a.s.l. The largest discharge typically occurs in the summer months to meet demands of agricultural water supply. The largest use for water in the catchment is for cotton production (Green et al. 2011).

3.2.2 The thermal curtain

AMOG Consulting developed the concept design and Geotechnical Engineering completed the construction of the structure. The thermal curtain is a flexible, cylindrical, reinforced polypropylene fabric curtain attached around the base of the outlet. It extends up to the surface-mixed water layer. Rigid support rings at intervals along the length of the curtain and at the opening provide structural stability. A chain-and-pulley mechanism allows the curtain's height below the water surface to be controlled via computer programming. The depth of the curtain was set to 7 m below the surface for the first year of its use, which began on the 7th May, 2014.

A photo taken in 1962 during the construction of Burrendong Dam has been used to illustrate the off-take structure as it appears in its submerged state (Fig. 2). An overlay of a simplified thermal curtain has been added to illustrate the design. Figure 3 provides more technical detail of the thermal curtain design. The in-depth schematic shows how the thermal curtain will sit around the intake tower and direct water to the low-level off-take.

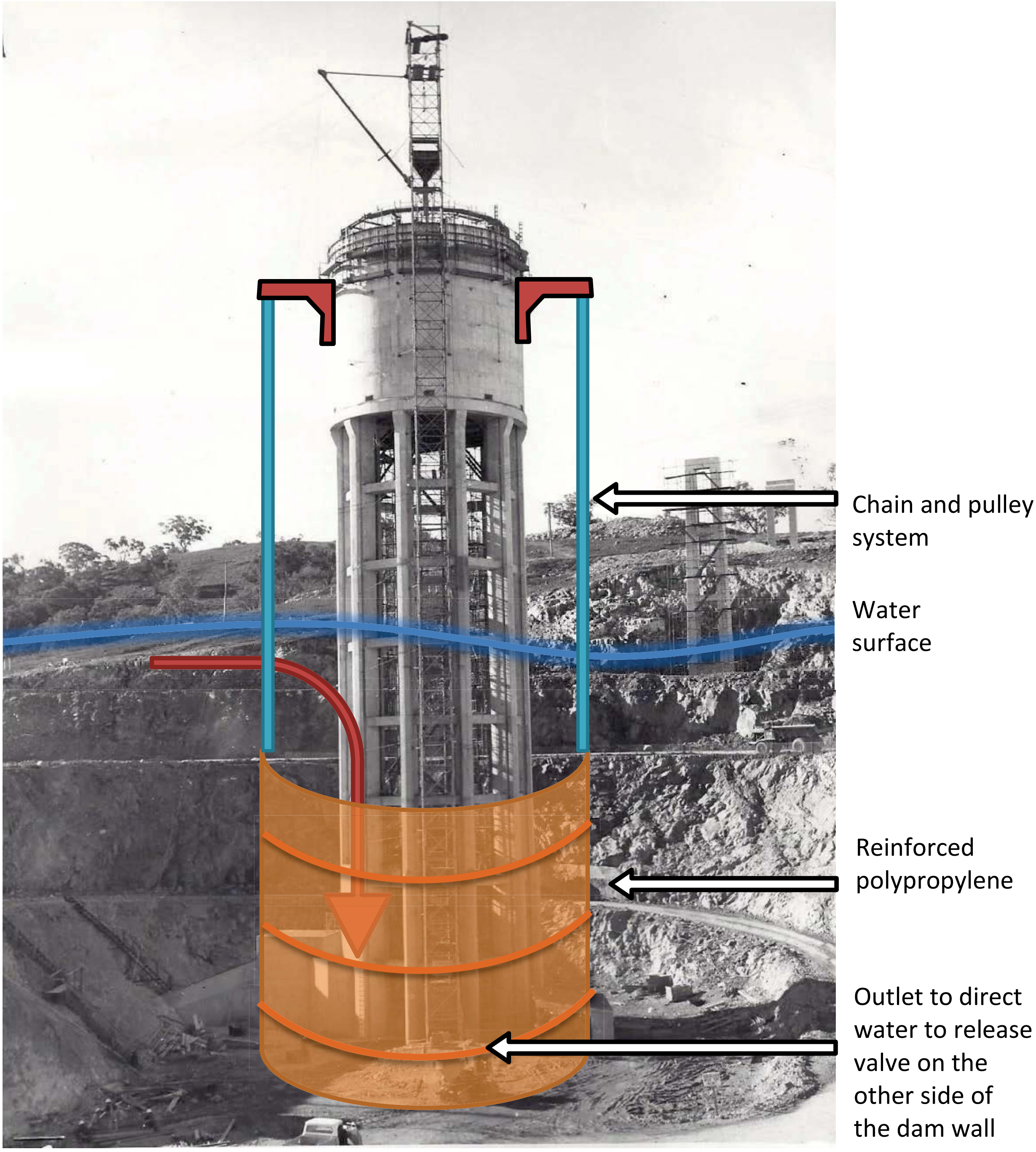


Figure 2: Historical photo of the Burrendong Dam off-take tower overlaid with a simplified diagram of the thermal curtain. Original photo source: Oxley Museum, Wellington NSW.

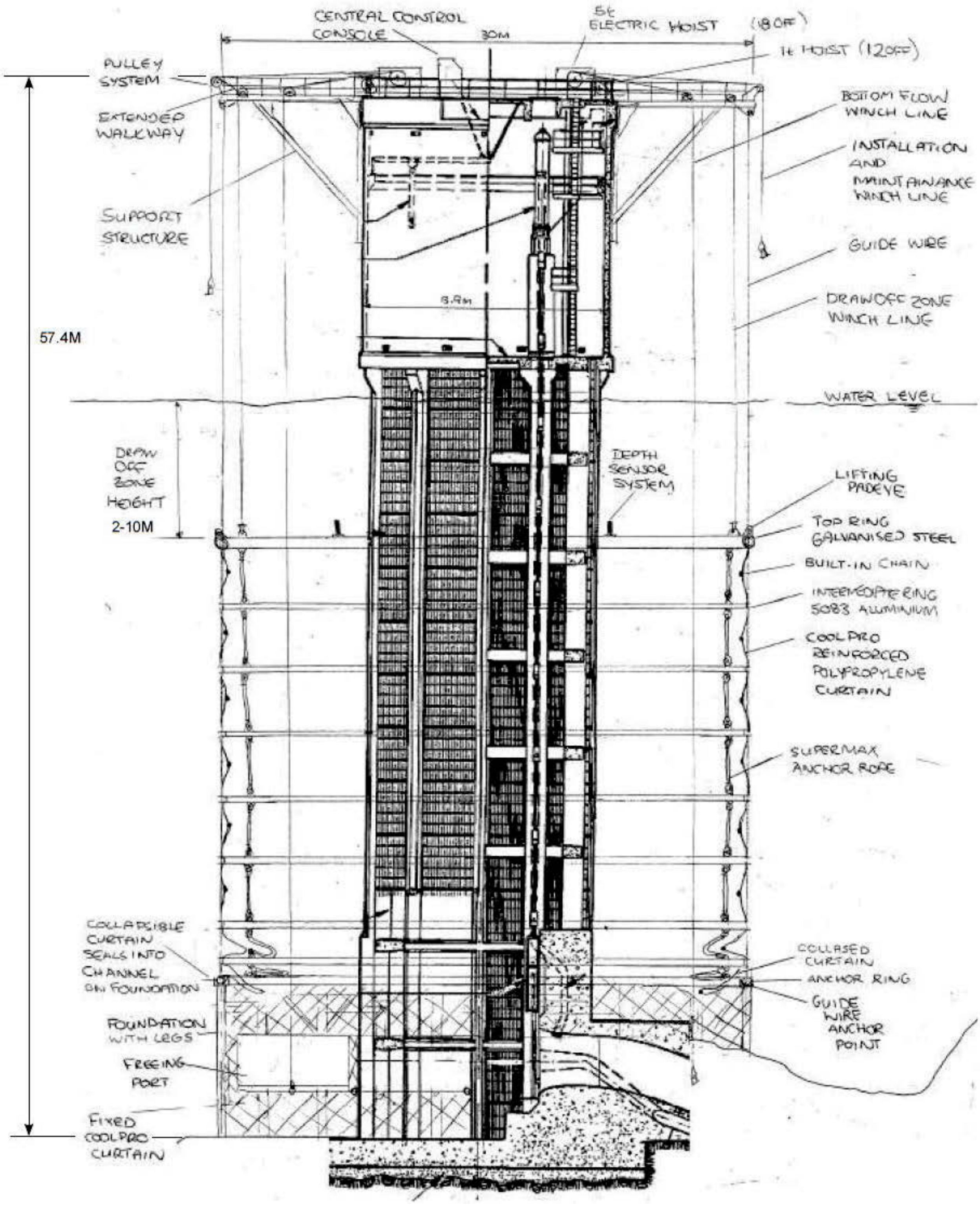


Figure 3: A detailed schematic of the thermal curtain design by AMOG. Source: AMOG Consulting (2010).

3.2.3 Study sites

Twelve sites were located at intervals along the Macquarie River downstream of Burrendong Dam, with the first site located at the outlet immediately below the dam wall and the final site at Warren weir (312 km downstream). Three sites were chosen upstream of the dam on the Macquarie River to be used as natural, unaltered references compared to downstream sites. The sites were chosen based on established monitoring sites with DPI Water. Though the upstream sites are a geographically long distance from the reservoir, the temperature data should provide a conservative natural reference, as temperature generally increases longitudinally with distance downstream. The Cudgegong River, although a significant inflow to Burrendong Dam, is thermally altered by Windamere Dam. As such, the river is not representative of natural conditions, and so this tributary was not used for thermal monitoring. Thermal depth profiles were recorded across three sites in the reservoir, near the off-take tower (Fig. 4).

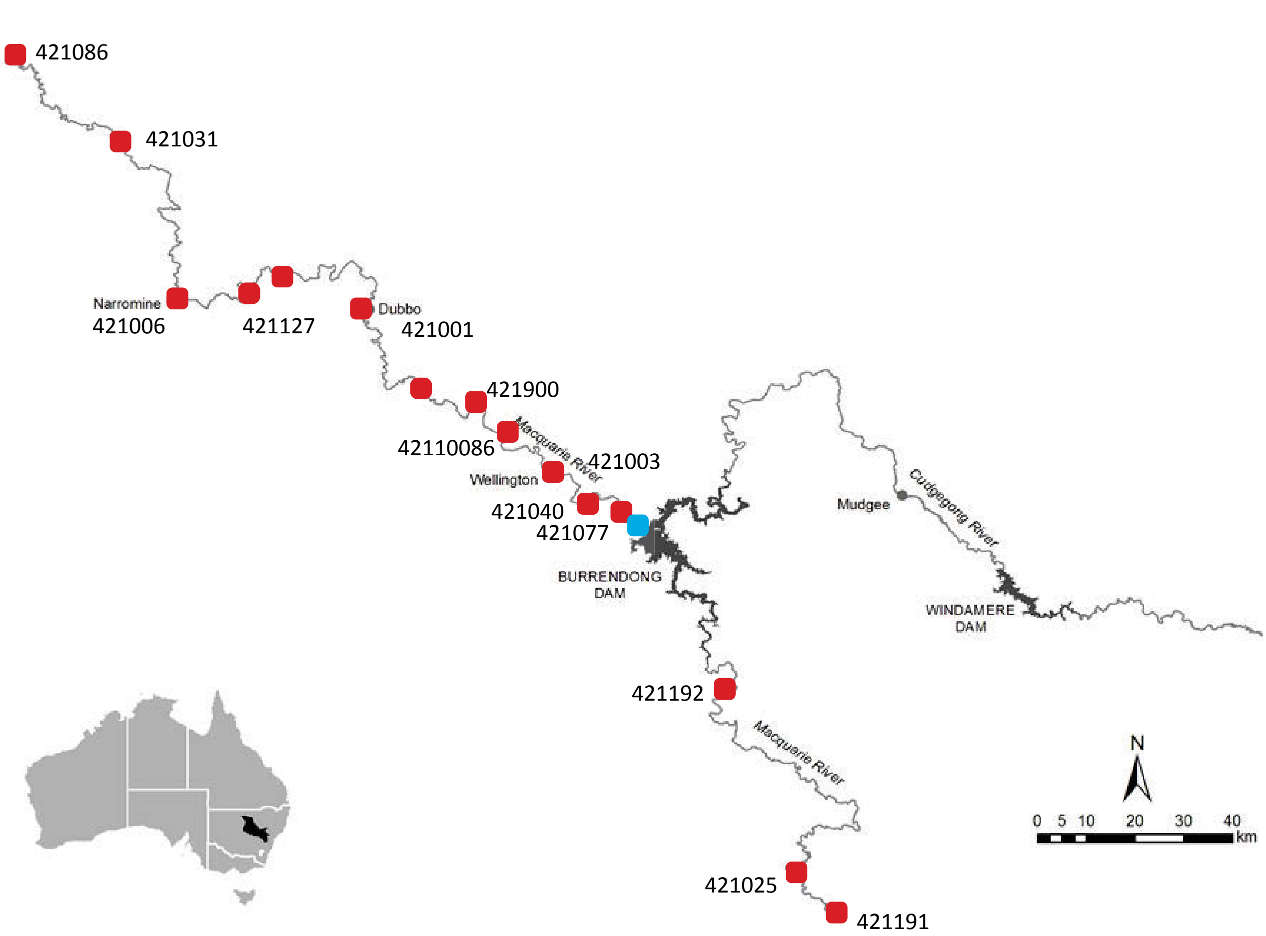


Figure 4: Map of the Macquarie and Cudgegong Rivers. Sites where river temperatures were monitored are indicated with red squares. The blue dot indicates the monitoring location within the reservoir. Site details are given in Table 1.

3.3 Methods

3.3.1 Data collection and compilation

One reservoir site and fifteen river sites were established to form a longitudinal record of the Macquarie River water temperatures, with one station located within the reservoir near the dam wall. Records for most stations commenced in 2013 with the launch of eleven Hobo loggers. The remaining four sites have been logging river water temperatures since 1998 (Table 1).

Reservoir Sites

A thermistor chain attached to a pontoon located approximately 750 m from the off-take tower was commissioned by Water NSW and recorded temperatures three times a day at depth intervals of 0.5 m from 16/07/2012 to 10/07/2013 (Fig. 5, TC1). Monthly depth profiles of the reservoir were recorded from a boat at a buoy located approximately 400 m from the off-take tower (TC2, Fig. 5) from 05/06/2013 to 7/01/2015. A thermistor chain, comprised of Hobo loggers and Tidbit loggers (OneTemp Pty Ltd 2013), was deployed from the buoy, logging data from 8/07/2013 to 31/05/2014. Temperatures were logged at 30 minute intervals at depth increments of 2 m to a depth of 24 m, then at increments of 4 m to a depth of 52 m. Two additional depth profilers were commissioned by Water NSW to record depth data from the 5/06/2014, with one positioned at 100 m from the off-take tower (Fig. 5, TP1), and the other within the thermal curtain located at the off-take tower.

Long-term thermal depth profiles from the two thermistor chains and the thermal profiler were pooled to form a three year time-series. The depth profiler (TP1) was dysfunctional for unforeseen periods of time, resulting in substantial data gaps.

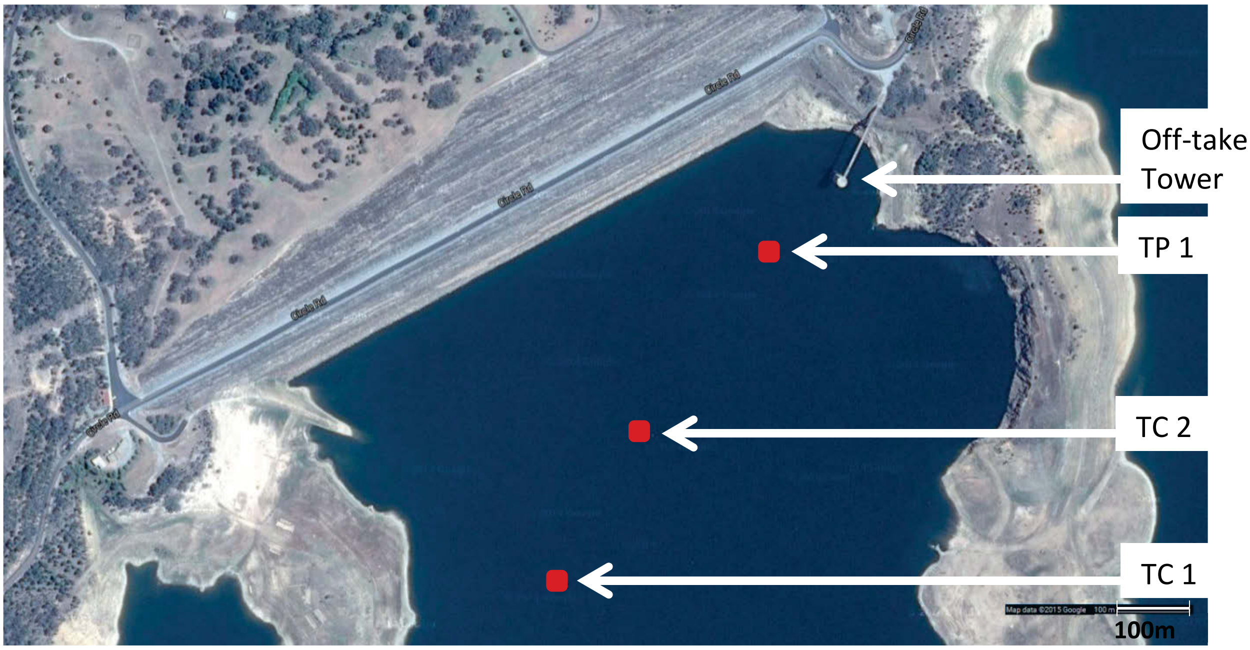


Figure 5: Aerial view of Burrendong Dam wall with arrows indicating the position of the off-take tower), thermistor chain 1 (TC1), thermistor chain 2 (TC2) and the thermal profiler (TP1). Source of image: Google Maps

Macquarie River sites

Historical water temperature and river flow data was sourced and analysed from the DPI Water Hydrometric database (Hydstra, Kisters Ltd.). Four sites have gauging stations installed with equipment designed to record long-term temperature and river flow (Table 1, Fig. 6a). Temperature loggers (OneTemp Pty Ltd 2013) were set up at gauging stations where temperature was not previously recorded and data was downloaded from the loggers approximately every 3 months using HOBO software. At each site, the buoyant loggers were attached to a chain using cable ties and floated in the water column at approximately 20 cm above the river bed, recording temperature at 30 minute intervals (Fig. 6b). Two loggers at each site provided insurance against data loss from damage incurred in the stream or a dysfunctional logger. Where possible, loggers were placed in flowing water of the river.

a. b.

Figure 6: An example of a gauging station shed at Downstream Burrendong gauging station (a) and the set-up of the Hobo loggers in river sites (b).

Table 1: Temperature data availability. This is inclusive of data recorded on Hobo loggers and historical data recorded at gauging stations through DPI Water monitoring programs (Hydstra). Grey squares indicate a period of six months with >5 months of data available.

Site Name	**Site Code**	**Data Record**	**Distance from dam (km)**	**Data availability** 1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015
Sites on the Macquarie River Upstream of Burrendong Dam																				
Yarracoona	421191	Hobo	108.5																	
Bruinbun	421025	Gauge	104.0																	
Long Point	421192	Gauge	73.0																	
Sites on the Macquarie River Downstream of Burrendong Dam																				
Burrendong Outlet	421077	Hobo	0.5																	
Downstream Burrendong	421040	Gauge	7.4																	
Wellington	421003	Hobo	33.0																	
Ponto Falls	42110086	Hobo	55.0																	
Wollombi	421900	Hobo	76.5																	
Geurie Bridge		Hobo	93.0																	
Dubbo	421001	Gauge	115.2																	
Rawsonville Bridge		Hobo	152.0																	
Baroona	421127	Hobo	162.5																	
Narromine	421006	Hobo	187.0																	
Gin Gin	421031	Hobo	254.5																	
Warren Weir	421086	Hobo	312.0																	

3.3.2 Data analysis

Three gauging stations (GS) upstream of Lake Burrendong on the Macquarie River: Yarracoona, Bruinbun, and Long Point, were selected as natural reference sites for comparison with Burrendong Dam releases. Releases were measured at the Outlet and the Downstream Burrendong GS, located several kilometres below the dam (Table 1). Temperature records for Bruinbun GS and Downstream Burrendong GS were available from 2008 onwards. Historical comparisons of water temperatures upstream and downstream of Burrendong Dam were made using this data.

3.3.3 Historical cold water pollution

Historical data accessed from the Hydstra database was used to analyse long-term thermal patterns along the Macquarie River. The data were downloaded as daily mean temperature time-series and real-time (data observations recorded at hourly intervals) records. Daily means were graphed from January 2008 to January 2015 to provide visual comparisons of the thermal regime between the upstream site, Bruinbun GS, and the downstream site, Downstream Burrendong GS.

Seasonal dynamics of the vertical temperature profile in the reservoir

To further improve understanding of the thermal dynamics within Burrendong Dam, monthly depth profiles of water temperatures were used to create a heat map. The map displays thermal stratification patterns as a two dimensional plot with depth on the y-axis, time along the x-axis and temperatures as colour gradients.

A plot was constructed using data from the time-series of temperature depth profiles for Lake Burrendong near the dam wall. Four depths (2 m, 10 m, 14 m, and 20 m) were chosen for plotting. These were displayed as time-series plots alongside temperature data recorded at the Downstream Burrendong gauging station and the upstream site, Yarracoona. Yarracoona was selected for this graph because the temperature data for this site was more up-to-date than Bruinbun GS (Table 1). The data records for Bruinbun and Long Point were incomplete for 2015 and so could not be used for 2015 comparisons (Table 1).

Effect of residence level and discharge volume on downstream temperature

Reservoir residence level data was obtained from the Water NSW website (http://realtimedata.water.nsw.gov.au). Residence levels were converted from elevations a.s.l. to heights above the outlet by subtracting the elevation of the outlet depth (312.57 m a.s.l.) from the residence level (expressed as m.a.s.l). This was done to show the relative distance of the outlet from the water surface on a time-series from July 2012 to April 2015. An in-depth investigation into the relationship between residence level, the distance of the thermocline above the low-level off-take, and the severity of downstream CWP was undertaken. Data on historical residence levels were used to identify years when the volume of water stored in the reservoir was similar to 2014 when the thermal curtain became operational. The years when the residence level was similar to 2014 were 2004, 2007 and 2013. Discharge from the storage was superimposed on the time-series plot of Downstream Burrendong GS temperatures to show the effect of changes in discharge on water temperature. The temperature time-series for 2004, the year with the most similar discharge pattern to 2014, was overlaid onto the plot to compare thermal regimes and potential improvements in water temperature brought about by surface releases in 2014.

Upstream versus downstream water temperatures

Temperatures for the upstream site, Yarracoona, were chosen as a reference condition for the natural, pre-dam temperatures downstream of the dam wall at Downstream Burrendong GS. This site was chosen because it had a complete temperature record from 2013 to 2015. The deviation of downstream temperatures from the natural regime was calculated by subtracting Yarracoona mean daily temperatures from Downstream Burrendong temperatures for the year prior to (2013) and after (2014) the point in time when thermal curtain became operational. Ideally, 2004 would have been included in comparisons of pre-thermal curtain use, but upstream data was unavailable for the summer of 2004 (Table 1). Monthly means were also calculated as an additional comparison of the monthly thermal trends prior to, and following, the use of the thermal curtain.

The role of discharge in the functionality of the thermal curtain

The potential role of reservoir discharge in modifying the effect of the curtain on the thermal regime of the Macquarie River was investigated. Discharge data was compared to the change in temperature inside the thermal curtain (surface temperature minus bottom water temperature within the curtain, ΔT) and the temperature recorded at the outlet over a period of 295 days.

The strength of thermal stratification was described by ΔT, where the larger the difference between the surface and bottom water temperatures within the thermal curtain, the greater the strength of thermal stratification.

ΔT was plotted as a time-series superimposed with discharge from Burrendong Dam and mean daily temperatures recorded at the outlet. Two peaks and one drop in outlet discharge were investigated further in relation to ΔT and outlet temperatures.

The outlet temperatures at these points were then plotted with temperatures observed upstream of the storage for the same period. Pearson's product-moment correlation was calculated for the discharge and ΔT values to test the strength of the linear relationship.

Diel range

Diel range, the thermal range within the 24 hour period of a day, was assessed for the 12 month periods of before and after the thermal curtain operation to further assess the effect of the curtain on the thermal regime of the downstream Macquarie River. Diel temperature range at the outlet was calculated from hourly data with daily minimum subtracted from daily maximum and was plotted as a time-series. This allowed a visual comparison of potential changes in the diel range after the release of surface waters with the thermal curtain. Monthly means with standard errors for diel thermal range at Yarracoona and Burrendong outlet were calculated and plotted as a time-series. The monthly means were used to show the overall trends of the diel range at the outlet compared with what was observed under assumed natural stream conditions.

Thermal recovery distance

The data collected by the Hobo temperature loggers were used for analysis of thermal recovery with distance downstream of the dam. There are two ways of viewing recovery: 1) by comparing water temperatures downstream of the dam to reference sites, and 2) by calculating recovery with distance using an asymptotic equation.

1) By relating the downstream temperatures with water temperatures at upstream reference sites, taking into account the variability in temperatures:

December water temperatures were recorded for both years at Yarracoona, but were available only for December 2013 at Bruinbun and Long Point. December was chosen as the historical temperature data indicates it is the month worst affected by CWP (Fig. 7). December was therefore chosen to assess the effectiveness of the thermal curtain in mitigating CWP during a period when the worst expected thermal effects of the dam are most likely to occur. Separate 95% confidence intervals were calculated for all upstream sites for 2013 and 2014 with available data for the December months. A zone delimited by the upper limit for Long Point and the lower limit for Bruinbun was highlighted in blue on the graph of longitudinal thermal recovery. These intervals were used as a broad reference band for downstream water temperatures, as temperatures within this range might be expected to represent naturally occurring temperatures downstream of the dam wall.

2) When the recovery curves plateau (i.e. mean December temperatures approach an asymptote, as estimated by the T_{max} parameter).

Recovery in surface water temperatures with distance downstream of the Burrendong Dam wall was expected to occur rapidly close to the dam before tapering off to an asymptotic temperature far from the dam. Mean December water temperatures at sites downstream of Burrendong Dam were modelled using the following negative exponential temperature recovery model:

$$T = T_0 + (T_{max} - T_0)(1 - e^{-kS})$$

where

T_0	*represents the water temperature at the outlet (distance of 0 km)*
T_{max}	*is the asymptotic temperature (above T_0)*
k	*is a rate constant (or shape parameter) controlling the rate of recovery with distance (S)*
S	*distance downstream from the dam wall (km)*

The parameters of the model were estimated by nonlinear weighted least-squares (Bates and Chambers 1992) using the nls function in R (Fox and Weisberg 2010). The nls function uses the Gauss-Newton algorithm to iteratively minimise a weighted residual sum of squares.

Both approaches (1) and (2) were used in this study to assess the thermal recovery of the Macquarie River downstream of the dam wall. The predicted temperatures with distance projected by method 2 were used to identify the distance of recovery according to the 95% confidence intervals calculated for upstream sites using method 1. This gave an estimate of the distance to the point of thermal recovery for both years. Data for December 2014 was unavailable for Geurie Bridge and Narromine, so these two sites were omitted from the 2014 recovery model.

Five sites with intact time-series (i.e. few or no missing values) were selected for further assessment of the impact of the thermal curtain on downstream river temperatures to support and compliment the above analyses. Short gaps in the time-series of less than a week were filled by linear interpolation with neighbouring values. The five chosen sites (Outlet, Wellington, Baroona, Gin Gin and Warren) spanned distances from immediately downstream of the dam wall to Warren, 312 km downstream of the dam. The periods from 1 July 2013 to 31 Dec 2013 (pre-curtain operation) and 1 July 2014 to 31 Dec 2014 (post-curtain operation) were chosen for comparison. This time of year captures much of the thermal depression which develops from spring into summer

A Kolmogorov-Smirnov two-sample test (Conover 1999) was used to compare the deviations in daily water temperatures on the same day of the year in 2013 and 2014 between sites.

Biological implications: Thermal effects on native fish

Lake (1967) and Merrick and Schmida (1984) give the minimum temperatures within a given time period from for the cue of spawning of three native fish species in the Macquarie River. These were overlayed on a time-series plot of the water temperatures at the outlet GS from the time period of August 2013 to June 2014, and August 2014 to May 2015. This was done to use a tool to denote just one possible effect of water temperature in the Macquarie River immediately below the dam wall on the biological constituents of the river.

Data in this chapter were formatted and graphed using Microsoft Excel 2010 and SigmaPlot 12.5 software. Statistical analyses were performed using R 3.1.2 (R Core Team 2015).

3.4 Results

3.4.1 Historical cold water pollution

There was a consistent difference in water temperature between Bruinbun and Downstream Burrendong GS over a period of 9 years (Fig. 7). It is clear that CWP is regularly occurring during the warm months of most years from September to March. CWP was particularly noticeable in the summers of 2008, 2010, 2011 and 2012 when maximum daily mean temperature differences between the two sites reached 9.4°C, 13.5°C, 11°C and 10°C, respectively. This is a strong indication that CWP has been a long-term problem on the Macquarie River below Burrendong Dam.

Warmer than natural water releases are another thermal effect resulting from dam discharges. Temperatures below the dam wall are regularly warmer than upstream temperatures during periods of reservoir mixing (cooler months) with recorded temperature differences of up to 8.1°C warmer occurring downstream of the dam (June 2012).

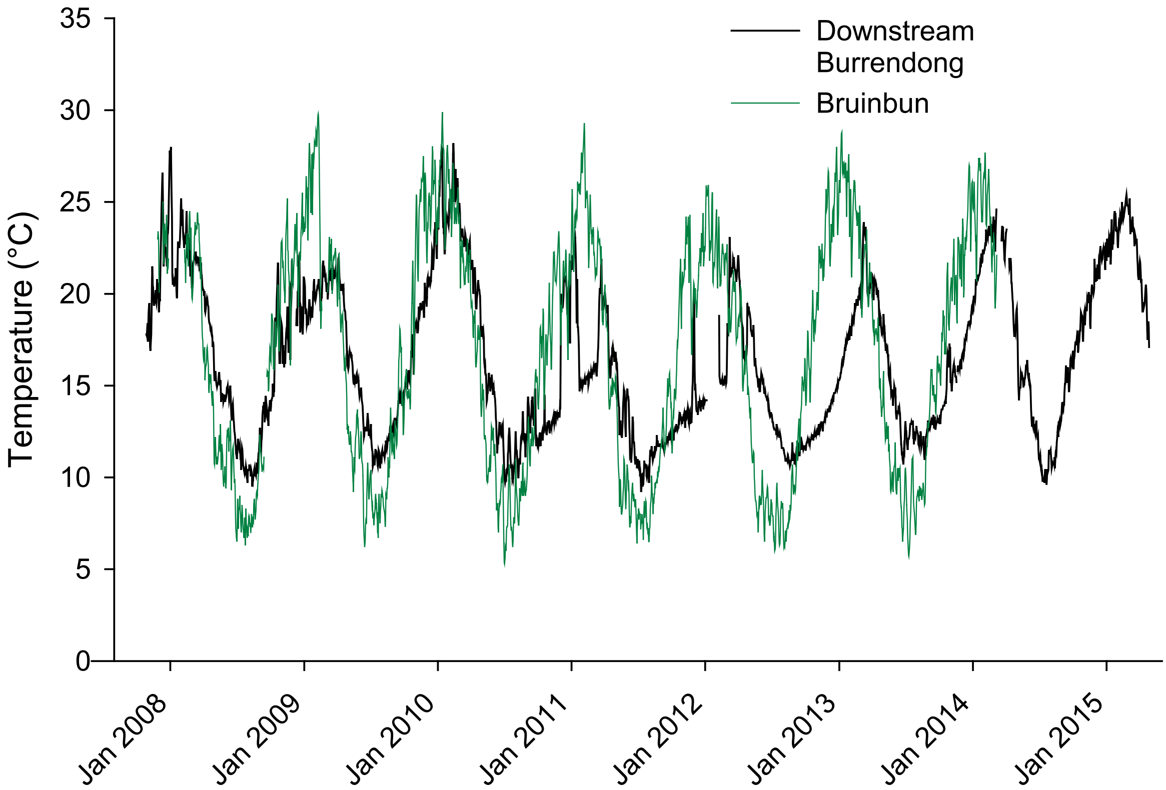


Figure 7: Thermal regime of the Macquarie River at upstream site Bruinbun (green) and downstream site Downstream Burrendong (black) over a period of 7 years.

3.4.2 Seasonal dynamics of the vertical temperature profile in the reservoir

To understand the cause of CWP below Burrendong Dam on the Macquarie River, the internal thermal dynamics of the reservoir were examined. A heat map displaying the thermal patterns of the water column demonstrates the seasonal cycle, the persistence of stratification and the subsequent mixing events within the reservoir (Fig. 8). Through the months of April to September, the water body was isothermal (mixed), but stratification developed from October and extended through to March. The development of the epilimnion and hypolimnion occurred annually with up to 13.5°C difference in temperature (January 2013, Fig. 9).

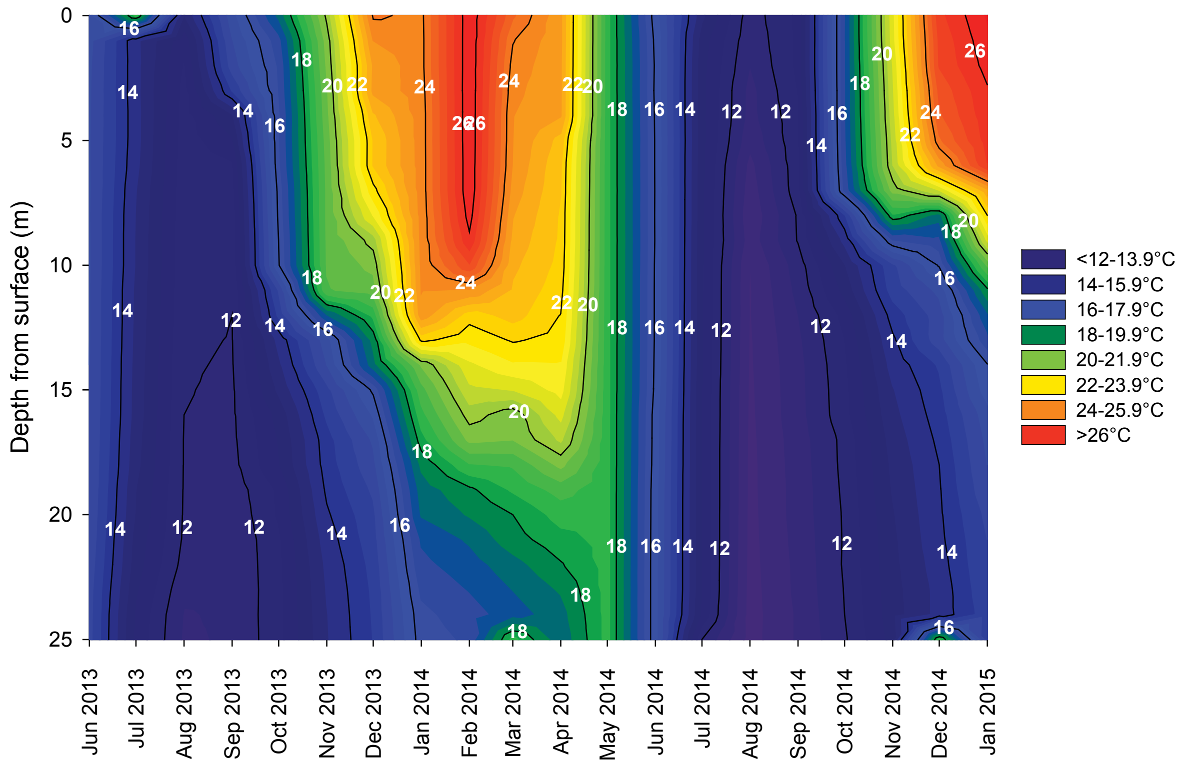


Figure 8: Heat map showing the thermal patterns within Burrendong Dam from June 2013 to January 2015. Thermal profiles are shown over time (x-axis) with depth (m) from the surface on the y-axis.

3.4.3 Effectiveness of the thermal curtain in reducing the magnitude of cold water pollution

A time-series comparison of the release temperatures measured at the outlet with the temperatures within the water column of the dam (Fig. 9) illustrates how the internal dynamics of the dam affect the release temperatures. The dam releases were sourced from bottom waters using a bottom release valve prior to May 2014, after which the thermal curtain was in use. In the summer of 2012 (Dec 2012 – Feb 2013) the outlet temperature is similar to temperatures observed within the hypolimnion at 20 m depth. For example, on the 3rd February the daily mean temperature of the downstream site, Downstream Burrendong GS, was 17.7°C, one degree above the 16.67°C recorded at 20 m within the hypolimnion. On the 2nd March 2013 the daily mean temperature of the downstream site was 19.4°C compared to 19.3°C at 20 m depth within the dam. At Long Point, an upstream reference site, the temperature recorded on this date was 6.7 °C above the observed temperature of the outlet.

For the upstream site temperatures to be matched with dam releases, they must exist within the dam. That is, if the water temperature within the reservoir is not warm enough to mirror upstream water temperatures, the temperature of releases from the reservoir cannot resemble those of the upstream site. The light blue line in figure 9 tracks the surface water (2 m, dark red line) very closely during periods of thermal stratification. During the period when the reservoir was thermally stratified from September 2014, downstream temperatures were similar to the surface water temperatures in the reservoir. A large block of missing thermal profile data occurred in 2014/15 due to equipment failure. Nevertheless, it is clear from the available data that the thermal curtain affected release temperatures at the outlet. For example, on the 2nd of February, 2015, water temperature at the outlet was 25°C, compared to 23.9°C at 2 m depth and 13.43°C at 20 m depth in the reservoir at the same time (Fig. 9). This data links the temperatures recorded at the outlet to the level of water withdrawal from the off-take within the dam.

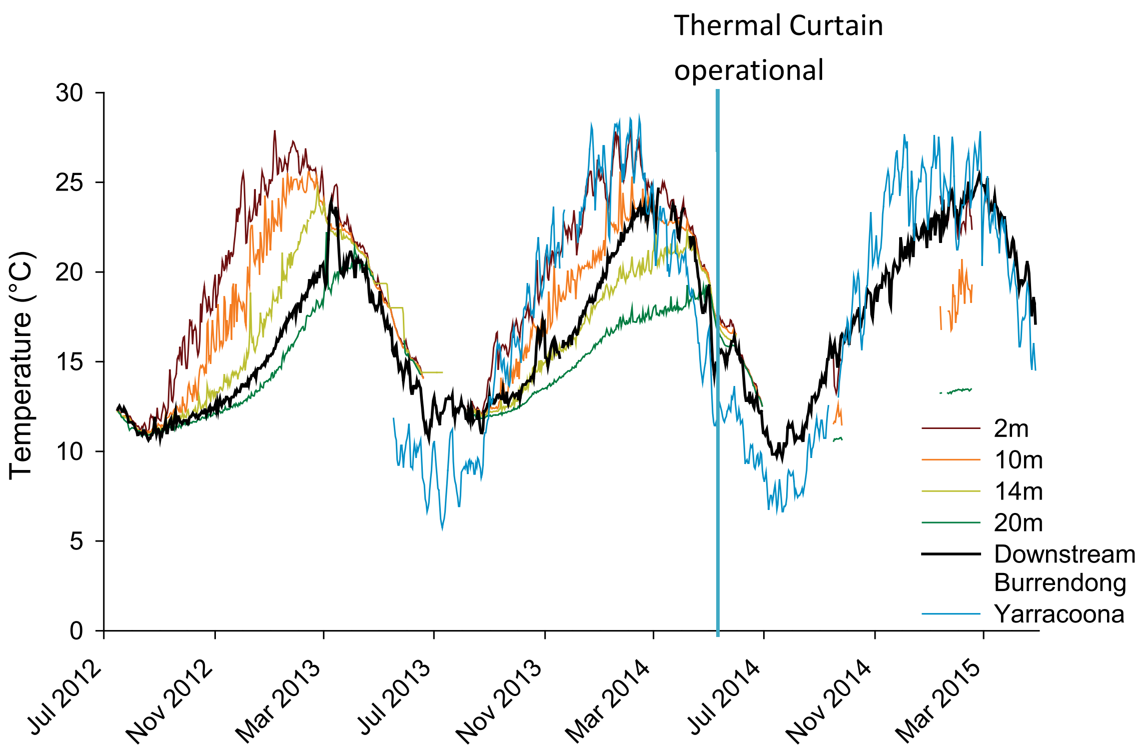


Figure 9: Time-series of mean daily temperature (°C) observations at various depths within the Burrendong Dam storage compared with mean daily temperature of Downstream Burrendong gauging station. Depths shown are 2 m (dark red), 10 m (orange), 14 m (green) and 20 m (dark green) and also Downstream Burrendong (black) and Yarracoona (light blue). The vertical blue line indicates the point from when the thermal curtain became operational (May, 2014).

3.4.4 Effect of residence level and discharge volume on downstream temperature

The residence level of the reservoir varied over time from July 2012 to March 2015 (Fig. 10). The depth of the off-take in relation to the surface water is affected directly by residence level and was similar in the summers for the two years 2013 and 2014 (Fig. 10). The off-take was 10 m below the water surface, when the residence level was around 322 m a.s.l. Therefore, the depth was similar between the two summers of comparison, and so would have impact on release temperatures in a similar manner.

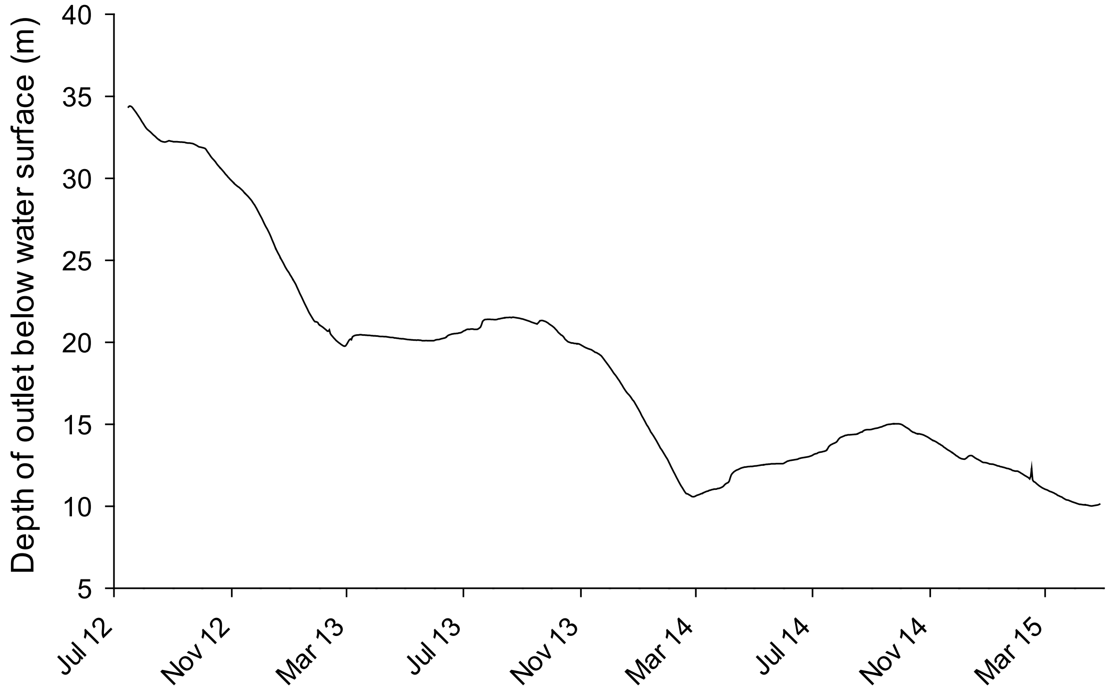


Figure 10: Depth (m) of bottom outlet in Burrendong Dam from water surface.

The residence level of the dam during the summers of 2004, 2007 and 2013 was similar to the residence level of 2014; the summer after operation of the thermal curtain began (http://realtimedata.water.nsw.gov.au/water.stm) (Fig. 10). This enabled additional comparisons of downstream temperature change in relation to temperatures in the reservoir close to the dam outlet prior to and following commencement of the thermal curtain operation.

Historically, there is an inverse relationship between discharge volume and temperature in the warmer months when releases have been sourced from the hypolimnion. In years of similar residence levels, decreases in discharge to relatively low volumes corresponded to a rapid increase in water temperature at the Downstream Burrendong GS (Fig. 11). In 2004, there were two periods of low discharge (October and December), which coincided with an increase in downstream temperatures of almost 6°C (Fig. 11a). In 2007, after three months of relatively low discharge below 500 ML/day, a four-fold increase in discharge volume coincided with a 7.5°C drop in temperature at the downstream site (Fig. 11b). Despite considerable variation of relatively large discharge volumes in 2013 (mostly greater than 2000 ML/day), there appeared to be little impact on temperature at the downstream Burrendong GS (Fig. 11c).

In 2014 with warmer surface releases, this relationship does not appear to be true (Fig. 11d). With increasing discharge the water temperatures at the downstream Burrendong GS are not reduced, rather in some cases they increase (mid-December and mid-January, for example). Overall in the period shown in figure 11d, the discharge does not appear to have as great an effect on water temperatures at the downstream Burrendong GS.

Discharge volumes in 2004 were the most similar to 2014 (Fig. 12) therefore temperature comparisons between these two years are the most appropriate. Except for three occasions in 2004 when the discharge fell below 100 ML/day (October, November and December), water temperatures were consistently 2-3°C lower than 2014 (Fig. 13). Monthly means of temperature at the DS Burrendong GS from October to February in 2004 and 2014 indicate an improvement in monthly temperature of 2.5°C in January (Fig. 14). This comparison of hypolimnial (2004) and epilimnial (2014) releases during periods of similar residence level and reservoir discharge, demonstrates the effect of the thermal curtain in moderating downstream temperatures.

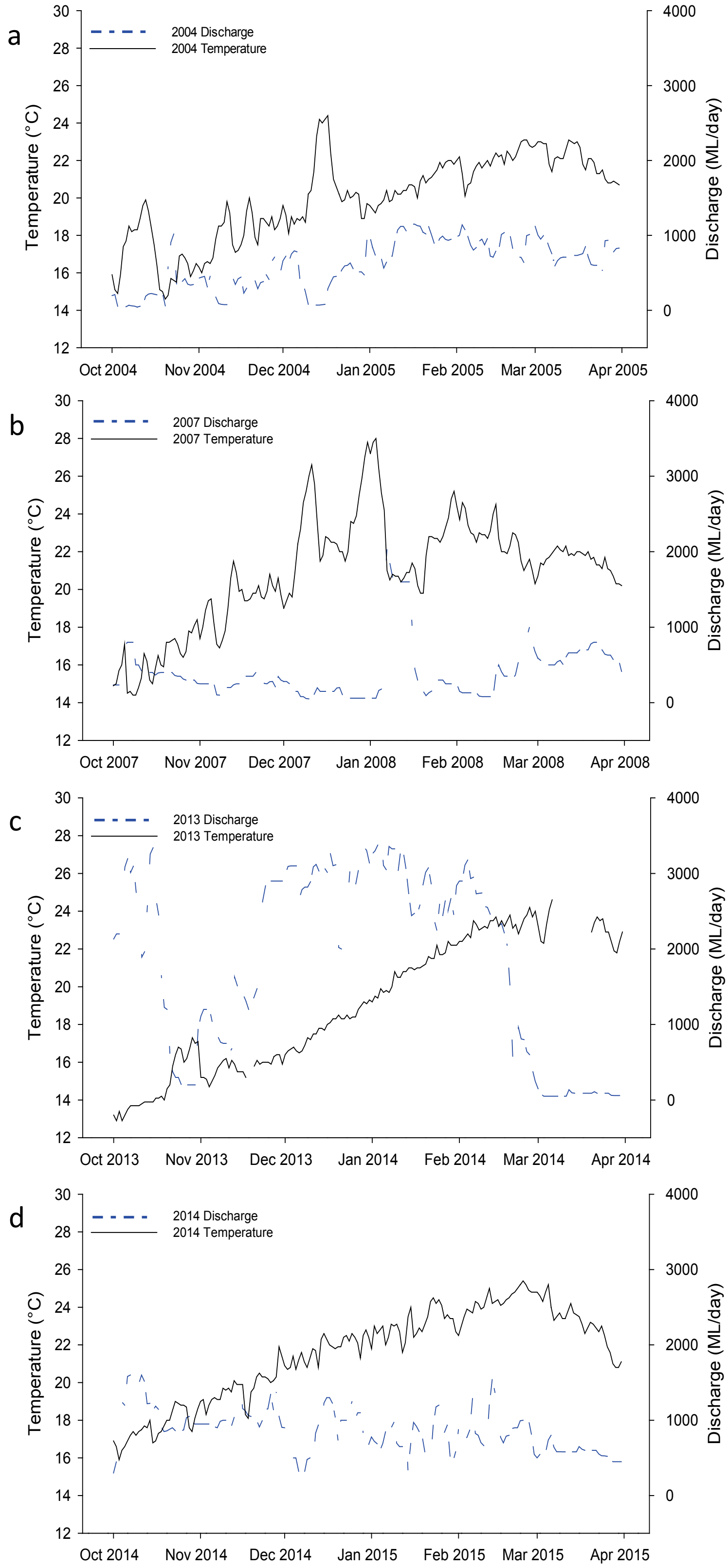


Figure 11: Four years of similar residence level within Burrendong Dam: a.) 2004, b.) 2007, c.) 2013 and d.) 2014 showing mean daily temperature (°C) of releases at Downstream Burrendong gauging station (black lines) and daily discharge from the reservoir (ML/day) (blue dashed lines). Data is shown for the period of October to April.

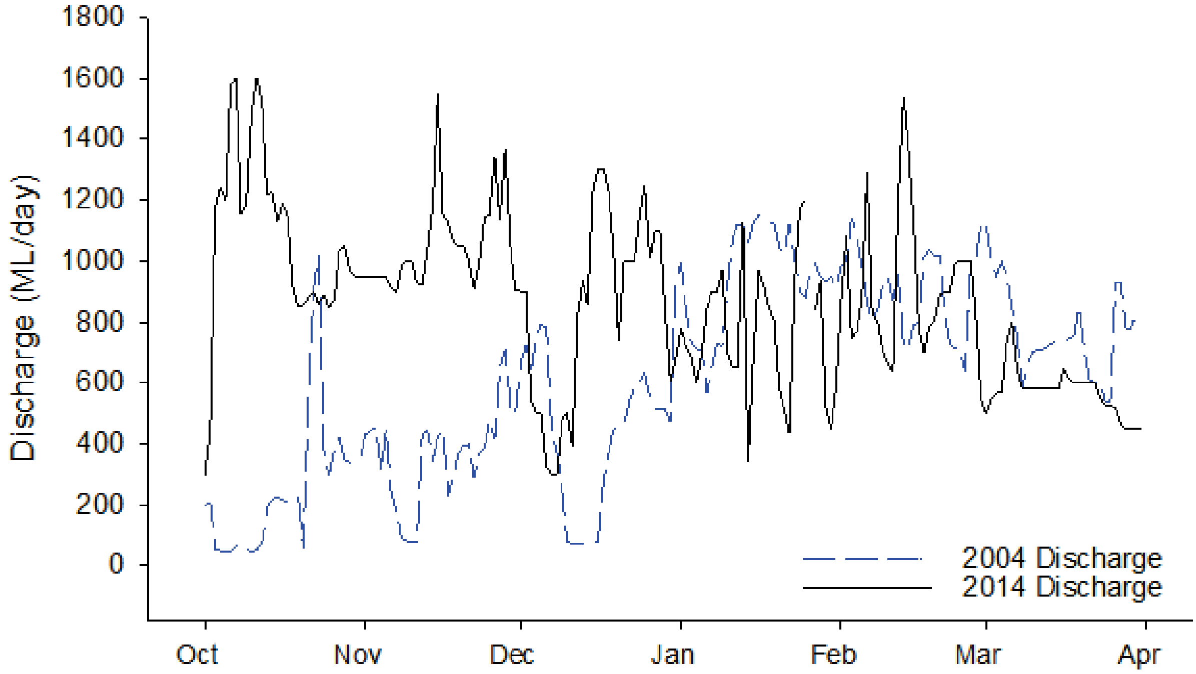


Figure 12: Comparison of daily discharge from Burrendong Dam between 2004 (blue) and 2014 (black).

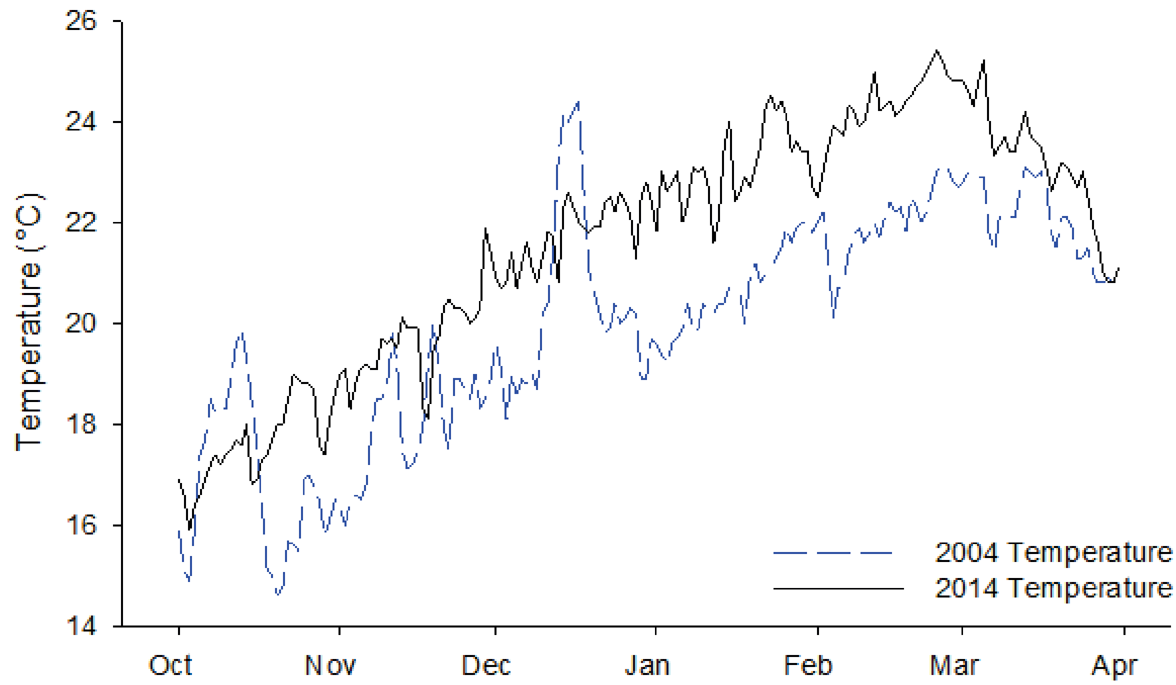


Figure 13: Comparison of daily mean temperatures (°C) recorded at Downstream Burrendong gauging station between 2004 (blue) and 2014 (black).

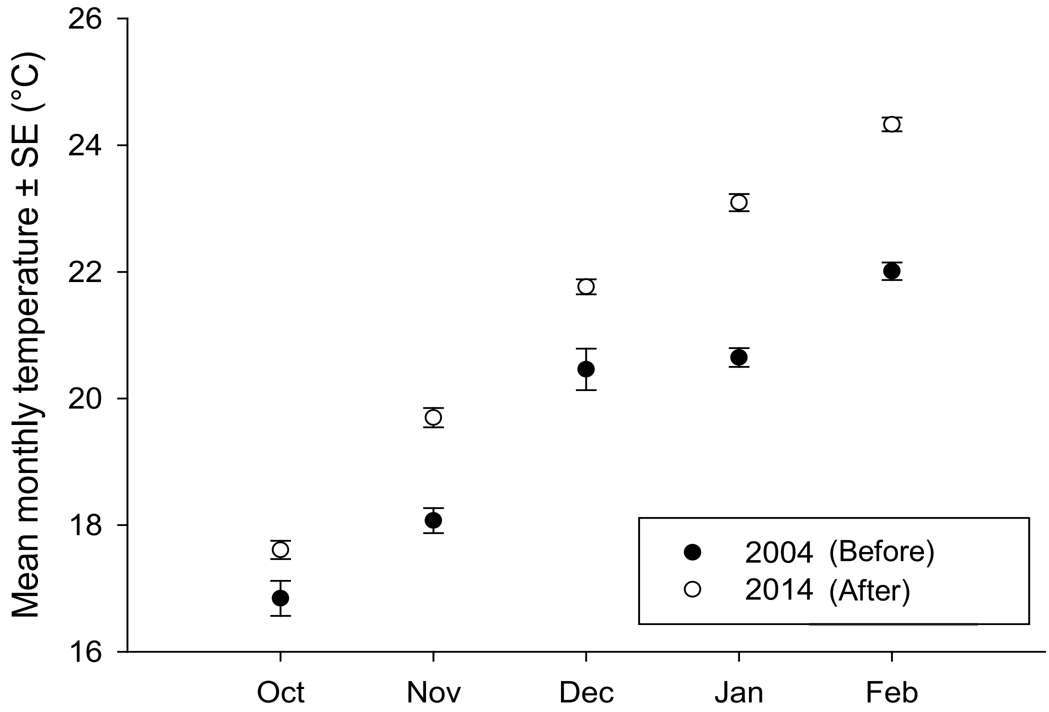


Figure 14: Comparison of mean monthly temperatures (°C) recorded at Downstream Burrendong gauging station between 2004 (black) and 2014 (white). Error bars indicate standard error.

3.4.5 Upstream versus downstream water temperatures

The pairwise differences in daily water temperatures between the upstream reference site at Yarracoona and the reservoir outlet in summer periods for 2013 and 2014 indicate the magnitude of temperature depression caused by Burrendong Dam releases (Fig. 15). In the absence of the dam, the temperature differences should be close to 0°C. Periods of CWP are shown by positive differences in temperature from September to March. The temperature differences in 2014 were often several degrees lower than the differences recorded in 2013, and rarely higher than in 2013. In 2013 the difference in daily mean temperature between the two sites was consistently several degrees higher than 0°C on most days during the period of stratification. In

2014, temperatures at the outlet replicated upstream temperatures on several occasions in October, mid-December and particularly from February to April.

Under the operation of the thermal curtain, the average monthly temperature difference between the upstream site, Yarracoona, and the dam outlet was reduced in 2014, compared with 2013 (Fig. 16). The largest discrepancy in monthly temperature differences between 2013 and 2014 occurred in December and January when there was an increase in monthly mean temperature of 3°C.

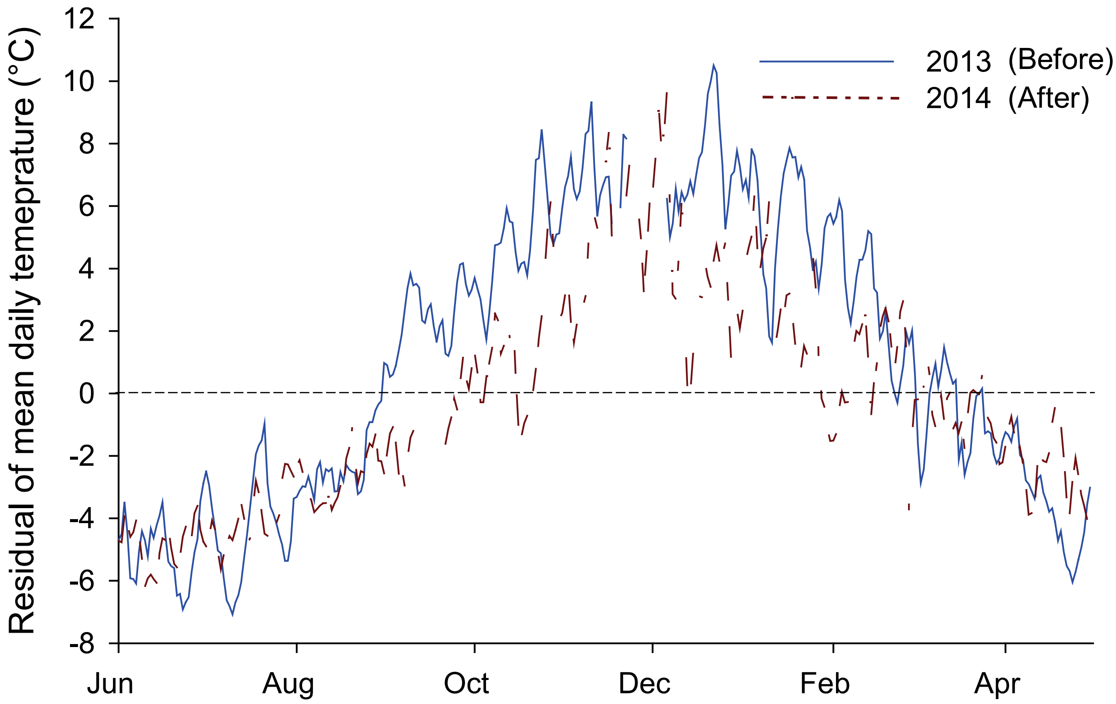


Figure 15: Difference in mean daily temperature (°C) between the upstream reference site (Yarracoona) and the Outlet in the year before thermal curtain use, 2013 (blue), and while the thermal curtain was active in 2014 (dashed red). Temperature differentials above 0°C indicate cold water pollution, while temperature differences below 0°C indicate warm water pollution.

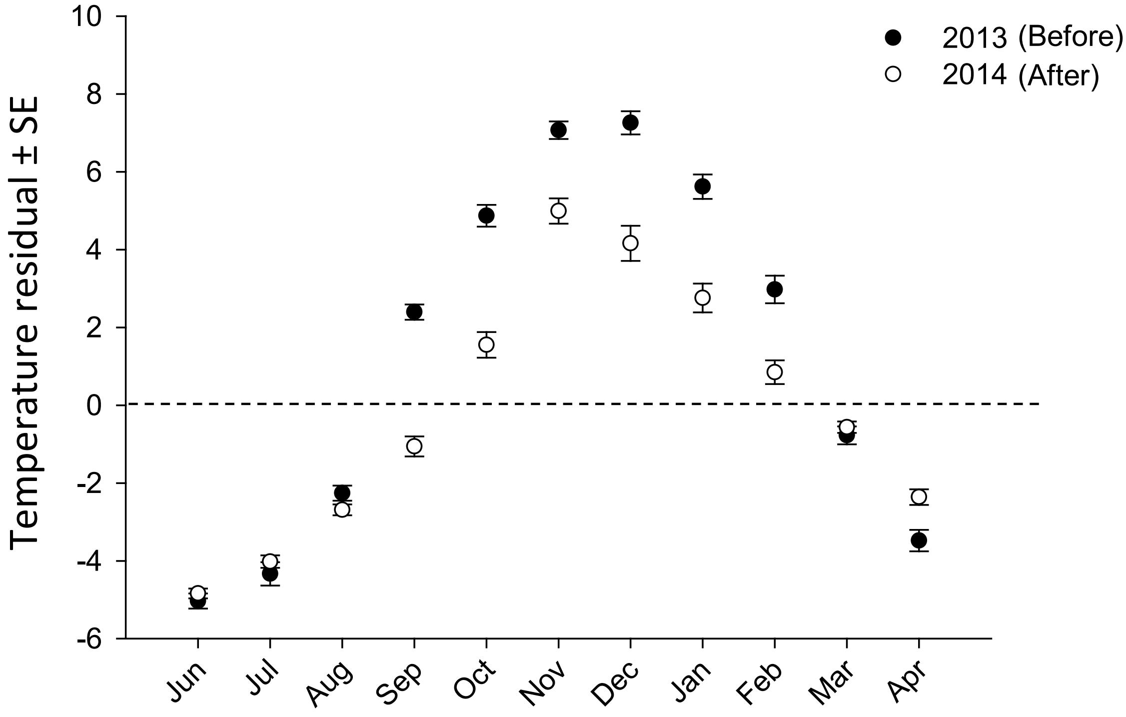


Figure 16: Difference in mean monthly temperature (°C) between upstream reference site Yarracoona and the Outlet in the year before thermal curtain use, 2013 (black), and white the thermal curtain was active in 2014 (white). Temperatures above 0°C indicate cold water pollution, while temperatures below 0°C indicate warm water pollution. Error bars indicate standard error.

3.4.6 The role of discharge in the functionality of the thermal curtain

Large fluctuations in discharge were observed during the period of thermal stratification from the point in time when the thermal curtain had been in operation (Fig. 17: *i* = drop, *ii*. and *iii*. = increases). The change in temperature between the surface and bottom of the water column within the thermal curtain (ΔT) is shown in figure 17b. When ΔT is higher, thermal stratification is stronger. At point *i*, the discharge decreased by 1070 ML/day in 9 days corresponding with a peak in ΔT (Fig. 17b), showing that thermal stratification increased inside the thermal curtain. Water temperatures at the outlet dropped as a result of this (Fig. 17c). Peaks in figure 16a indicated by *ii* and *iii* coincide with drops in ΔT (Fig. 17b), indicating that the increased flow of water through the thermal curtain prevented the development of strong stratification within the curtain. Corresponding temperatures at the outlet increased.

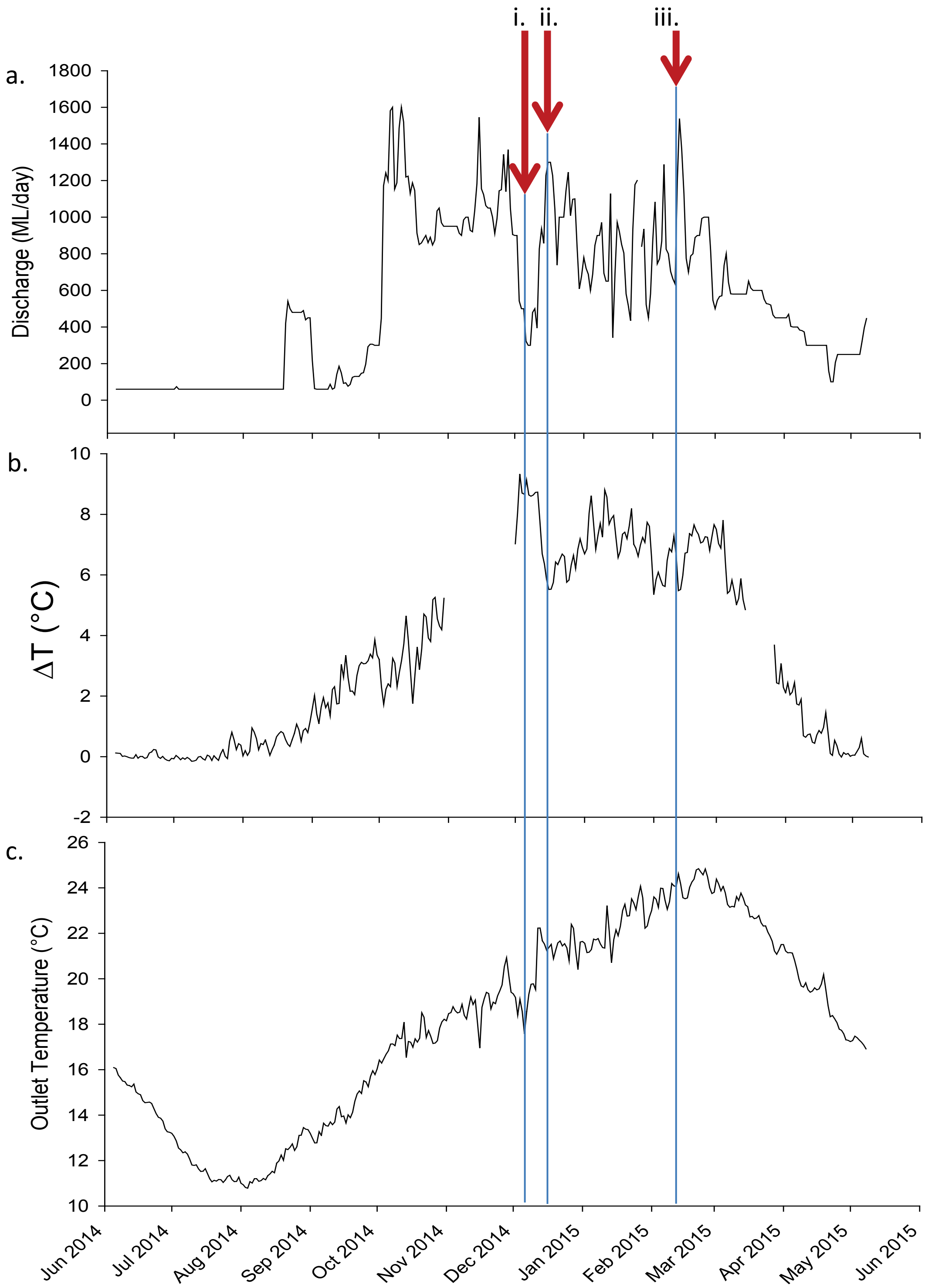


Figure 17: Three time-series comparing daily discharge from Burrendong Dam (ML/day), change in mean daily temperature between the surface and bottom of the water column within the thermal curtain (a), ΔT (°C)(b), and temperature (°C) observations at the Outlet gauging station (c). Drops (*i.*) and peaks (*ii. and iii.*) in daily discharge (ML/day) are compared with ΔT (°C) and outlet temperatures.

Figure 18 compares the temperatures recorded at the outlet during the lows and peaks in discharge (*i, ii* and *iii*) with temperatures recorded at the upstream reference site, Yarracoona. Increased discharge breaks down thermal stratification in the curtain, resulting in the release of warmer water through the outlet. At point *i*, when discharge dropped to 300 ML/day, the temperature at the outlet was 6.15°C cooler than the upstream temperature (Fig. 18). During a peak flow of 1300 ML/day, stratification within the thermal curtain weakened and the temperature at the outlet was 2.85°C cooler than the upstream temperature (Fig. 18 *ii*). At the highest discharge peak (1539 ML/day) during the period of thermal stratification within the reservoir, the observed temperature at the outlet was only 1.18°C cooler than the upstream site (Fig. 18 *iii*).

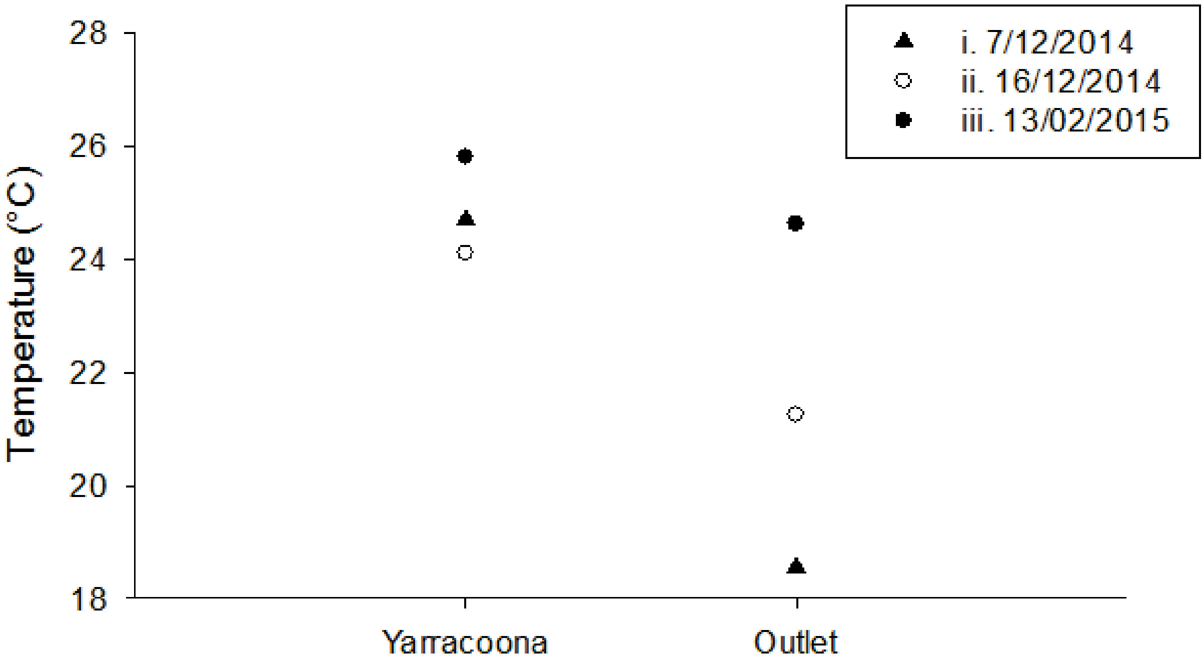


Figure 18: Mean daily temperature (°C) at the Outlet and upstream reference site Yarracoona are compared at the points of low (*i.*) and high (*ii.* and *iii.*) daily discharge (ML/day) identified in Figure 15.

There is a highly significant linear relationship between the magnitude of thermal stratification in the thermal curtain, ΔT, and daily discharge volume (Fig. 19). As discharge increases (Q), ΔT decreases ($\Delta T = 10.2 – 0.0044Q$, RSQ = 0.486, $F_{2, 115} = 108.8$, $p < 0.001$). When release volumes exceed 800 ML/day, the relationship between ΔT and discharge begins to break down.

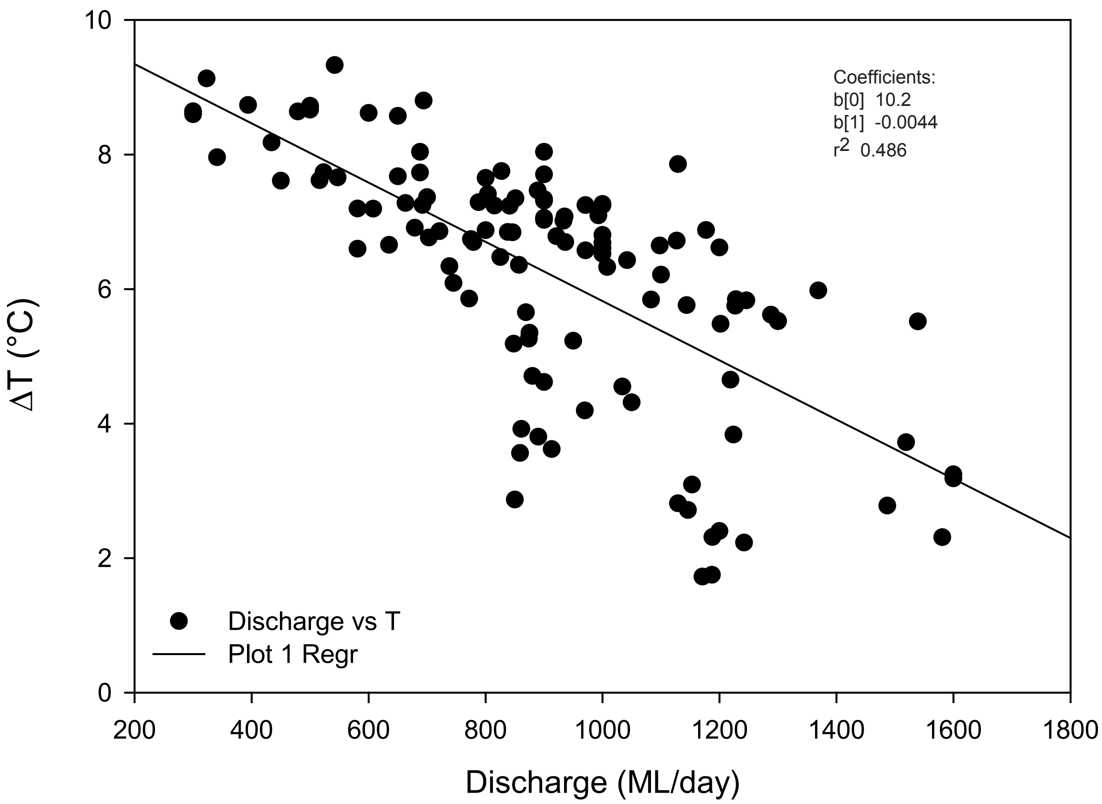


Figure 19: Regression of the mean daily temperature difference between the surface and bottom of the water column within the thermal curtain, ΔT (°C), and daily discharge (ML/day) from Burrendong Dam (October 2014 - February 2015).

3.4.7 Diel range

Diel temperature range at the outlet visibly increased with the use of the thermal curtain (Fig. 20). The maximum value of the diel range rose by a factor of 1.5 (up from a maximum of 2.88°C) prior to thermal curtain use (pre 6/05/2014) to 4.66°C afterwards (post 7/05/2014). Mean monthly diel thermal variation observed at the outlet was much lower than at Yarracoona throughout the summer of 2013 when releases from Burrendong were sourced from below the thermocline (Fig. 21). The highest mean monthly diel range observed at Burrendong outlet prior to operation of the thermal curtain was 1.5 ± 0.1°C in March 2014. This improved to 2.55 ± 0.2°C over the 2014 summer, after surface releases through the curtain had commenced. Furthermore, the monthly diel ranges at Yarracoona were mostly higher in 2013 summer than in 2014 summer. In contrast, the opposite was true for Burrendong Outlet, where the monthly diel ranges were mostly higher in 2014. Overall, the diel thermal range of the downstream site improved with the use of the thermal curtain.

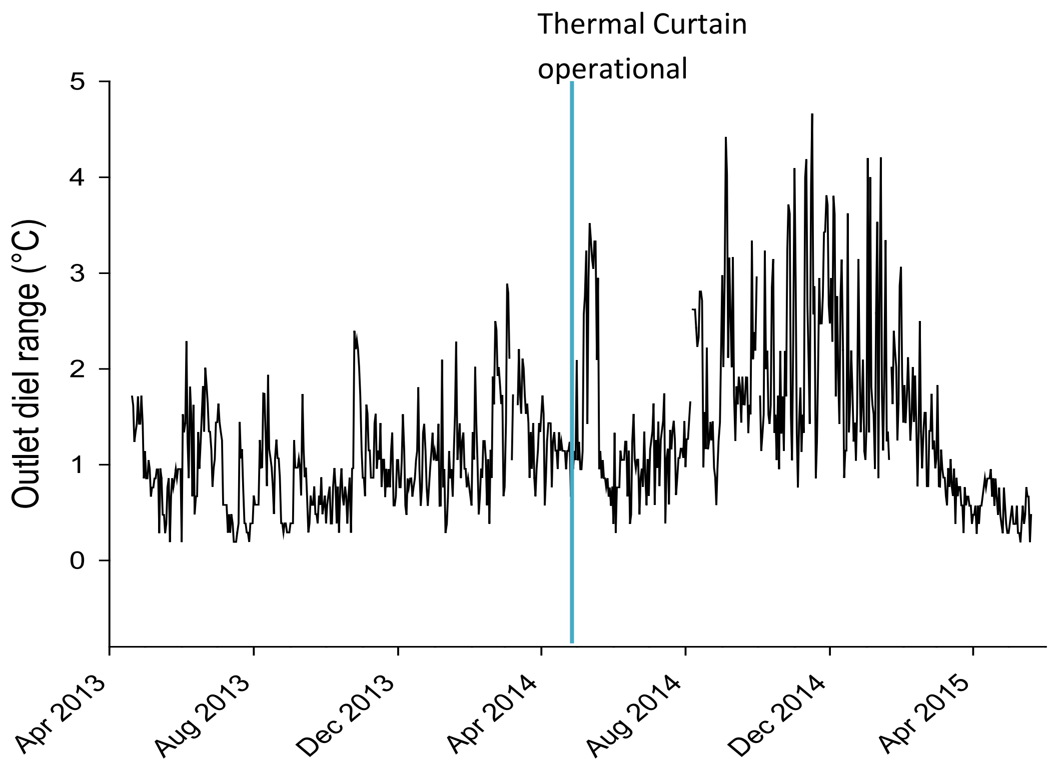


Figure 20: Diel range from May 2013 to May 2015 at the Outlet site, just below Burrendong Dam. The blue line indicates the start of the use of the thermal curtain.

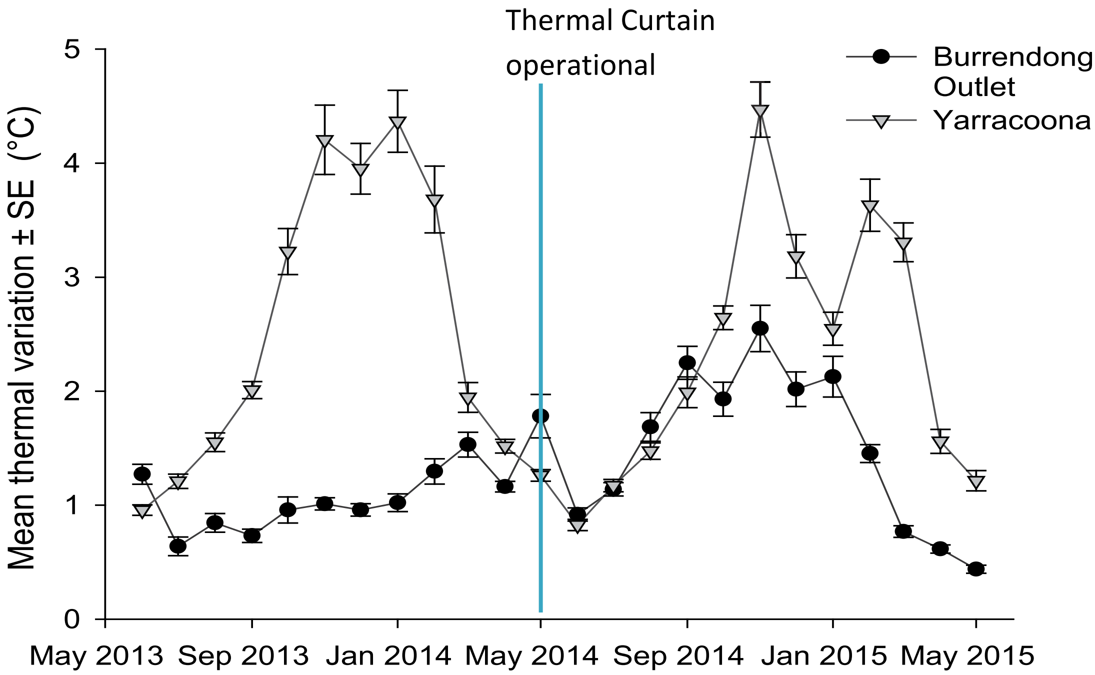


Figure 21: Comparison of the mean monthly diel ranges between the upstream site Yarracoona (grey triangles) and Burrendong Outlet (black circles) in the period prior to and after the use of the thermal curtain. The light blue vertical line indicates the point in time from when the thermal curtain was active.

3.4.8 Thermal recovery distance

A comparison of longitudinal temperature patterns on the Macquarie River was made for the two years preceding and following use of the thermal curtain. The monthly average water temperature (calculated from data logs at hourly intervals) for December 2013 and December 2014 at 7 sites along the Macquarie River were used in the comparison (Fig. 22.). Three sites upstream from Lake Burrendong on the Macquarie River (Yarracoona, Long Point and Bruinbun) were used as unregulated reference sites.

For this study, the water temperatures in the river downstream of Burrendong Dam was considered to be within the natural temperature range if they fell within the 95% confidence limits of upstream temperatures for December of both years. The December mean water temperature for the upstream site with data for both years (Yarracoona) was 24.8 ± 0.35°C in 2013, and 24.8 ± 0.30°C in 2014. The mean December temperature at the outlet differed by -7.5°C in 2013 and -4.2°C in 2014 in from the upstream value of Yarracoona. The 95% confidence interval was defined by the upper limit of Long Point (2013; 26.1°C) and the lower limit of Bruinbun (2013; 23°C).

The predicted asymptotic temperature (T_{max}) was not significantly different between years ($t = 0.170$, 3 d.f., $p > 0.05$), allowing a clearer comparison between the years. The rate of recovery (k) was far more rapid in 2014 than for 2013 (Table 2). In 2013, when releases were sourced from the hypolimnion, the thermal recovery of the Macquarie River downstream of the dam wall required a greater distance than that of 2014, when releases were sourced from the epilimnion with the thermal curtain (Table 2). In 2013, a distance of 125 km (37.5 km upstream of Barroona GS) was required for the mean water temperature to fall within the 95% confidence interval of upstream sites (Fig. 22). When releases were sourced from the epilimnion in 2014, the recovery distance was much shorter, at 20 km (13 km upstream of Wellington GS) downstream of the dam wall. This is an improvement of 105 km with the use of the thermal curtain.

Table 2: Parameter estimates for the negative exponential model of water temperature with distance downstream from Burrendong Dam.

Year	T_0	T_{max}	k	Residual std error (d.f.)
2013	17.33 (0.151)	26.67 (0.321)	0.0077 (0.00070)	0.168 (3 d.f.)
2014	20.68 (0.275)	26.56 (0.135)	0.0262 (0.00363)	0.277 (5 d.f.)

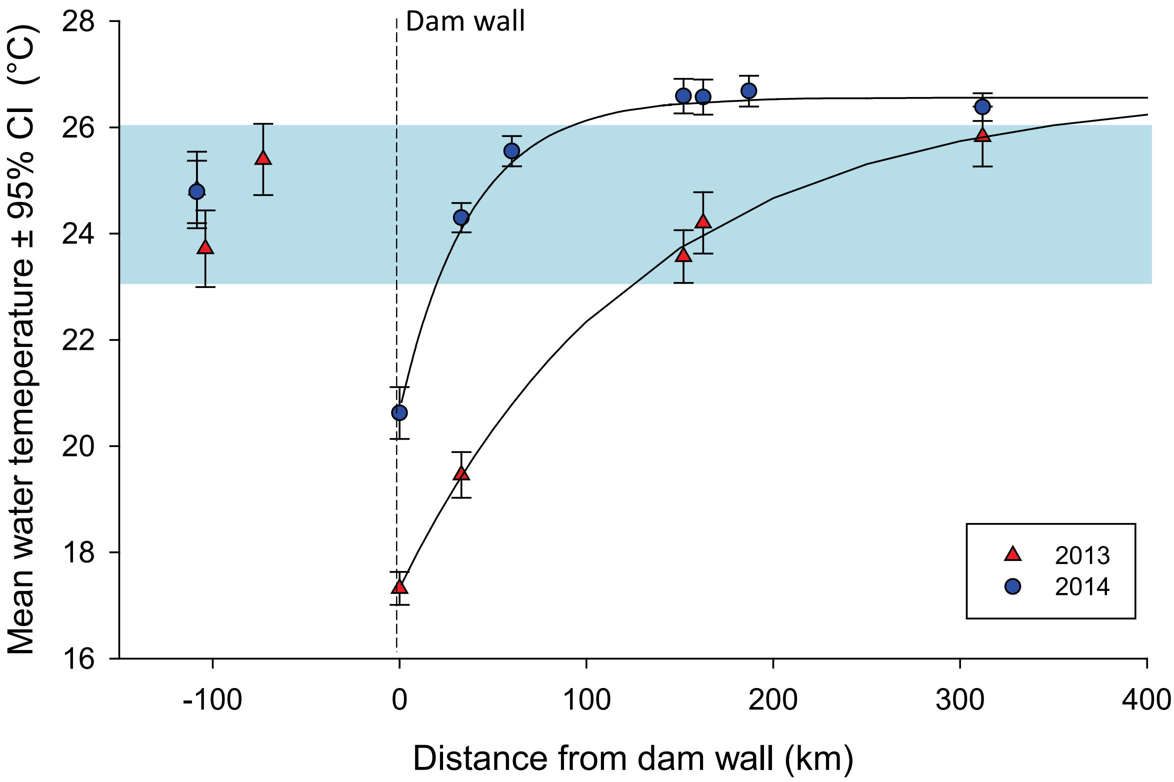


Figure 22: Average of December daily mean temperatures (°C) with standard error bars for December 2013 (red triangles) and December 2014 (blue circles) at sites on the Macquarie River with distance relative to Burrendong Dam wall. Asymptotic curves were overlayed for each year, indicating the temperature recovery between each site. The dashed vertical line indicates the location of the dam wall (0 km).

Visual comparison shows that the 2013 water temperatures were continuously less than the 2014 temperatures at sites close to the dam wall (Fig. 22). At the outlet, 2013 temperatures did not cross with 2014 temperatures during the stratified period. Differences between 2013 and 2014 diminished with distance downstream of the dam. The 2013 and 2014 water temperatures were similar at sites Baroona and Warren, downstream of the recovery distance identified in figure 21. The water temperature deviations fluctuated little near the dam outlet compared with the other sites further downstream (Fig. 23).

Daily temperature differences between these years decreased with distance from the dam wall (Fig. 24). The continuing reduction in residual means (Table 3; Fig. 24) indicates that the water temperatures may still have been recovering at Warren in 2013. The daily temperature deviations between Baroona and Wellington differed significantly ($p = 0.0002$) (Table 4). This supports the recovery distance of 125 km identified using the 95% confidence intervals and asymptotic curve in figure 21.

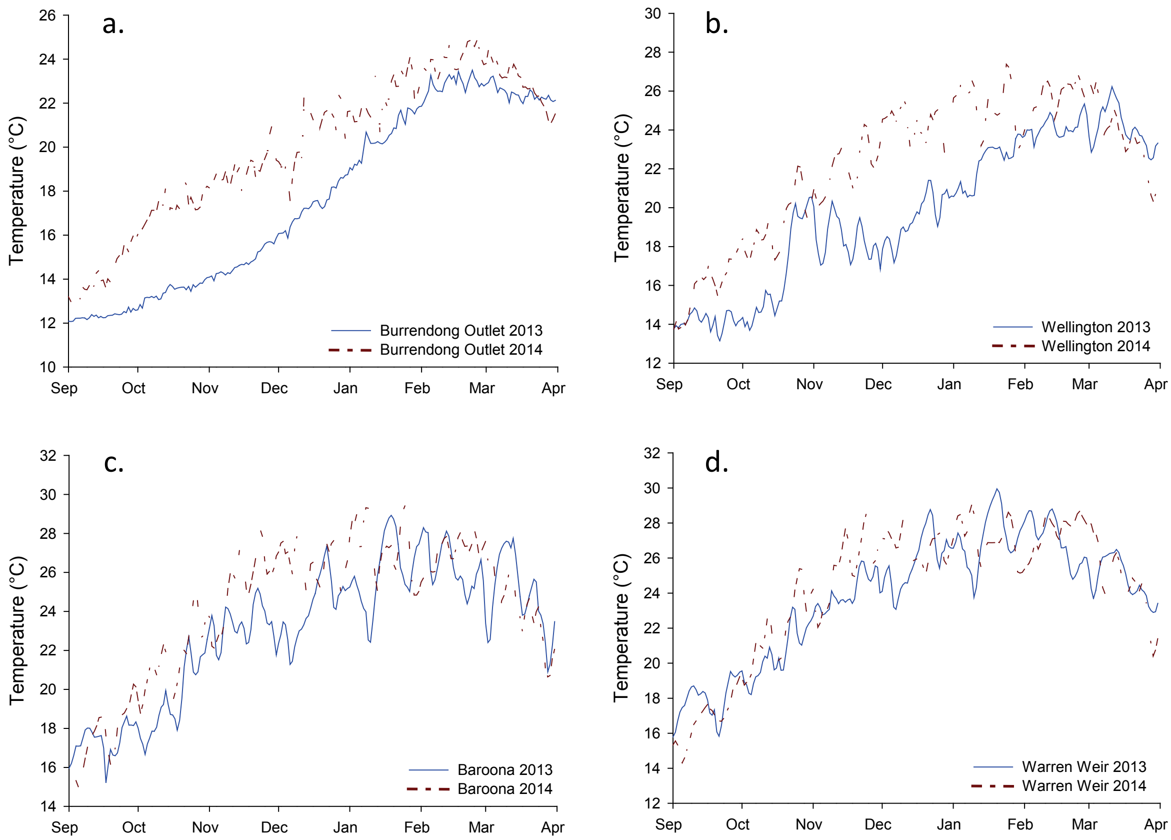


Figure 23: Daily temperature deviations in 2014 from 2013 observed at four sites with increasing distance from the dam wall. The observations were recorded from September 2013 to April 2014 (blue) and September 2014 to April 2015 (red dashed). The sites shown are Burrendong outlet (a), Wellington (b), Baroona (c) and Warren weir (d).

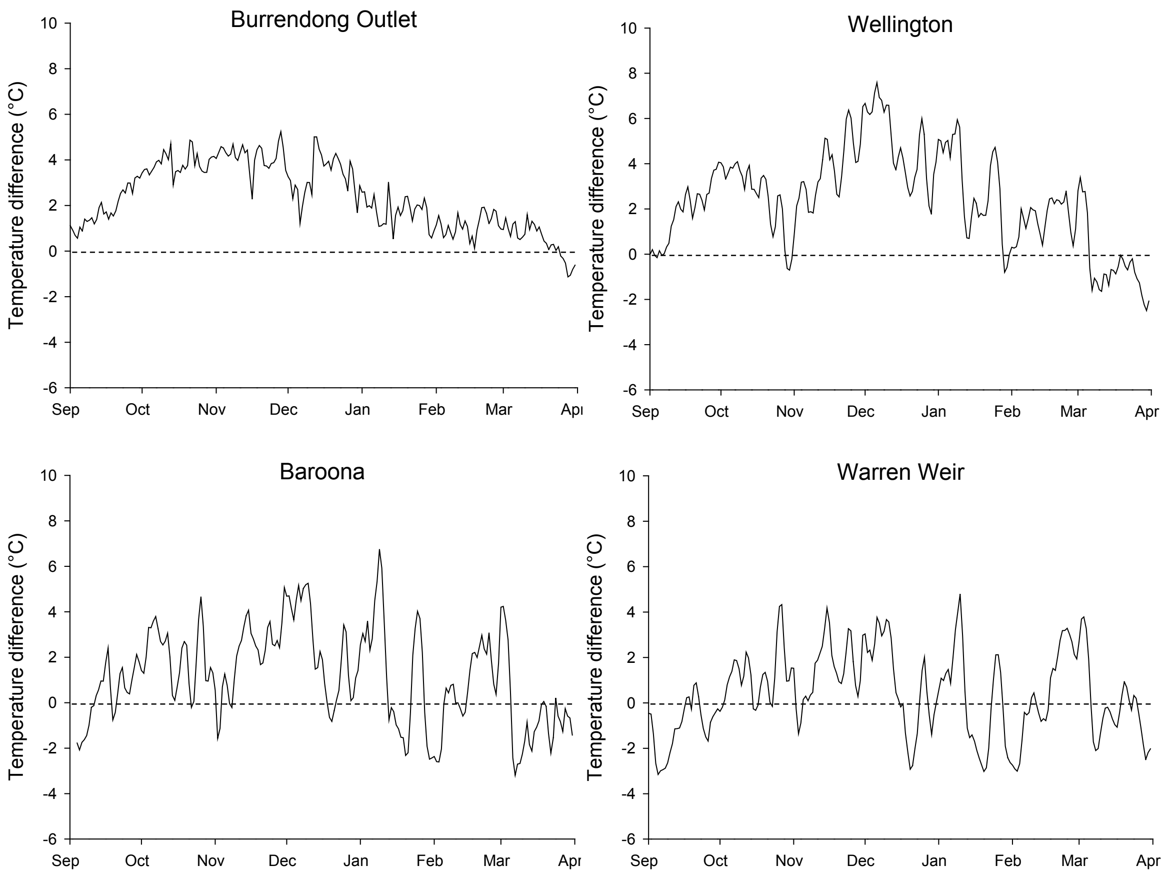


Figure 24: The residuals from the difference in temperature (°C) between the year before (2013) and after (2014) the use of the thermal curtain, with distance from the outlet. The sites shown are Burrendong outlet, Wellington, Baroona and Warren weir.

Table 3: Summary statistics (mean, median, median absolute deviation, minimum and maximum values) for matched daily differences (time period: July-December) in mean water temperature for 5 selected sites at increasing distances downstream of the dam wall.

Site	Mean	Median	MAD	Min	Max
Dam outlet	2.04	2.6	2.22	-1.6	5.2
Wellington	1.80	2.1	2.97	-4.7	7.6
Baroona	0.96	0.9	2.22	-4.3	5.3
Gin Gin	0.76	0.75	2.00	-2.8	4.2
Warren	0.43	0.3	1.78	-3.2	4.4

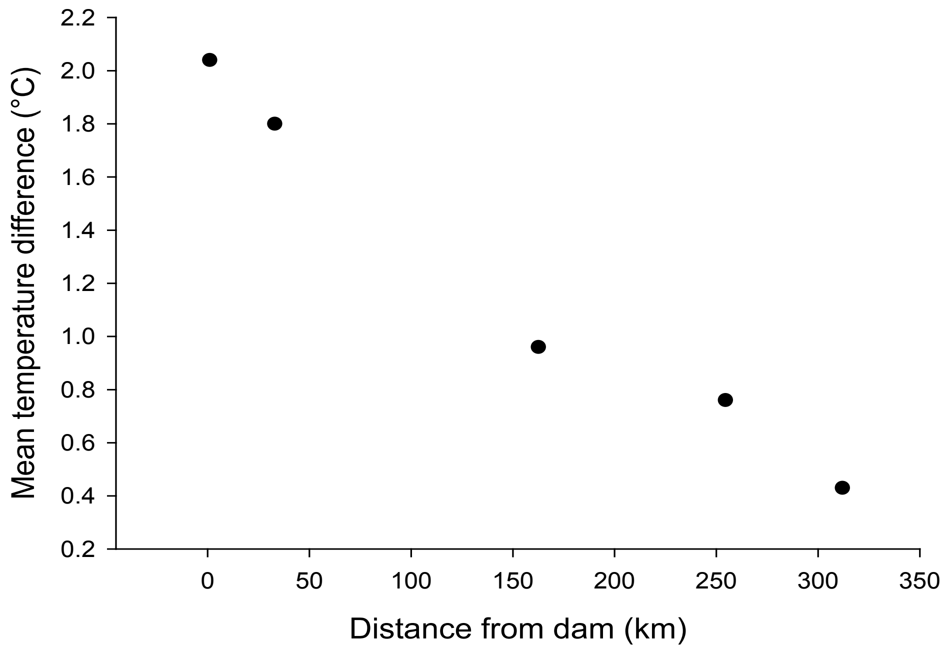


Figure 25: Mean daily temperature differences between July-December of 2013 and 2014 at five sites downstream of Burrendong Dam wall. The sites, plotted with relative distance from the dam wall, are Burrendong outlet, Wellington, Baroona, Gin Gin and Warren weir.

Table 4: Summary of Kolmogorov–Smirnov two-sample test comparisons of daily temperature deviations for 2013-14 (July-December) between monitoring sites on the Macquarie River downstream of Burrendong Dam.

Site Comparison	D statistic	p-value
Gin Gin vs. Warren	0.122	0.224
Gin Gin vs. Baroona	0.110	0.285
Baroona vs. Wellington	0.226	0.0002
Wellington vs. Dam outlet	0.120	0.144

3.4.9 Biological implications: Thermal effects on native fish

Thermal ranges required to cue the onset of spawning of three native fish to the Macquarie River are shown in figure 26. Silver perch require a minimum of 23°C to be reached for spawning in September to late January, Murray cod need a rise from 16.4°C to 20.6°C in early summer, and golden perch need a minimum temperature of 23.5°C during their spawning season from October to March (Lake 1967, Merrick and Schmida 1984). In 2013 the lowest observed temperatures during the window of reproduction were almost 6°C below the minimum temperature for the spawning of Murray cod, 8°C for silver perch, and at times up to 10.5°C below required minimum temperatures for golden perch. The temperature ranges were only briefly met for golden perch episodically near the end of the spawning season and not at all for Murray cod and silver perch. In the summer starting in 2014, with the curtain operational, thermal ranges were met more of the time for Murray cod, and some of the time for silver perch and golden perch.

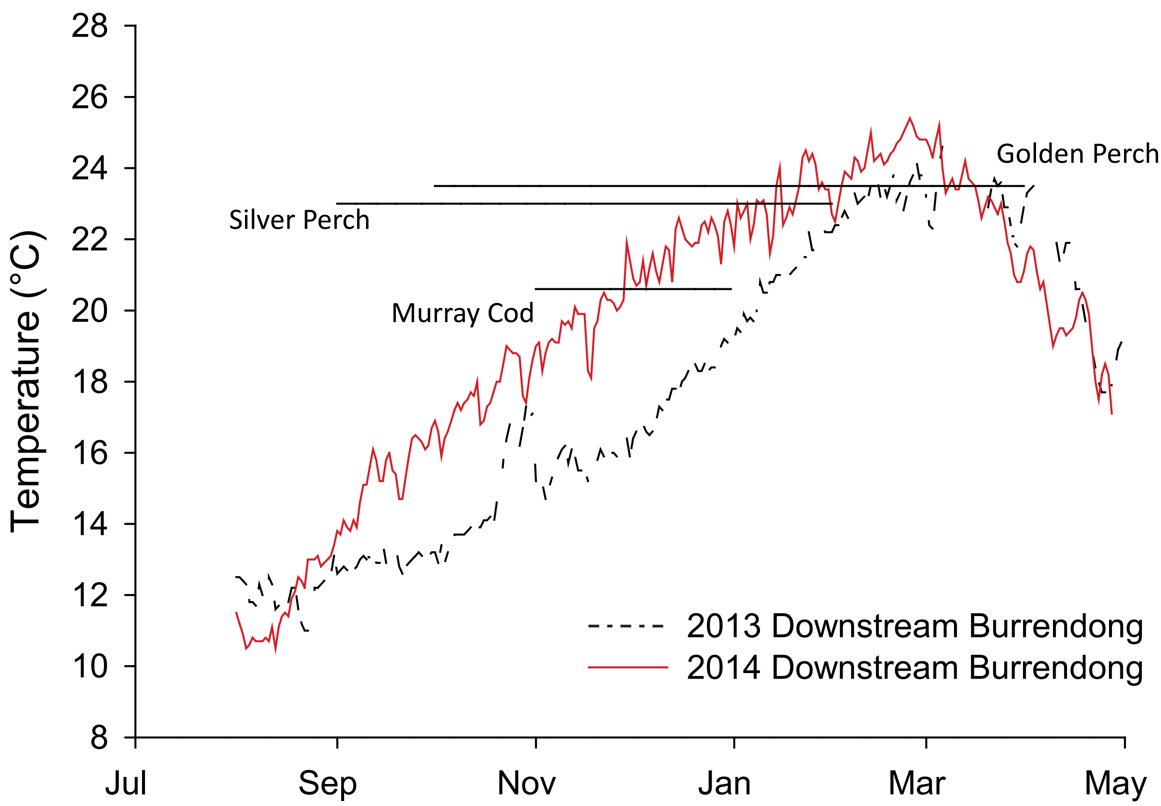


Figure 26: Daily mean temperature (°C) observed at Downstream Burrendong gauging station in the summer months of 2013 (black dashed) and 2014 (red) are overlayed on a time-series. Three horizontal lines indicate minimum spawning temperature over the spawning period of three native fish in the Macquarie River.

3.5 Discussion

3.5.1 Historical cold water pollution

When thermally stratified, reservoirs with low-level off-takes draw water from the unnaturally cold hypolimnion. This can result in depressed downstream river temperatures compared to natural, unregulated conditions (Acaba et al. 2000, Preece and Jones 2002, Ryan and Preece 2003, Preece and Wales 2004). Historical thermal patterns of the Macquarie River immediately below Burrendong Dam showed this with frequent periods of temperature depression. The most severe CWP events occurred in the spring-summer periods when maximum differences in daily mean temperature between upstream and downstream sites often exceeded 10°C. This coincided with thermal stratification of the reservoir, high reservoir residence level and high discharge volumes from the hypolimnion. High residence levels in large reservoirs with low-level outlets have been linked to more extreme temperature depression downstream. A greater residence level raises the thermocline further from the outlet, ensuring that cold water is drawn in releases (Preece and Jones 2002, Sherman 2005). Because of the high specific heat capacity of water, large waterbodies are exceptionally stable to thermal variation (Gordon et al. 2004, Sherman 2005). The temperature of water at high discharges is less responsive to atmospheric conditions than at low discharge because of the greater volume, resulting in the low temperatures observed at the source persisting further downstream (e.g. headwaters or hypolimnetic releases) (Gu et al. 1998).

These factors were pertinent when CWP was most prominent in the years 2010, 2011 and 2012 (Fig. 6), when residence level was greater than 340 m a.s.l. (at this depth, the outlet was 27.43 m below the water surface) and dam discharge exceeded 5000 ML/day. The water temperatures of the Downstream Burrendong gauging station reflected the temperatures observed at 20 m depth of within the reservoir water column in 2012 (Fig. 9).

3.5.2 Effectiveness of the thermal curtain in reducing cold water pollution

Thermal curtain use improved the water temperature below Burrendong Dam. A shift in temperatures between the summers of the two years was observed, with a sustained 2-3°C temperature increase in mean daily temperature in 2014 (Fig. 13), and an increase of 2.5°C in mean monthly temperature (Fig. 14). The improvement in temperature can be attributed to the selective release of warmer surface water with the thermal curtain. These thermal improvements have resulted in water temperatures immediately downstream of the dam that more closely resemble the upstream thermal regime. Improvements of around 3°C in mean monthly temperatures can be seen in January and December after implementation of the thermal curtain (Fig. 16). However, discharge volume may have impacted on the effectiveness of the thermal curtain in mitigating CWP.

In the year prior to the installation of the thermal curtain (2013), the thermal regime downstream of the dam was seldom representative of natural upstream temperatures. From the time when the thermal curtain began to withdraw surface waters, the difference in temperature between upstream and downstream sites was reduced. On 6 occasions there was no temperature difference, indicating that the thermal curtain has been successful in reducing CWP and creating a more natural thermal range for the Macquarie River. Windamere Dam, located upstream of Burrendong Dam on the Cudgegong River, has been similarly successful in mitigating CWP via selective withdrawal of surface waters using a MLO during periods of low discharge (Burton 2000a). In periods of high discharge from Lake Windamere (e.g. water delivery to Burrendong Dam), there is potential for hypolimnetic releases to reduce temperatures up to 8°C below natural temperatures. However, selective withdrawal can reduce this to 4°C below natural.

It is important to note that the volume of the reservoir throughout the period of this study was low (10-20% capacity). There were times when the outlet was only 10 m below the surface of the water. With low residence level, the thermocline is closer to the outlet, meaning warmer temperatures will be released downstream. When the dam has a higher residence level, the thermocline is raised further from the outlet, resulting in more severe CWP. The results of this study are modest considering the low residence level, and it is likely that when the dam is holding a larger percentage of water to full capacity, the potential for cold water pollution mitigation will be greater.

3.5.3 Diel range

Water temperature naturally varies on diel and seasonal cycles, largely in response to climatic conditions including air temperature and solar radiation (Brown 1969, Sinokrot and Stefan 1994, Evans et al. 1998, Caissie 2006). Diel and short term variability in water temperature is highly important in the ecological health of a river. Fluctuations in river temperature provide an overlap in niche conditions for aquatic organisms. Subsequently, the thermal fluctuations alternately favour different species (Hutchinson 1953, Sweeney and Schnack 1977). Furthermore, diel variation also serves as an important environmental cue for spawning and egg and larval development of many fish, copepods and invertebrates (Lehmkuhl 1972, Sweeney and Schnack 1977, Sweeney 1978).

Releases from the hypolimnion of a dam reduces the thermal variation within a day, and buffers the natural thermal extremes of winter and summer (Preece and Jones 2002). The hypolimnion is physically isolated from the atmosphere, meaning it does not respond to changes in air temperature and other climatic circumstances as much as surface waters. The surface water of a reservoir is exposed to the atmosphere and as a result, though still largely buffered, does respond to the thermal conditions it encounters on a day to day basis. Consequently, the diel range of hypolimnial releases is minimised, as seen in the Macquarie River (Figs. 20 & 21). By channelling surface waters for release into the Macquarie River with the thermal curtain, an improvement in the diel thermal range was clear and much more representative of natural conditions. The diel ranges of upstream sites, which tend to be more shallow and narrow streams, are generally more extreme than the larger, deeper downstream portions of a river. Therefore it is expected that the diel variation seen at the outlet will have some deviations from the natural reference site, which is more than 100 km upstream of the dam.

3.5.4 The role of discharge in the functionality of the thermal curtain

This study has shown that thermal stratification develops within the thermal curtain at low discharge volumes, possibly accounting for the variation in the difference between daily temperature means of the upstream and downstream sites in 2014 despite surface water withdrawal. The low discharge allowed the body of water within the thermal curtain to stratify, most likely through similar mechanisms to the rest of the reservoir. There may have been heat loss through the thermal curtain when the temperature within it was greater than that of the reservoir water body outside it.

There was a breakdown of stratification within the thermal curtain structure as discharge increased. This indicates that the stratification, which reduces the effectiveness of the thermal curtain by lowering the temperature of the water before it is released, can be managed. Increased water movement is known to disrupt stratification patterns, which requires still or very slow moving water bodies to develop (Reinfelds and Williams 2012). This pattern was then linked to observed outlet water temperatures, which exhibited a significant linear relationship with outlet discharge. High discharge through the thermal curtain coincided with increasing downstream temperatures in summer, demonstrating the increased effectiveness of the curtain. A discharge threshold of 800 ML/day was identified using linear regression (Fig. 19), where the negative relationship between discharge and ΔT is strongest up until this point and weakens at greater discharge volumes. With weakened stratification within the curtain, the water temperature of the releases will be warmer than periods of stronger stratification. This identifies a threshold for the effective management of thermal stratification within the thermal curtain. The weakening of this occurrence of stratification in the curtain allows the transport of warmer surface waters to the outlet, without excessive heat loss.

This information is important for the effective management of the thermal curtain. The highest water demands occur in summer to meet irrigation needs for agriculture downstream. High discharge volumes during thermal stratification are known to cause severe and far-reaching CWP downstream of the dam. With curtain operation, high discharge volume means these agricultural needs may be met in the future without the increased flow being detrimental to the thermal regime of the Macquarie River.

This relationship also may be beneficial when allocating environmental flows, such as regulating timing and volume of flow patterns to maintain riverine ecological health (Kingsford and Auld 2005, Rolls et al. 2013, Bino et al. 2014). The effectiveness of environmental flows may be nullified when the thermal regime is not considered

(Olden and Naiman 2010). The incorporation of a river's natural thermal regime is essential for the success of the flow events in maintaining ecological health (Lugg 1999, Lugg and Copeland 2014). The identification of the discharge threshold to improve the thermal outcomes with use of the curtain should be considered in future environmental releases on the Macquarie River. Releases should be above 800 ML/day to better mimic the natural thermal regime.

3.5.5 Thermal recovery distance

The month that consistently experiences the most severe CWP (December) was selected to assess the longitudinal persistence of the thermal effects of Burrendong Dam in the year before and after the installation of the thermal curtain (2013 and 2014). A comparison of the monthly means of the two December periods indicates the thermal curtain is effective in improving the thermal recovery distance (Fig. 21). Results indicate that CWP in December 2013 persisted for 125 km. The total mean recovery distance was reduced by 105 km, with the thermal effects of the dam persisting to only 25 km downstream in 2014. The use of asymptotic curve against the 95% confidence intervals of upstream sites allowed these distances to be identified. These distances were further supported by temperature residuals between 2013 and 2014 at 5 sites downstream of the dam wall (Table 3), indicating that daily temperature differences between these years decreased with distance from the dam (Fig. 24). The significantly different residual means of Wellington GS and Baroona GS represent the closest sites to the recovery distance of 125 km in 2013 and 20km in 2014, further supporting the finding that the mean temperature of the river recovered to upstream temperatures by Baroona GS.

Harris (1997) was the first to assess the persistence of CWP on the Macquarie River in creating a longitudinal thermal profile of the river and concluding that over 300 km of river was seriously affected by CWP below Burrendong Dam (defined as 5°C below predicted natural). The average of two readings on consecutive days was compared to predicted natural temperatures (produced from a regression of best fit). Interestingly, Burton (2000b) presented a study of monthly means and compared the data to 20th and 80th percentiles calculated from temperature data immediately upstream of the dam. He concluded 117 km of the Macquarie River was suppressed by 1-2°C below natural temperatures. The two studies used different methods for assessing the impact of the dam resulting in different conclusions on the extent of the problem.

This study differs by comparing short time-series segments of the summers before and after the implementation of the thermal curtain and by focusing on monthly means of the worst affected month in 2013 and 2014. The combination of these methods accounts for the daily variation in temperature over the full period of CWP, allowing a wider view of the thermal trends in 2013 and 2014 summers.

3.5.6 Biological and ecological implications: Fish

Rising water levels and temperature are both essential environmental cues for the onset of spawning in many native fish (Lake 1967). Golden perch, silver perch and Murray cod are just a few of the native fish residing in the Macquarie River which require a minimum water temperature to be reached during their respective reproductive seasons to cue the onset of spawning (Lake 1967, Merrick and Schmida 1984). When comparing these minimum required thermal ranges over their respective time frames to the observed daily mean temperature at the outlet of Burrendong

Dam, it is clear that the thermal regime with low-level releases was usually not suitable for these native fish (Fig. 26). With surface releases however, the thermal range is much more favourable. This study suggests that cold water releases from the dam may have been previously directly impacting negatively on the viability of native fish species. With the use of the curtain there is evidence that the thermal reproductive requirements of these fish are now being met.

Altered thermal regimes have been linked to low population density and body mass of native fish in affected rivers (Astles et al. 2003, Sherman et al. 2007, Donaldson et al. 2008). For example, juvenile silver perch exhibit low growth and survival rates when exposed to water temperatures up to 10°C below their natural range (Astles et al. 2003). Eggs and larva of the Murray Cod exposed to temperatures below 13°C (3-8°C below natural ranges) presented a 0% survival rate (Todd et al. 2005). Further research is needed in the Macquarie River below the dam to assess the effects of the thermal curtain on the native fish community.

Positive ecological implications are also tied with improved diel thermal ranges. The Macquarie River more closely follows natural patterns after the use of the thermal curtain. Fish growth rate is affected by diel variation in water temperature, whereby increased growth rates are associated with fluctuating temperatures in comparison to constant water temperatures (Diana 1984, Lyytikainen and Jobling 1998, Sadati et al. 2011). Oxygen consumption and metabolic activity also increase in these fluctuating conditions (Lyytikainen and Jobling 1998) and immunity is positively affected by diel thermal variation in some fish (Sadati et al. 2011).

3.5.7 Biological and ecological implications: Macroinvertebrates and plankton

Other aquatic organisms including plankton and macroinvertebrates respond positively to improved diel thermal ranges more reminiscent of unregulated rivers. Egg and larval development of copepods often improve when exposed to water with larger diel thermal variation, and higher mortality rates in groups exposed to steady temperature in lab experiments (Sweeney and Schnack 1977). There were similar findings in in-situ experiments with macroinvertebrates (Sweeney 1978). It was suggested that aquatic copepods and macroinvertebrates are more metabolically efficient in an environment with a fluctuating thermal regime. In one instance, temperatures considered lethal to insects when exposed at a constant rate were used as the maximum in a diel cycle, resulting in a stimulatory response in the exposed insects (Sweeney and Schnack 1977).

Significant changes in the community structure of benthic organisms have been noted below impoundments, contributed mainly to the downstream lag and depression of the summer maximum temperature (Spence and Hynes 1971). Impacts on invertebrates include reduction in species diversity and the increase in the population number of some species favoured by the altered thermal regime (Spence and Hynes 1971, Ward 1976). The delay and reduction of seasonal maxima in spring and summer may fail to stimulate the emergence of some insects. Correlation between hatching time and temperature is well established, indicating extended hatching time and high mortality rates result from exposure of eggs to temperatures above or below their natural thermal range (Elliott 1978, Friesen et al. 1979).

3.6 Conclusions

Hypothesis 1a was confirmed when CWP was identified as a long term problem on the Macquarie River, caused by the release of hypolimnial waters from a thermally stratified reservoir. Downstream temperatures were improved through the use of the thermal curtain, both immediately downstream of the dam wall, and on a longitudinal scale, supporting hypotheses 1b and 1c. Diel variation was negatively affected by hypolimnial releases and improved with the change to surface water releases. The downstream thermal regime more closely reflected upstream reference sites with epilimnial releases through the thermal curtain.

This study has also identified thermal stratification within the thermal curtain as a potential problem whereby with low discharge may reduce the effectiveness of the structure in mitigating CWP. This can be controlled through the management of flows, where increased discharge above 800ML/day threshold will result in the optimised use of thermal curtain for the mitigation of CWP.

It is important to note that the volume of the reservoir throughout the period of this study was consistently low, with the thermocline located closer to the outlet. When the dam has a higher residence level, the thermocline is raised further from the outlet, resulting in more severe CWP. Therefore, the results of this study are modest, and it is likely that with increased residence level in Burrendong Dam, the effectiveness of the thermal curtain at mitigating cold water pollution will be more profound.

Chapter 4: Nutrients and Water Quality

4.1 Introduction

Nutrients are important determinants of ecosystem functioning and health. Nitrogen and phosphorus are essential for primary productivity, required in the process of photosynthesis (Nicholls and Dillon 1978). These nutrients are absorbed by phytoplankton, benthic algae, macrophytes and other primary producers (Nicholls and Dillon 1978), which are the base of the food web. Silica is also highly relevant to the health of a riverine ecosystem (Egge and Aksnes 1992, Conley et al. 2000). Diatoms require silica for the construction of their cell walls, thus making it an essential nutrient for diatom growth and survival (Lund 1950, Lewin 1957). Restriction of these nutrients can affect the ability for the plants and algae to grow, impacting on the rest of the food web. Conversely, excess nutrients can be problematic through overgrowth of algae. Eutrophication is a risk in aquatic environments where excess amounts of nutrients lead to excessive aquatic plant growth, and commonly algal blooms (Reynolds and Walsby 1975). Blooms can have adverse effects on the water quality of a river or lake. The decomposing algal cells when blooms collapse can deplete the dissolved oxygen (DO) in the water column (Stringfellow et al. 2009). The release of toxins from cyanobacteria can also have adverse effects on aquatic organisms and human users of water (Paerl et al. 2001).

Environmental disturbances can affect the longitudinal movement and cycles of nutrients in an aquatic environment (Ward and Stanford 1983). One such disruption is the presence of a large reservoir, which can impact the nutrients within a river by several means. Dams can prevent nutrient-rich particles from travelling to downstream river reaches, resulting in low nutrient levels downstream (Burford et al. 2012). Thus through sedimentation, reservoirs act as a sink to nutrients including nitrogen, phosphorus and silica, preventing passage to downstream stretches of river (Conley et al. 2000, Humborg et al. 2000, Kunz et al. 2011a, Kunz et al. 2011b). Also, during thermal stratification in summer an oxycline often develops in conjunction with a thermocline, with the hypolimnion often becoming anoxic (Welsh and Eller 1991). This results in the reduction of the redox potential at the sediment-water interface and the subsequent release of nutrients such as phosphorus, nitrogen and iron from sediments into the water column (Mortimer 1941, 1971, Amirbahman et al. 2003, Beutel 2006, Baldwin and Williams 2007, Baldwin et al. 2010). Reservoirs with low-level outlets can release anoxic water with high concentrations of nutrients and metals and low silica (Conley et al. 2000, Marshall et al. 2006). Therefore, reservoirs can alter the nutrient flux to downstream stretches of river, acting as a sink during cool seasons of lake mixing, and a source for nitrogen, phosphorus and iron during stratification in warm months.

The downstream nutrient effects of hypolimnial releases can be reduced in one of two ways; destratification or selective withdrawal (Sherman 2000). Destratification of a reservoir breaks down the thermocline and oxycline. Aeration of the hypolimnion eliminates bacteria-mediated reduction processes, and the subsequent release of sediment-bound nutrients into the water column (Beutel 2006). One method commonly used to achieve this is the installation of air compressors to create bubble plumes from the bottom waters to the surface. The bubbles oxygenate the water and entrain hypolimnetic water to the surface, inducing mixing (Miles and West 2011, Fernandez et al. 2012). This is often unsuitable for large reservoirs due to the energy and associated costs required for the operation and maintenance of the system (Sherman 2000).

Alternatively, selective withdrawal allows for water to be drawn from a specific region of the water column, allowing surface waters with lower nutrient levels to be released downstream instead of nutrient-rich hypolimnial water. The use of a multiple level off-take (MLO) tower is less energy-intensive than destratifying a large reservoir. MLO's can be time consuming and costly to manage, often taking at least two workers and at least one working day to change off-take levels. Furthermore, the cost of retrofitting a dam with an MLO is expensive, for example, to retrofit Burrendong Dam would cost around $25 million in 2000, approximately equivalent to just over $37 million in 2015 (Sherman 2000).

The Macquarie River flows west through central NSW, providing irrigation and town water supply to inland towns and cities in western NSW. Burrendong Dam is located on the Macquarie River and has a capacity of 1,188 gigalitres. Thermal stratification occurs on an annual basis every summer when high volume irrigation releases are drawn from the hypolimnion through a low level off-take. CWP is a problem below Burrendong Dam (Lugg 1999). However, there is little in the literature about the associated problems such as how nutrient concentrations in the Macquarie River are affected by hypolimnial releases. Such releases have been found to alter nutrient concentrations in other rivers worldwide (Byren and Davies 1989, Marshall et al. 2006) and therefore it is possible that releases from Burrendong Dam could be affecting the Macquarie River in a similar way. If there is an increase in nutrient concentrations in the downstream Macquarie River associated with hypolimnial releases from the dam, the release of surface waters through the use of the thermal curtain may reduce this.

Three specific hypotheses are addressed in this chapter:

1. During mixing, nutrient concentrations and DO will be homogenous through depth in the reservoir and during stratified periods the hypolimnion will have greater nutrient concentration.
2. Nutrient concentrations will be greater downstream than upstream of the dam during stratified periods with low-level releases.
3. Downstream nutrient concentrations during reservoir stratification will be reduced with epilimnial releases compared to hypolimnial releases after the curtain is implemented.

4.2 Study sites

The sites used in this chapter are the same as for Chapter 3 (Temperature) and have been previously described. One sampling site exists upstream of Burrendong Dam on the Macquarie River (Long Point), one immediately downstream of the dam wall (Outlet) and another within the reservoir near the off-take tower. The Cudgegong River is a significant tributary to Burrendong Dam, however no sites located on this river were used for monitoring as the water quality is potentially altered by Windamere Dam.

4.2 Methods

4.2.1 Data collection and sample analyses

Sampling was performed between April 2013 and April 2015 at 2 sites along the Macquarie River, and one site in Burrendong Dam (Fig. 1). Long Point and the outlet were sampled on a monthly basis from April to September, and fortnightly from October to March over a period from April 2013 to March 2015. Total phosphorus (TP), total filterable phosphorus (TFP), nitrogen oxidised as N (NO_x), total nitrogen (TN), total filterable nitrogen (TFN), ammonia (NH_3), silica (SiO_2) and filterable reactive phosphorus (FRP) were collected by hand 20 cm below the surface of flowing water. Burrendong Dam depth samples were collected monthly. Nutrients were collected at depths of 0.25, 1, 3, 5, 10 and 20 m in Burrendong Dam using an 8 L Van-Dorn depth sampler. Depth profiles were sampled from a buoy located approximately 750 m from the off-take tower.

Temperature (°C), electrical conductivity (µS/cm), dissolved oxygen (% saturation and mg/L) and pH were measured with a Hydrolab Surveyor and MS5 Sonde probe. Turbidity samples (NTU) were measured using a Hach 2100 turbidimeter.

TP and TN were collected in sample rinsed 50mL PET bottles. NO_x, ammonia, silica, FRP, TFP and TFN were filtered through a 0.45 µm syringe filter using sample rinsed syringes into sample rinsed 50 ml PET bottles. All nutrient samples were frozen for storage before analysis. Nutrients were analysed at a NATA certified lab, using a segmented flow analyser (OI Analytical Model FS3100) according to standard methods (APHA 2005). TSS was sampled in sample rinsed 500 ml PET bottles and chilled until analysed according to standard methods (APHA 2005).

4.2.2 Data analysis

Monthly depth profiles of DO and nutrient analytes in Burrendong Dam were used to analyse the internal water quality dynamics of the dam. DO measurements taken at 1 m depth increments were used to create a contour map. This demonstrates the DO patterns as a two dimensional plot with depth on the y-axis, time along the x-axis and DO concentration as colour gradients. The contour map was used to visually express the annual development and breakdown of an oxycline.

DO was used to determine periods of stratification and mixing. Changes in water chemistry are expected to be observed at oxygen concentrations less than 2 mg/L (Mortimer 1971). Therefore, for the purposes of this study, periods when the water body was considered to be stratified and where there is expected to be an effect on analytes, were when DO measurements were less than 2 mg/L at 20 m depth. The water body was considered to be mixed during periods when DO concentration was greater than 4.5 mg/L at 20 m of depth.

The periods have been referred to in this study by the year in which they began. Hence, the period of mixing in 2013 included June to October 2013, and the period of stratification of 2013 was inclusive of December 2013, and February, March and April of 2014. January 2014 was excluded from the stratification calculations due to the occurrence of partial destratification which oxygenated the bottom waters to a concentration greater than the 2 mg/L threshold. The 2014 mixing period included July to September 2014, and 2014 stratification December 2014 and January 2015. Data was not available for the rest of the season as sampling had ceased after January.

Water samples were collected monthly near the dam wall at five depths (0.25, 1, 3, 5, 10 and 20 m). Mean values and standard error were calculated for DO and analyte concentrations in these time frames for each depth. DO and analyte mean values and standard error were also calculated for the outlet for the same periods of stratification and mixing. These values were compared to trigger values outlined by the Australia and New Zealand Environment and Conservation Council (ANZECC) to assess water quality within the dam and impact of surface releases on downstream water quality (ANZECC and ARMCANZ 2000).

4.3 Results

4.3.1 Seasonal dissolved oxygen dynamics within Burrendong Dam

The vertical distribution of dissolved oxygen (DO) in Burrendong Dam was homogenous from July 2013 to October 2013, and from July 2014 to September 2014. An oxycline developed within Burrendong Dam from November 2013 to June 2014, and from November 2014 to the end of the sampling period in January 2015 (Fig. 27). This oxycline often led to oxygen values below 2 mg/L in the hypolimnion, below 15 m depth from the surface.

In 2013 when the dam was mixed, mean DO showed little fluctuation in the top 10 m (Table 5). At 0.25 m depth, the DO was 8.69 ± 0.81 mg/L and declined with depth from the surface to 7.87 ± 0.65 mg/L at 10 m, and finally to 6.63 ± 0.50 mg/L at 20 m. During the period of stratification, an oxycline developed, reflected by a more marked reduction in the oxygen content of the water column with depth from the surface. At the surface, mean values were 8.31 ± 0.29 mg/L, whilst at 10 m it reduced to 4.55 ±

0.80 mg/L. Rapid decline in DO is seen from 10 m to 20 m where values were 0.13 ± 0.10 mg/L.

In 2014, DO near the surface of the water column had a mean of 8.87 ± 0.70 mg/L during the period of mixing (Table 6). This then reduced with depth to 7.21 ± 0.40 mg/L at a depth of 20 m. A more distinct change occurred with stratification which demonstrated the development of an oxycline within the water body. DO ranged from 8.29 ± 0.04 mg/L in the surface layer to 8.04 ± 0.13 mg/L at 5 m, then 0.15 ± 0.05 mg/L and 0.15 ± 5.0×10^{-03} mg/L at 10 and 20 m, respectively. Releases were sourced from the surface layer in May 2014 and onwards.

4.3.2 Burrendong Dam nutrient changes with depth during mixed and stratified conditions

Low-level releases (2013)

There was little variation with depth for most of the analytes during the mixed periods in 2013 (Table 5). At all depths, ammonia, iron, FRP, TN, TP and dissolved P were all consistent. NO_x and silica both fluctuated minimally from the water surface to 20 m, ranging from a mean of 0.182 ± 0.03 mg/L to 0.254 ± 0.03 mg/L and from 0.858 ± 0.18 mg/L to 1.240 ± 0.11 mg/L, respectively.

During stratified periods in 2013, all analytes showed some degree of accumulation within the hypolimnion, with the exception of FRP (Table 5). Ammonia increased sixteen-fold from 0.023 ± 4.9×10^{-03} mg/L at the surface to 0.323 ± 0.15 mg/L in the hypolimnion. Iron also increased in the hypolimnion, where a value of 0.133 ± 0.02 mg/L was recorded in the surface water, and 1.280 ± 0.59 mg/L at 20 m. NO_x

increased in concentration with depth from 0.017 ± 0.01 mg/L at 0.25 m to 0.090 ± 0.07 mg/L at 20 m, and silicon from 0.785 ± 0.22 mg/L in the surface water and 2.600 ± 0.70 mg/L in the hypolimnion.

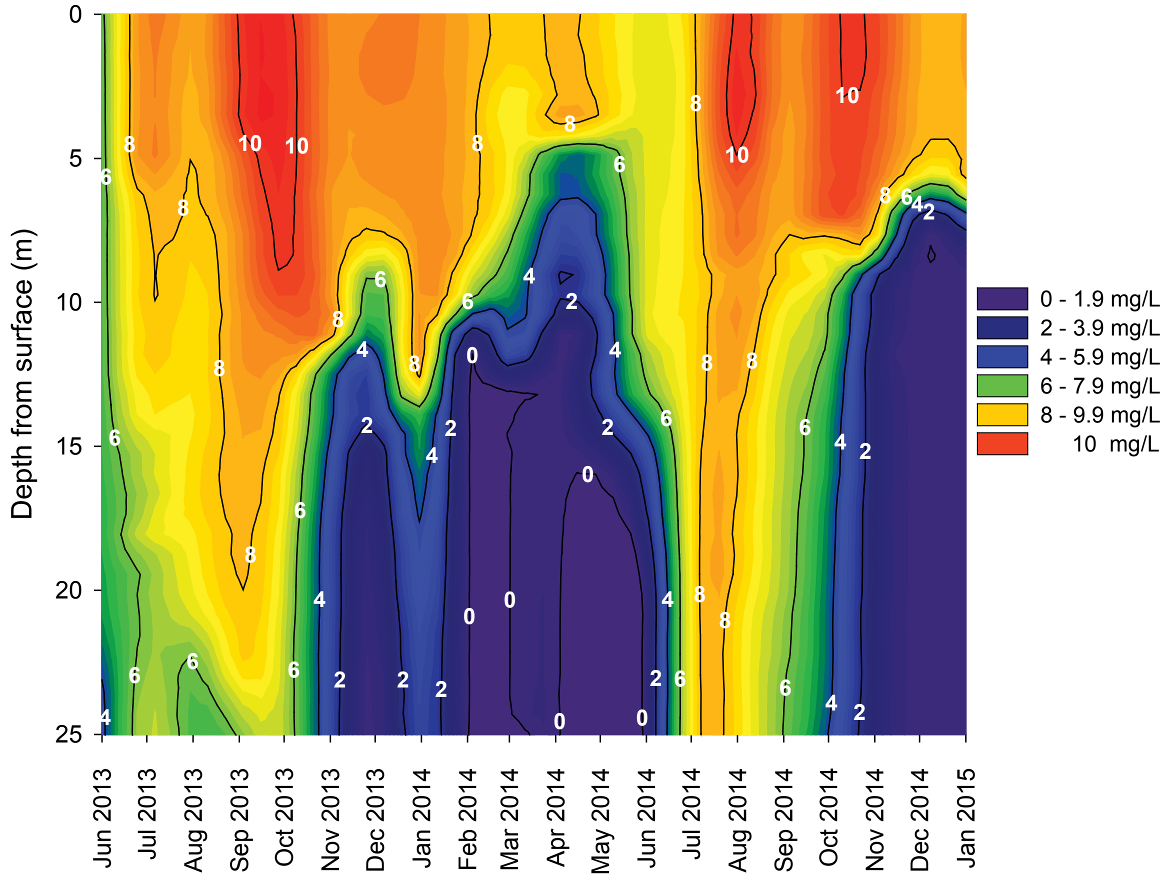


Figure 27: Heat map showing the DO (mg/L) patterns within Burrendong Dam from June 2013 to January 2015. Oxygen profiles are shown over time (x-axis) with depth (m) from the surface (y-axis).

Surface level releases (2014)

Similar to results obtained from the depth profile of 2013, little variation in analyte concentration was observed during the period of mixing in 2014 (Table 6). Observed values of ammonia, iron, TN, TFN, FRP, TP and TFP were all fairly consistent with depth. Silica varied from 0.440 ± 0.28 mg/L at 3 m to 0.700 ± 0.14 mg/L at 20 m. Observed increases in some analytes occurred during stratification and when the oxygen content of the hypolimnion was very low. Ammonia increased from concentrations below the detectable limit in surface waters to 0.143 ± 0.16 mg/L in the hypolimnion. Iron increased from 0.084 ± 0.01 mg/L at 0.25 m in depth to 1.400 ± 0.25 mg/L at 20, silica from 0.885 ± 0.03 mg/L at 3 m to 2.950 ± 0.72 mg/L at 20 m and TN from 0.435 ± $3.3x10^{-03}$ mg/L at 0.25 m to 0.630 ± 0.13 mg/L at 20 m.

4.3.3 Downstream versus upstream analyte concentrations during low-level releases

In the sampling period during low-level releases, disparities existed between analyte concentrations at the outlet and Long Point. Concentrations were greater at the outlet than Long Point for ammonia during the stratification period of the dam, TN during the mixing period, and NO_x and TFN during both periods. Greater concentrations of ammonia were found in releases measured at the outlet, which contained 0.057 ± 0.01 mg/L, compared with 0.010 mg/L at the upstream site. TN had a minor increase of 0.172 from 0.478 ± 0.16 mg/L at Long Point in comparison to 0.650 ± 0.02 mg/L at the outlet during mixing. In both periods, NO_x was greater at the outlet than Long Point by a similar magnitude. During mixing, NO_x at the outlet was 0.230 ± 0.03 mg/L compared to 0.162 ± 0.14 mg/L at Long Point, and during stratification it was 0.098 ± 0.04 mg/L at the outlet and 0.014 ± $2.4x10^{-03}$ mg/L at Long Point. TFN experienced increases of around the same magnitude at the outlet in both periods, indicating no change with dam stratification. Phosphorus in the form of FRP, TP and TFP were of similar concentrations at the outlet and Long Point during both periods of mixing and stratification.

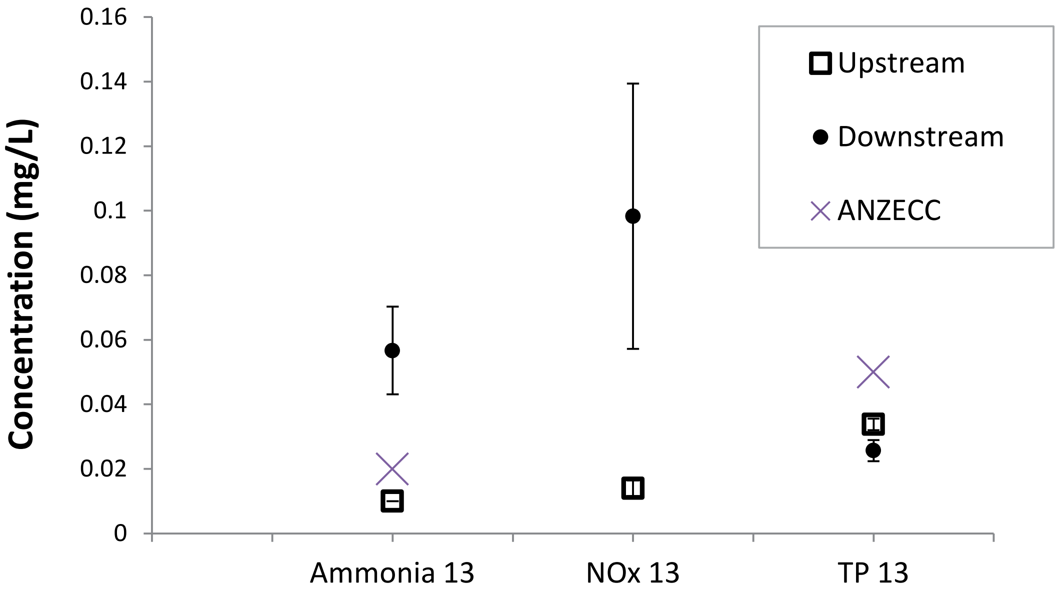


Figure 28: Mean nutrient concentrations during the period of stratification are presented for the upstream site, Long Point (square) and downstream site, the outlet (circle) with standard error bars. ANZECC trigger values are indicated by the "X". The ANZECC trigger value for NOx is 0.5 mg/L.

Silica was lower at the outlet compared with Long Point through both periods in 2013. Values observed at the outlet during mixing of the dam were 1.483 ± 0.18 mg/L, 0.849 mg/L less than Long Point, which had a silica concentration of 2.332 ± 1.03 mg/L. Again, a similar pattern emerged during stratification, where 1.717 ± 0.11 mg/L was observed at the outlet, compared to 5.600 ± 0.86 mg/L at Long Point; a difference of 3.883 mg/L. Iron concentrations during mixing and stratification were also greater at Long Point than at the outlet as well as for TN during stratification.

4.3.4 Downstream analyte values during reservoir stratification: surface release versus low-level release

Low-level releases in 2013 had little effect on DO values recorded at the outlet (Table 5). During the period of mixing, the DO measured at the outlet was 8.99 ± 0.66 mg/L and during stratification 8.22 ± 0.47 mg/L, regardless of the oxygen values present at the release depth of 20 m. In both mixed and stratified periods with surface releases, the oxygen values at the outlet did not appear to reflect DO measurements of the surface water of the dam. Outlet DO values during mixing were 10.52 ± 0.20 mg/L compared to 8.87 ± 0.70 mg/L at the surface, whereas they were 6.67 ± 0.82 mg/L during stratification, where 8.29 ± 0.04 mg/L was recorded within the surface water.

Changes of the concentrations of analytes in releases, sampled at the outlet, occurred from 2013 to 2014 (Tables 5 and 6). Ammonia, NO_x and TN exhibited no change between the two years during the mixing periods. During stratification, water from surface releases recorded at the outlet in 2014 saw a reduction in the concentration of the same analytes compared with low level releases in 2013. Ammonia concentrations reduced from 0.057 ± 0.01 mg/L in 2013 to 0.023 ± 0.01 mg/L in 2014, there was a decrease 0.098 ± 0.04 mg/L to 0.020 ± 0.01 mg/L in NO_x concentration and from 0.545 ± 0.05 mg/L to 0.460 ± 0.02 mg/L in TN concentration, respectively.

TFN was also lower overall at the outlet with surface releases compared to low-level releases. Interestingly, there was an increase in TFN concentrations during mixing from 0.532 ± 0.05 mg/L in 2013 to 0.600 ± 0.03 mg/L in 2014. Phosphorus in the form of FRP, TP and TFP, all exhibited minimal fluctuations in both periods with both surface and low-level releases.

Table 5: Mean nutrient concentrations of DO, ammonia, iron, NO_x, FRP, SiO_2, TN, TFN, TP, and TFP, all in mg/L. Averages (±SE) are shown for periods of mixing within the reservoir (June – October 2013) and stratification (December 2013 – March 2014, excluding January 2014). These time frames were also used for calculations of averages for the outlet and Long Point GS.

2013	**DO**		**Ammonia**		**Iron**		**NO_x**		**FRP**	
Site	Mixed	Stratified	Mixed	Stratified	Mixed	Stratified	Mixed	Stratified	Mixed	Stratified
Dam: 0.25 m	8.69 ± 0.81	8.31 ± 0.29	0.018 ± 4.9x10^{-03}	0.023 ± 0.01	0.110 ± 0.03	0.133 ± 0.02	0.182 ± 0.03	0.017 ± 0.01	0.009 ± 1.0x10^{-03}	0.007 ± 1.5x10^{-03}
Dam: 1 m	8.67 ± 0.82	8.38 ± 0.35	0.020 ± 0.01	0.020 ± 0.00	0.096 ± 1.0x10^{-03}	0.134 ± 0.02	0.222 ± 0.02	0.013 ± 3.3x10^{-03}	0.009	0.006 ± 3.3x10^{-04}
Dam: 3 m	8.62 ± 0.84	8.10 ± 0.35	0.016 ± 4.0x10^{-03}	0.017 ± 0.01	0.120 ± 0.02	0.140 ± 0.02	0.268 ± 0.06	0.017 ± 0.01	0.007	0.006 ± 6.7x10^{-04}
Dam: 5m	8.42 ± 0.79	7.25 ± 0.92	0.026 ± 0.01	0.027 ± 3.3x10^{-03}	0.185 ± 0.09	0.150 ± 0.02	0.244 ± 0.02	0.013 ± 3.3x10^{-03}	0.007 ± 1.0x10^{-03}	0.007 ± 6.7x10^{-04}
Dam: 10 m	7.87 ± 0.65	4.55 ± 0.80	0.018 ± 3.7x10^{-03}	0.043 ± 0.03	0.121 ± 0.01	0.145 ± 0.04	0.230 ± 0.02	0.020 ±0.01	0.006 ± 0.00	0.007 ± 1.0x10^{-03}
Dam: 20 m	6.63 ± 0.5	0.13 ± 0.10	0.020 ± 0.01	0.323 ± 0.15	0.118 ± 0.01	1.280 ± 0.59	0.254 ± 0.01	0.090 ±0.07	0.006 ± 4.1x10^{-04}	0.033 ± 0.02
Outlet	8.99 ± 0.66	8.22 ± 0.47	0.023 ± 0.01	0.057 ± 0.01	0.220 ± 0.05	0.222 ± 0.04	0.230 ± 0.03	0.098 ± 0.04	0.009 ± 8.8x10^{-04}	0.011 ± 1.5x10^{-03}
Long Point	9.45 ± 0.34	10.03 ± 0.56	0.010	0.010	0.544 ± 0.20	0.586 ± 0.11	0.162 ± 0.14	0.014 ± 2.4x10^{-03}	0.007 ± 5.8x10^{-04}	0.007 ± 3.3x10^{-04}
	SiO_2		**TN**		**TFN**		**TP**		**TFP**	
Site	Mixed	Stratified	Mixed	Stratified	Mixed	Stratified	Mixed	Stratified	Mixed	Stratified
Dam: 0.25 m	0.858 ± 0.18	0.785 ± 0.22	0.596 ± 0.05	0.500 ± 0.00	0.490 ± 0.06	0.490 ± 0.03	0.018 ± 2.3x10^{-03}	0.022 ± 4.6x10^{-03}	0.020 ± 3.5x10^{-03}	0.014 ± 1.7x10^{-03}
Dam: 1 m	0.964 ± 0.20	0.830 ± 0.27	0.676 ± 0.03	0.617 ± 0.05	0.606 ± 0.05	0.623 ± 0.03	0.021 ± 3.0x10^{-03}	0.021 ± 1.2x10^{-03}	0.018 ± 1.7x10^{-03}	0.014 ± 1.5x10^{-03}
Dam: 3 m	1.010 ± 0.25	0.750 ± 0.18	0.696 ± 0.03	0.587 ± 0.03	0.590 ± 0.06	0.517 ± 0.02	0.019 ± 2.4x10^{-03}	0.020 ± 6.7x10^{-04}	0.017 ± 1.0x10^{-03}	0.013 ± 1.2x10^{-03}
Dam: 5m	1.176 ± 0.33	0.765 ± 0.24	0.604 ± 0.06	0.637 ± 0.04	0.580 ± 0.06	0.557 ± 0.07	0.018 ± 1.1x10^{-03}	0.024 ± 3.1x10^{-03}	0.017 ± 2.9x10^{-04}	0.015 ± 0.00
Dam: 10 m	1.072 ± 0.20	0.847 ± 0.38	0.660 ± 0.06	0.533 ± 0.08	0.618 ± 0.05	0.500 ± 0.08	0.019 ± 2.7x10^{-03}	0.019 ± 2.0x10^{-03}	0.017 ± 2.8E^{-03}	0.023 ± 0.00
Dam: 20 m	1.240 ± 0.11	2.600 ± 0.7	0.686 ± 0.02	0.817 ± 0.06	0.662 ± 0.01	0.807 ± 0.06	0.021 ± 2.7E^{-03}	0.098 ± 0.04	0.018 ± 2.6E^{-03}	0.089 ± 0.07
Outlet	1.483 ± 0.18	1.717 ± 0.11	0.650 ± 0.02	0.545 ± 0.05	0.532 ± 0.05	0.527 ± 0.03	0.022 ± 2.3x10^{-03}	0.026 ± 3.3x10^{-03}	0.020 ± 2.0x10^{-03}	0.019 ± 1.7x10^{-03}
Long Point	2.332 ± 1.03	5.600 ± 0.86	0.468 ± 0.16	0.832 ± 0.24	0.430 ± 0.19	0.446 ± 0.04	0.025 ± 3.7x10^{-03}	0.034 ± 1.8x10^{-03}	0.018 ± 1.8x10^{-03}	0.016 ± 1.2x10^{-03}

Table 6: Mean nutrient concentrations of DO, ammonia, iron, NO_x, FRP, SiO_2, TN, TFN, TP, and TFP, all in mg/L. Averages (±SE) are shown for periods of mixing within the reservoir (July - September 2014) and stratification (December 2014 – March 2015, excluding February 2015 due to missing data). These time frames were also used for calculations of averages for the outlet and Long Point GS.

2014	**DO**		**Ammonia**		**Iron**		**NO_x**		**FRP**	
Site	Mixed	Stratified	Mixed	Stratified	Mixed	Stratified	Mixed	Stratified	Mixed	Stratified
Dam: 0.25 m	8.87 ± 0.70	8.29 ± 0.04	0.015 ± 1.8x10^{-03}	Less than detection limit	0.080	0.084 ± 0.01	0.240 ± 0.03	0.008 ± 1.2x10^{-03}	0.008 ± 1.0x10^{-03}	Less than detection limit
Dam: 1 m	8.87 ± 0.75	8.30 ± 0.04	0.023 ± 0.01	0.015 ± 1.0x10^{-03}	0.092	0.110 ± 0.02	0.257 ± 0.06	0.029 ± 0.01	0.007 ± 5.0x10^{-04}	Less than detection limit
Dam: 3 m	8.91 ± 0.83	8.22 ± 0.03	0.022 ± 0.01	0.018 ± 0.01	0.070 ± 0.02	0.097 ± 0.01	0.273 ± 0.04	0.013 ± 3.2x10^{-03}	0.006 ± 1.0x10^{-03}	Less than detection limit
Dam: 5m	8.65 ± 0.69	8.04 ± 0.13	0.015 ± 1.9x10^{-03}	0.015 ± 1.5x10^{-03}	0.050	0.100 ± 0.01	0.273 ± 0.03	0.019 ± 2.9x10^{-03}	0.006 ± 5.0x10^{-04}	Less than detection limit
Dam: 10 m	7.82 ± 0.41	0.15 ± 0.02	0.020 ± 4.1x10^{-03}	0.031 ± 0.10	0.073 ± 0.02	0.180 ± 0.02	0.267 ± 0.02	0.035 ± 0.01	0.005 ± 0.00	0.009 ± 0.01
Dam: 20 m	7.21 ± 0.40	0.15 ± 5.0x10^{-03}	0.027 ± 0.01	0.143 ± 0.16	0.080	1.400 ± 0.25	0.297 ± 3.3x10^{-03}	0.051 ± 0.03	0.008	0.012 ± 0.02
Outlet	10.52 ± 0.20	6.67 ± 0.82	0.012 ± 1.0x10^{-03}	0.023 ± 0.01	Less than detection limit	0.185 ± 0.03	0.230 ± 0.03	0.020 ± 0.01	0.006 ± 5.0x10^{-04}	0.009 ± 1.9x10^{-03}
Long Point	10.68 ± 0.23	9.20 ± 0.21	Less than detection limit	0.015 ± 2.6x10^{-03}	0.180 ± 0.04	0.638 ±0.22	0.063 ± 0.06	0.007 ± 1.5x10^{-03}	0.006 ±1.0x10^{-03}	0.005
	SiO_2		**TN**		**TFN**		**TP**		**TFP**	
Site	Mixed	Stratified	Mixed	Stratified	Mixed	Stratified	Mixed	Stratified	Mixed	Stratified
Dam: 0.25 m	0.605 ± 0.40	0.995 ± 0.08	0.660 ± 0.01	0.435 ± 3.3x10^{-03}	0.670 ± 0.02	0.400 ± 0.03	0.013 ± 1.5x10^{-03}	0.017 ± 8.8x10^{-04}	0.014 ± 2.5x10^{-03}	0.009 ± 6.7x10^{-04}
Dam: 1 m	0.615 ± 0.39	0.990 ± 0.09	0.793 ± 0.11	0.565 ± 0.04	0.663 ± 0.06	0.580 ± 0.04	0.017 ± 2.8x10^{-03}	0.018 ± 1.2x10^{-03}	0.010 ± 8.8x10^{-04}	0.012 ± 1.8x10^{-03}
Dam: 3 m	0.440 ± 0.28	0.885 ± 0.03	0.783 ± 0.03	0.580 ± 0.05	0.753 ± 0.06	0.520 ± 0.04	0.017 ± 1.2x10^{-03}	0.020 ± 2.2x10^{-03}	0.011 ± 1.2x10^{-03}	0.011 ± 1.2x10^{-03}
Dam: 5m	0.457 ± 0.25	1.000 ± 0.08	0.803 ± 0.10	0.480 ± 0.01	0.697 ± 0.01	0.490 ±0.02	0.016 ± 6.7x10^{-04}	0.018 ± 1.2x10^{-03}	0.010 ± 1.5x10^{-03}	0.010 ± 5.8x10^{-04}
Dam: 10 m	0.600 ± 0.20	1.525 ± 0.68	0.753 ± 0.05	0.605 ± 0.06	0.633 ± 0.05	0.590 ±0.06	0.020 ± 3.9x10^{-03}	0.020 ± 0.01	0.010 ± 1.2x10^{-03}	0.015 ± 0.01
Dam: 20 m	0.700 ± 0.14	2.950 ± 0.72	0.763 ± 0.02	0.630 ± 0.13	0.740 ± 0.01	0.610 ±0.14	0.016 ± 8.8x10^{-04}	0.043 ± 0.04	0.010 ± 8.8x10^{-04}	0.033 ± 0.06
Outlet	0.527 ± 0.22	2.380 ± 0.87	0.657 ± 0.03	0.460 ± 0.02	0.600 ± 0.03	0.448 ± 0.05	0.018 ± 1.5x10^{-03}	0.024 ± 1.4x10^{-03}	0.010 ± 5.8x10^{-04}	0.014 ± 1.9x10^{-03}
Long Point	2.900 ± 1.19	15.460 ± 6.94	0.430 ± 0.12	0.562 ± 0.07	0.343 ± 0.08	0.460 ± 0.05	0.023 ± 0.01	0.031 ± 4.2x10^{-03}	0.011 ± 1.7x10^{-03}	0.011 ± 1.0x10^{-03}

4.4 Discussion

4.4.1 Nutrient dynamics within Burrendong Dam during mixing and stratification

It was expected that during mixing, the oxygen and nutrient content of the water column would be homogenous. Results indicate that this was the case in both 2013 and 2014 (Tables 1 and 2). When the water body had undergone mixing during the cooler months, all depths of the water column were oxygenated. Overall, the analyte concentrations were fairly consistent throughout the water body, as expected (Preece and Jones 2002). There was an exception for silica in both years, which exhibited a substantial increase from 0.858 ± 0.18 mg/L to 1.240 ± 0.11 mg/L in 2013, and from 0.440 ± 0.28 mg/L at 3 m to 0.700 ± 0.14 mg/L in 2014. This pattern exhibited by silica of increasing concentration with depth is possibly a result of sinking diatom cells (Ptacnik et al. 2003). Typical algal community succession within a lake shows diatoms to dominate in periods of mixing, and some species sink during water column stability with thermal stratification (Ptacnik et al. 2003). It is possible diatom losses also occur to a lesser degree during mixing, accounting for greater silica concentrations at 20 m compared to other depths. Sinking occurs because these cells do not have apparatus to support propulsion or gas vacuoles to actively control their buoyancy and position in the water column (Garnier et al. 1995).

The development of anoxia in the hypolimnion in the summer of both years was likely to be a major factor in the increased concentrations of ammonia, iron, FRP, silica, TN, TFN, TP and TFP in 2013 and increases in concentrations of ammonia, iron, silica and TN with depth within the reservoir. Oxygen concentration within a body of water can influence the oxidation-reduction potential. Anoxic conditions at the sediment-water interface can result in an environment favourable to the release of sediment-bound nutrients and ions (Mortimer 1941, 1971). This can lead to an increase in the concentration of phosphorus, iron, and nitrogen in hypolimnial waters. This process is a key factor in the internal loading of nutrients and metals.

The dissolved oxygen concentration of a lake is required to reach below 1 or 2 mg/L at the sediment-water interface before a significant change in chemistry in the overlying waters is likely to be observed (Mortimer 1971). This circumstance was met in Burrendong Dam at 20 m depth in 2013, and from 10 to 20 m in 2014. Ferric substances (FE(III)) are reduced to soluble iron (FE(II)) ions under these conditions. When ferric-phosphates are reduced, the phosphates and iron are released into the water column (Mortimer 1941, Stumm and Morgan 1996, Amirbahman et al. 2003). Iron was almost 10 times greater in concentration in the hypolimnion than surface waters and phosphorus was 5 times greater. Ammonia and silica also showed substantial increases in concentration. These concentration changes only occurred to this extent during periods when oxygen concentration was less than 2 mg/L. A study involving 12 lake cores found that all sites released ammonia in an anoxic environment, with little released in oxygenated conditions (Beutel 2006).

4.4.2 Low-Level versus surface level releases: Effect of the thermal curtain on internal nutrient dynamics

An oxycline appears to develop more distinctly during stratification with epilimnial releases compared to the prior year with hypolimnial releases. In 2014, DO concentrations of less than 2 mg/L occurred in the water column from a depth of approximately 7 m from the surface, and extended to the sediment layer. In 2013, the depth affected by low oxygen concentrations did not extend any closer to the surface than 10 m, where at this depth the mean DO was 4.55 ± 0.80 mg/L for the period of stratification. This may be due to the release of water from the epilimnion causing the depth of the anoxic zone to rise toward the surface water (Nurnberg 2007, Caliskan and Elci 2009, Zouabi-Aloui et al. 2015) . Epilimnial releases may also increase the duration of the anoxic event, sometimes by several weeks, as a result of increased residence time of the hypolimnion (Zouabi-Aloui et al. 2015). Nutrients and metals can accumulate as a result of this longer duration of anoxia, and also due to the lack of hypolimnial flushing which occurs with low-level releases (Nurnberg 2007). Overall, the analyte concentrations were lesser in 2014 than in 2013.

4.4.3 Downstream versus upstream analyte concentrations during low-level releases

It was hypothesised that the analyte concentrations at the outlet during low-level releases in 2013 would be greater than the upstream site during periods of stratification. Ammonia, NO_x, and TFN were found to have a greater concentration at the outlet compared to Long Point during stratification in 2013. However, this was also the case for NO_x and TFN during periods of mixing. This is likely due to the depth of withdrawal and the coinciding analyte values during this time. The depth of the off-take for much of the duration of the study was from 10-15 m below the surface. When considering this, the nutrient concentrations reflect the depth of withdrawal (Table 1). Previous studies have illustrated similar findings of increased nutrient concentrations downstream of reservoirs with hypolimnial releases (Camargo and de Jalon 1990, Camargo et al. 2005, Baldwin et al. 2010). Higher conductivity, alkalinity, and hardness values were measured downstream of the Burgomillodo Reservoir compared to reference sites during periods of hypolimnial releases (Camargo and de Jalon 1990). This was attributed to the release of ions such as calcium, magnesium, manganese and bicarbonate from the sediment-water interface in the hypolimnion, which was then released downstream. Further, nitrate, ammonium and phosphate were found to be at greater concentrations downstream of four reservoirs in central Spain, causing eutrophication in downstream rivers (Camargo et al. 2005). The Hume Dam in Australia was also found to be a net exporter of carbon, nitrogen, phosphorus and iron due to low-level releases during periods of thermal stratification (Baldwin et al. 2010).

Throughout the duration of this study, the residence level of the dam has been relatively low, rarely exceeding 20% capacity. It is anticipated that with a greater residence level, the oxycline would be further from the off-take and downstream analyte concentrations would therefore more closely reflect the concentrations observed in the hypolimnion.

4.4.4 Downstream analyte concentrations during stratification: surface release versus low-level release

It was expected that downstream nutrient concentrations during reservoir stratification would be reduced with epilimnial releases compared to hypolimnial releases, after implementation of the thermal curtain. There was a decrease in concentrations of ammonia, oxidised nitrogen and TN in the dam releases with the change from hypolimnial to surface releases, during stratification periods from 2013 to 2014. When a reservoir is stratified, accumulation of nutrients in the hypolimnion is common, with lower concentrations observed in the epilimnion. Thus by channelling the surface layer containing low nutrient concentrations in 2014, the values were decreased at the outlet compared to 2013 when hypolimnial waters were released.

When considering internal reservoir water quality, hypolimnial releases have proven to be favourable over surface releases (Fontane et al. 1981, Nurnberg 2007). Low-level withdrawals cause a decrease in the duration and depth of anoxia, reducing the accumulation of nutrients and metals within the hypolimnion. However, the downstream effects of bottom-water discharge have been acknowledged as undesirable for the ecological health and anthropogenic use of river water. Such downstream impacts include eutrophication, unnatural thermal regime, oxygen depletion and the development of an unpleasant odour (Marshall et al. 2006, Nurnberg 2007).

When hypolimnial waters are released through bottom level outlets, the stretch of river directly downstream is often exposed to anoxic waters (Amirbahman et al. 2003, Marshall et al. 2006). In 2013 and 2014 with low-level releases, the oxygen saturation of the water recorded at the outlet did not reflect the anoxia observed within the hypolimnion during stratified periods. Daily discharge of hypolimnial releases from the Burgomillodo Reservoir in Spain directly resulted in low oxygen concentrations immediately downstream of the dam (Camargo and de Jalon 1990). The study identified photosynthetic activity to be directly involved in the downstream recovery of oxygen values. Further, hypolimnetic releases to the receiving streams downstream from two impoundments in the USA frequently contained dissolved oxygen levels of 5-6 mg/L, reflecting aeration within the discharge structure (Marshall et al. 2006). It is likely that substantial aeration of the water within the outlet works and valve release structure of Burrendong Dam is responsible for the greater DO concentrations in hypolimnial releases. Therefore, the probability of DO contributing to poor water quality below Burrendong Dam is low.

4.4.5 Biological implications

The concentrations of nutrients within and released from the dam need to also be considered in terms of their impact on aquatic environments. Trigger values outlined by ANZECC indicate the point whereby environmental disturbance is likely to occur due to high levels of nutrients and are a useful comparison. Within the dam, increases in nutrient concentrations were seen in the hypolimnion during thermal stratification.

All concentrations observed in the water column of Burrendong Dam for ammonia, NO_x, FRP, TN and TP in 2013 breached ANZECC trigger values for water quality of a reservoir. Though concentrations within the reservoir were generally lower in 2014, all of these nutrients at all depths were still greater than the trigger values, with the exception of FRP, which was below the detection limit from 0.25 to 5 m in depth. The trigger value for ammonia and NO_x is 0.01 mg/L, and for TN is 0.35 mg/L. The greatest observed concentrations of these nutrients in 2014 were 0.143 ± 0.16, 0.297 ± 3.3x10-03 and 0.803 ± 0.10, respectively. Additionally, the trigger values for TP and FRP in a reservoir are 0.01 mg/L and 0.005 mg/L. The greatest observed mean values in 2014 were 0.043 ± 0.04 mg/L and 0.012 ± 0.02 mg/L, respectively. The greatest mean concentration for these nutrients with the exception of NO_x occurred during periods of stratification. These results indicate the nutrient concentrations within the lake are a point of concern for environmental disturbance.

ANZECC trigger values were exceeded at the river sites of Long Point and at the outlet frequently in both 2013 and 2014. Despite reductions in concentrations at the outlet with the use of the thermal curtain in 2014, the nutrients values continued to exceed ANZECC trigger values, with the exception of ammonia and TP during mixed periods. The trigger value for ammonia, NO_x and TN in an upland river are 0.013, 0.015 and 0.25 mg/L, respectively, and the trigger values for FRP are 0.02 mg/L and TP are 0.015 mg/L. In 2014, the greatest mean value of ammonia, NO_x, TN, FRP and TP for mixed and stratified periods seen at the outlet were 0.023 ± 0.01, 0.023 ± 0.03, 0.657 ± 0.03, 0.009 ± 0.9×10^{-03}, and 0.024 ± 1.4×10^{-03} mg/L, respectively.

Taking the ANZECC guidelines into consideration, the fluctuations in all analytes at all sites over the duration of the study could have adverse ecological impacts. Similar ranges in nutrient concentrations were found in another study based in the Macquarie Marshes (Kobayashi et al. 2011), indicating the eutrophication in the Macquarie River extends as far downstream as the RAMSAR protected wetland nature reserve, potentially disturbing the health of the environment. The eutrophication may be a result of the land use of the catchment, which primarily used for agricultural grazing and cropping (Green et al. 2011). These concentrations may support excessive algal growth, including potentially toxic cyanobacteria if downstream conditions are optimal, such as with low river discharge, making these nutrients a concern for dam management.

4.5 Conclusions

During mixing, nutrient and DO concentrations were homogenous through depth in the reservoir in both years. When the reservoir was stratified in 2013 and 2014, the hypolimnion saw increases in the concentrations of nutrients when the oxygen values were less than 2 mg/L, supporting hypothesis 2a. These increases in nutrients do not appear to be further affected by the thermal curtain. The downstream nutrient concentrations were higher than upstream concentrations during periods of low-level releases when the dam was stratified, confirming hypothesis 2b. There was a reduction in nutrient concentrations at the outlet after sourcing surface water for releases with the use of the thermal curtain, though these values were often still greater than the upstream site, supporting hypothesis 2c. ANZECC trigger values that indicate ecological disturbance were exceeded by the upstream reference site, the outlet site and through various depths within in reservoir, thus concern for eutrophication and disruption for ecological health of the Macquarie River is considerable. It is recommended that monitoring of nutrient and iron concentrations continues, especially during periods when the storage is full.

Chapter 5: Phytoplankton

5.1 Introduction

Phytoplankton are fundamental constituent of riverine ecosystems. They often form the major primary producer group and make a large contribution to the oxygen status of a river (Oliver and Merrick 2006). Phytoplankton also play a critical role in the biological cycling of nutrients including nitrogen, phosphorus and silicon (Nyholm 1978, Conley et al. 2000, Oliver and Merrick 2006), and form an important part of the food web (Carpenter et al. 1985). Some phytoplankton species favour lakes while others have adaptations better suited to riverine environments (Fraisse et al. 2013).

Anthropogenic alteration of a river system through the construction of a dam can turn a flowing segment of a river (lotic system) into a lake (lentic system). Damming modifies hydrodynamics, water clarity and nutrient content of downstream river reaches, and homogenises downstream water temperatures (Baxter 1977, Marshall et al. 2006, Baldwin et al. 2010). Dam releases may also dictate the phytoplankton community composition in the downstream river reaches, as lentic populations are released from the reservoir (Coutant 1963). As a result of this, the community structure and longitudinal distribution of phytoplankton within rivers is altered (Coutant 1963, Petts 1985, Zalocar de Domitrovic et al. 2007).

Thermal stratification is common for about six months of the year in large reservoirs during the warm months in south-eastern Australia. The warm water temperatures and high light availability in the epilimnion of stratified reservoirs provides optimal conditions for phytoplankton growth. The hypolimnion on the other hand, is usually considerably cooler than the epilimnion and is often below the euphotic zone. The lack of mixing during this time typically results in high phytoplankton density in the epilimnion and low density in the hypolimnion. When hypolimnial water is released from a dam, there may be a decrease in chlorophyll *a* directly below dams in comparison to upstream river stretches (Coutant 1963).

Rivers and dams provide differing environmental conditions and as such each system favours different species of phytoplankton. The turbulent flows and varying light conditions associated with lotic systems select for species with large, round or elongated shaped cells (Wehr and Descy 1998, Coles and Jones 2000) and species that have a faster growth rate, like diatoms (Kohler 1994). The still and often clear waters of lentic systems select for species that can actively regulate their position in the water column to receive more light (Mitrovic et al. 2001). This is done by adjusting buoyancy through the use of gas vesicles, mucilage content, or through active motility permitted with the use of a flagellum (Fraisse et al. 2013). These species often do not compete well in lotic environments as they have slow growth rates (Kohler 1994).

Cyanobacteria are common in slow-flowing, stratified water bodies, such as reservoirs. Some strains, such as *Anabaena*, *Oscillatoria* and *Microcystis* can release cyanotoxins, which can lead to human health issues, fish kills, death of wildlife and livestock (WHO (World Health Organization) 2003). Cyanobacteria actively control their buoyancy via gas vesicles and spend more time in the surface, euphotic waters compared to non-motile, non-buoyant phytoplankton giving them a competitive advantage under stratified conditions (Mitrovic et al. 2001). Some species are also capable of nitrogen fixation, meaning nitrogen limitation in the epilimnion can give these species a competitive advantage over non-nitrogen fixing species (Kohler 1994). Surface releases from dams may contribute high levels of potentially harmful cyanobacteria to downstream receiving waters.

Burrendong Dam is located on the Macquarie River and sometimes has cyanobacterial blooms. In 1998 a major bloom occurred on the Macquarie River when very low dam levels and a bloom in the dam led to high concentrations transported for over 100 kilometres downstream causing a major incident (NSW DPI 2015). There is a high water demand by irrigated agriculture in downstream areas which requires high volume releases from the dam in spring and summer. These releases have previously been drawn from the hypolimnion through the reservoir's low-level bottom off-take.

In an innovative global first, a temperature control structure, a curtain with a cylindrical shaped design using flexible polypropylene fabric is attached to the outlet and extends up to 3-10m below the water surface. By releasing the surface water of Burrendong Dam whilst the water is stratified, there is a substantial risk of cyanobacteria being released to the downstream Macquarie River. This study aims investigate how the phytoplankton density and cyanobacterial content of phytoplankton communities is affected by the surface releases compared to low level releases.

Two hypotheses were tested:

1- Low chlorophyll *a* concentration downstream of the dam will occur during hypolimnial releases.
2- Chlorophyll *a* concentrations and cyanobacteria densities will increase downstream of the dam with epilimnial releases.

5.2 Study sites

The sites used in this chapter are the same as for Chapter 3 (Temperature) and have been previously described. Two sampling sites (Long Point and Bruinbun) were situated upstream of Burrendong Dam on the Macquarie River. Eight sites are located intermittently along the Macquarie River downstream of Burrendong Dam, with the first located at the outlet and the final site located 152 km downstream at Rawsonville Bridge. The Cudgegong River is a significant tributary to Burrendong Dam, however no sites located on this river were used for monitoring as the water temperature is influenced by Windamere Dam.

5.3 Methods

5.3.1 Sample collection

Phytoplankton samples were collected in 250 ml PET bottles and preserved with 2 ml Lugol's iodine according to standard method 10200B (Kantoush et al. 2008) for identification to genus level. For the determination of phytoplankton biomass as Chlorophyll a, a volume of 500 mls at each site (Fig. 1) was filtered using a Buchner Funnel and Filter Flask onto a GFC filter. Filter papers were placed immediately into 10 ml centrifuge tubes, wrapped in foil and chilled to prevent deterioration of the pigment. The samples were chilled in a powered car fridge or esky for a maximum of 6 hours until a freezer was accessible.

5.3.2 Chlorophyll a extraction

Filters containing phytoplankton pigments were stored in a frozen state until the point of chlorophyll a extraction. Chlorophyll *a* pigment was analysed using 90% ethanol extraction as described by ((Sartory and Grobbelaar 1984). In the lab, 10 ml of 90% ethanol was added to each sample before they were heated in a water bath at 75°C. Extracts were centrifuged at 1000 rpm for ten minutes before reading the absorbance at 750 and 655 nm against a 90% ethanol blank in a UV/VIS Spectrophotometer. Chlorophyll a was then calculated using the formula:

$$CHL - A\ \left(\frac{ug}{L}\right) = \frac{A \times Ve}{Kc \times Vs \times D}$$

Where:

A = A(665) – A(750)
Ve = volume of extract (ml)
Kc = 0.000082 L/µg/cm (spectral absorbance coefficient for chlorophyll *a* in 90% ethanol)
Vs = volume of sample (ml)
D = path length of optical cell (cm)

5.3.3 Algae identification

Samples were stored at room temperature, out of direct sunlight. In the laboratory, 50 ml samples were settled in measuring cylinders and the surface 40 ml was gently removed using a syringe and fine tubing. The concentrated samples were then analysed using the Sedgewick-Rafter method. 1 ml of each sample was added to the chamber of a Sedgwick-Rafter counting chamber and examined at x 200 magnification using an Olympus BX41 compound microscope. Algal cell counts were categorised into "cyanobacteria" and "other" types of algae. At least 100 units of the dominant taxa were counted in each sample, giving a precision of approximately ± 15% (Hotzel and Croome 1994).

5.3.4 Data analysis

Effect of hypolimnial and epilimnial releases on outlet chlorophyll a content: Outlet chlorophyll a *versus dam profile*

Long Point gauging station (GS) was selected as an upstream reference site as it is the closer of two monitoring sites to the reservoir, at 73 km upstream. Mean chlorophyll *a* with standard error for Long Point GS and outlet GS was calculated for the months of July and December, 2013 and 2014. These months were selected to assess chlorophyll *a* in winter and summer. The mean of chlorophyll *a* from the sampling completed in July 2013 (with hypolimnial releases) were plotted for comparisons with mean concentrations of the same month at the outlet to the concentration at Long Point. The mean chlorophyll *a* concentrations of July 2014 were overlayed to assess the effect of epilimnial releases on chlorophyll *a* content. This was then repeated for December of 2013 and 2014 to assess the effect of the thermal curtain during thermal stratification within the reservoir.

Chlorophyll *a* depth profiles of July 2013 and 2014 were plotted to allow a comparison of concentrations present within the mixed reservoir and at the depth of withdrawal to the values observed at the outlet, before and after thermal curtain operation. This was repeated for December 2013 and 2014 to compare the chlorophyll *a* concentrations during thermal stratification within the reservoir.

Effect of hypolimnial and epilimnial releases on seasonal longitudinal chlorophyll a concentrations

Mean chlorophyll *a* concentrations with positive and negative standard error for the seasons of winter (July – August), spring (September – November) and summer (December – February) of 2013 and 2014 were calculated for each monitoring site, encompassing a distance of 152 km downstream from the dam wall. They were plotted on a scale of distance in kilometres to assess the longitudinal patterns of chlorophyll *a*. The impacts of hypolimnial (2013) and epilimnial (2014) releases on the longitudinal phytoplankton density during the months of winter (June, July and August), spring (September, October and November) and summer (December, January and February) within the reservoir were assessed. Longitudinal recovery of chlorophyll *a* was assessed relative to the 95% confidence interval (95% CI) of upstream concentrations. A shaded region was overlayed on the plots to indicate this confidence range.

Effect of hypolimnial and epilimnial releases on the phytoplankton community structure at the outlet

Community structure is shown for cell counts performed on samples collected from the outlet GS once a month. The cell counts were plotted on to a time-series column graph. They were grouped into “cyanobacteria” and “other” categories, where full bars indicate 100% of the phytoplankton community. The drinking water guideline cyanobacterial cell count provided by the National Health and Medical Research Council (NHMRC) was overlaid on the y-axis. A blue line indicative of the point in time when the thermal curtain was installed was overlaid on the x-axis.

Data in this chapter were formatted and graphed using Microsoft Excel 2010 and SigmaPlot 12.5 software.

5.4 Results

5.4.1 Effect of hypolimnial and epilimnial releases on outlet chlorophyll a content: Outlet chlorophyll a versus dam profile

In the winter month of July, chlorophyll *a* content of the upstream site Long Point was greater than values detected at the outlet (Fig. 29a). This pattern was repeated both the year before (2013) and after (2014) the thermal curtain operation. In 2013, chlorophyll a values observed at the outlet were almost 6 µg/L lower than at Long Point, while in 2014, there was a decrease of 1.14 µg/L. The values detected at Burrendong outlet in December did not match upstream values in both sampling events (Fig. 30a). There was a difference of chlorophyll *a* concentration in 2013 of 6.08 µg/L, where 2.41 µg/L was observed at the outlet and 8.5 µg/L at Long Point. In 2014, there was a difference of 11.3 µg/L between chlorophyll *a* observed at the outlet and Long Point, where the concentration was 6 µg/L at the outlet, and 17.34 µg/L at Long Point.

The chlorophyll *a* depth profile within Burrendong Dam in July of 2013 differed to that of July 2014 (Fig. 29b). In 2013, the year prior to the thermal curtain use, algal density was generally low at less than 2 µg/L from 5 m to 20 m below the surface. Chlorophyll *a* was 4.5 µg/L at 0.25 m below the surface. In July of 2014, when the thermal curtain was in use, the chlorophyll *a* was overall homogenous throughout the water column at around 3.5 µg/L. In December of 2013 and 2014, the reservoir was thermally stratified. In December of both years, the chlorophyll *a* concentration was greater than in winter, peaking at 8.27 µg/L at 10 m in 2013, and 8.46 µg/L at 3 m in 2014 (Fig. 30b).

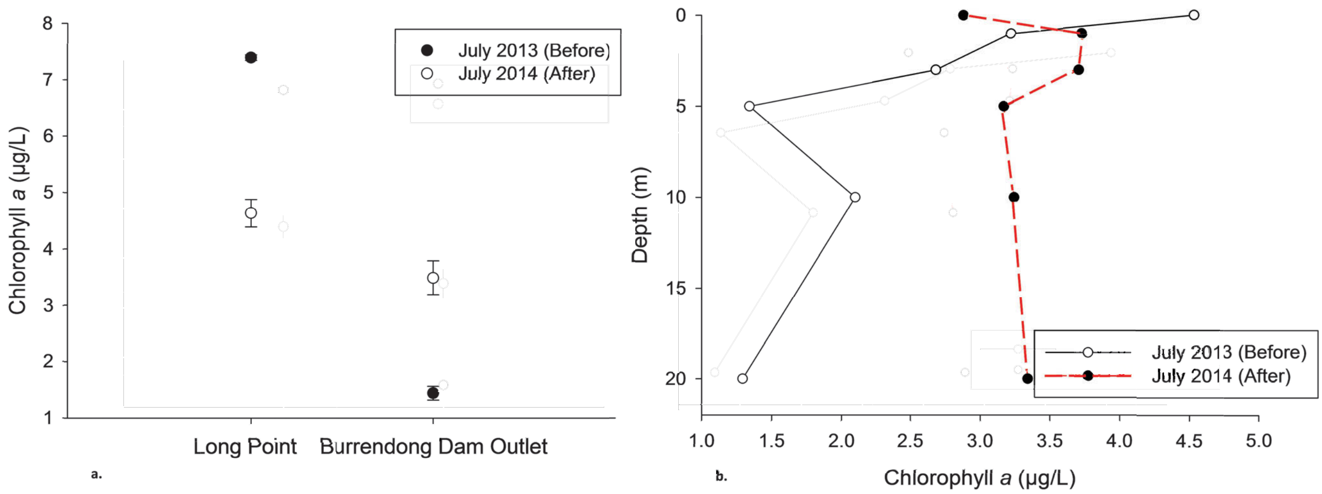


Figure 29: a. – Chlorophyll *a* of July is shown at upstream site Long Point and immediately below the dam outlet, Burrendong Dam Outlet. b. – A depth profile within Burrendong Dam shows the chlorophyll a concentrations with depth at 0.25, 1, 3, 5, 10 and 20 m. In both a. and b. data is shown for 2013 and 2014, the year before and after thermal curtain operation, respectively.

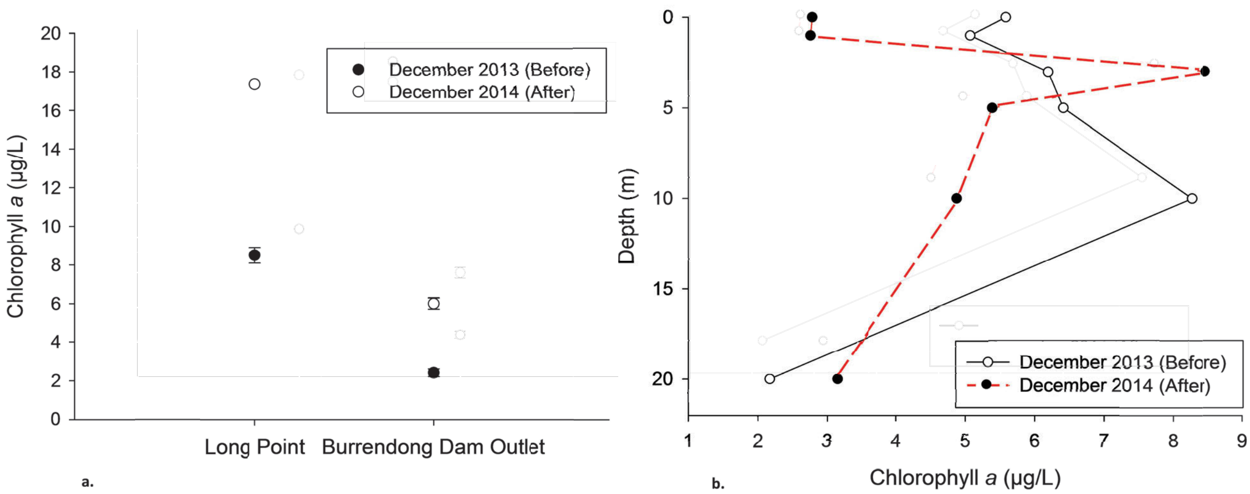


Figure 30: a. – Chlorophyll *a* of December is shown at upstream site Long Point and immediately below the dam outlet, Burrendong Dam Outlet. b. – A depth profile within Burrendong Dam shows the chlorophyll a concentrations with depth at 0.25, 1, 3, 5, 10 and 20 m. In both a. and b. data is shown for 2013 and 2014, the year before and after thermal curtain operation, respectively. Standard Error is indicated with error bars.

5.4.2 Effect of hypolimnial and epilimnial releases on seasonal longitudinal chlorophyll a concentrations

In all seasons of winter, spring and summer, there appears to be minimal impact of Burrendong Dam and subsequently, the thermal curtain on chlorophyll *a* concentration downstream of the dam wall (Fig. 31).

In winter, there is more variation in chlorophyll *a* concentration across the sites, both on a temporal and longitudinal scale (Fig. 31a). In 2013 chlorophyll *a* concentration (± 95% CI) gradually increased with distance from the dam wall, ranging from 1.4 ± 0.95 µg/L at the outlet to 35.9 ± 23.26 µg/L at Rawsonville Bridge (155 km downstream). In 2014 with epilimnial releases, a slight increase of 2.8 µg/L in 2014 is visible at the outlet compared to the year before with hypolimnial releases. The chlorophyll *a* concentration peaks at 76 km at 16.67 ± 4.35 µg/L downstream of the dam and stabilises in 2014. Observed values in spring were consistently around 2 µg/L greater in 2014 than 2013, with the exception of two sites (downstream Burrendong: 7 km, and Ponto falls: 55km downstream) (Fig. 31b).

The greatest change in chlorophyll *a* concentration between the two years was seen in summer (Fig. 31c). Algal concentration in the releases of both years were suppressed from values observed at upstream reference sites, however the chlorophyll *a* concentration increased more rapidly in 2014 with surface releases than in 2013 with hypolimnial releases.

The downstream recovery distance of phytoplankton density was measured against the 95% confidence interval range of the chlorophyll *a* concentration calculated for the upstream reference sites, Bruinbun (104 km upstream) and Long Point (73 km upstream). In all three seasons, the downstream longitudinal chlorophyll *a* concentrations all fell within the confidence interval range of the upstream sites.

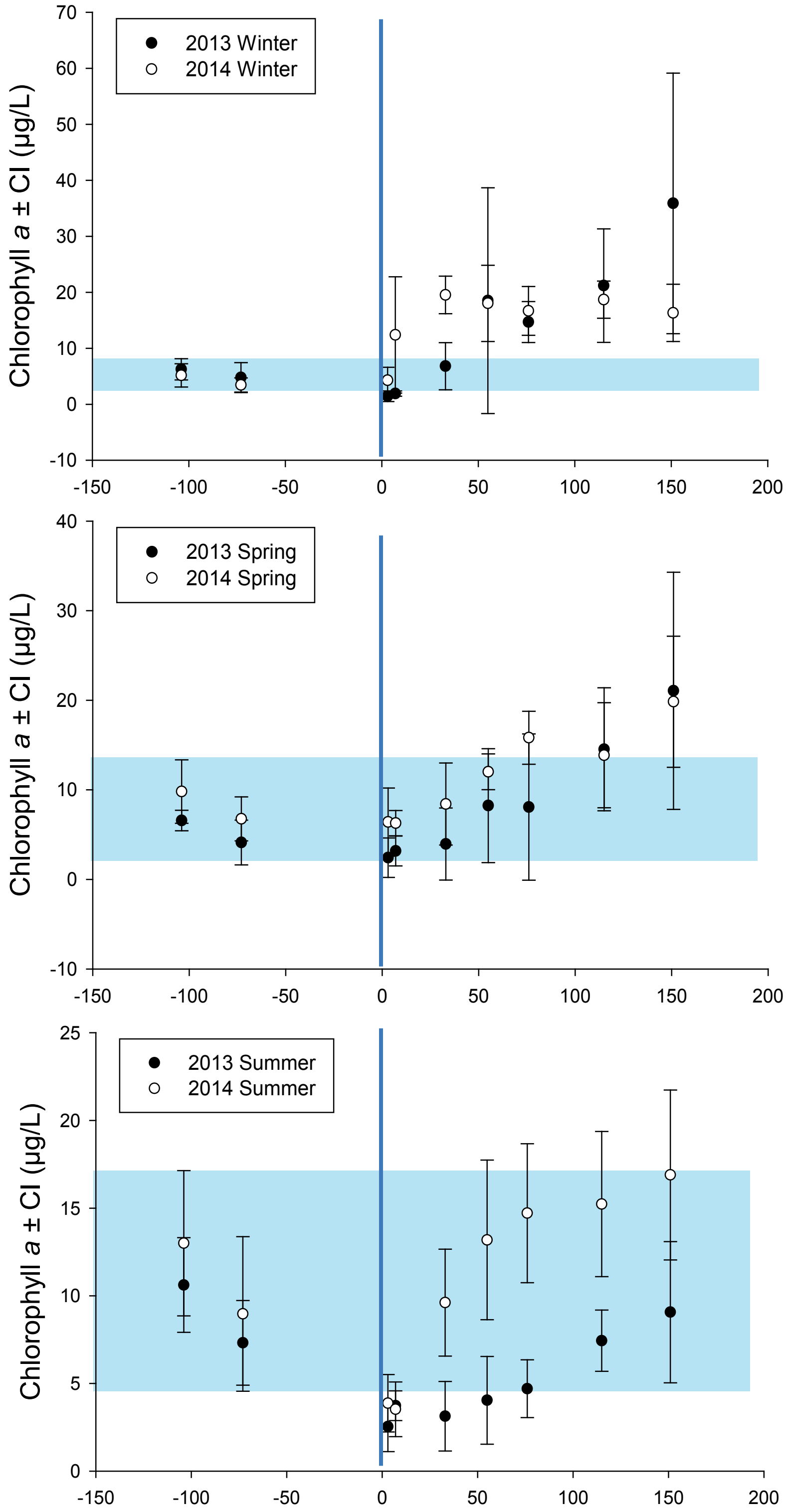


Figure 31: Longitudinal chlorophyll *a* values measured at nine sites (± 95% confidence interval) along the Macquarie River. The range of the 95% confidence interval of the two upstream sites is indicated by the blue shaded region. Distance (km) from the dam indicated on the x-axis. Seasonal mean was calculated for winter (June, July, August), spring (September, October, November) and summer (December, January February). The blue line indicates the location of the dam.

5.4.3 Effect of hypolimnial and epilimnial releases on the phytoplankton community structure at the outlet

Cyanobacteria accounted for very little of the phytoplankton community in the winter months when the dam was mixed (July 2013, May 2014 and July 2014). With the onset of stratification within the dam in September, the percentage of cyanobacteria released through the outlet increased in 2013 with low-level releases, and to a greater extent in 2014, when the thermal curtain was active (Fig. 32). Overall the proportion of cyanobacteria appears to have increased after the onset of thermal stratification, both prior to and after thermal curtain operation.

Since the change from low-level releases to surface water releases with the use of the thermal curtain, cell counts have increased drastically. The number of cyanobacterial cells detected in water samples from the outlet during the mixing period of the dam in July of both 2013 and 2014 were minimal, with other alga including diatoms and green algae forming the greater part of the community. In September of 2013, the algal community in the hypolimnial releases contained only just over 10% of cyanobacteria. In the following year with surface releases, this shifted to 52% of the community structure. Cyanobacteria dominated in January 2013, and September, October, and April 2014. The largest cell count observed in the year prior to the thermal curtain use was in January 2014, with 2153 cells/mL. Cyanobacterial cells accounted for 1844 cells/mL. In April the cell count sharply increased to 11460 cells/mL, with 10555 cells/mL being cyanobacteria (*Anabaena*).

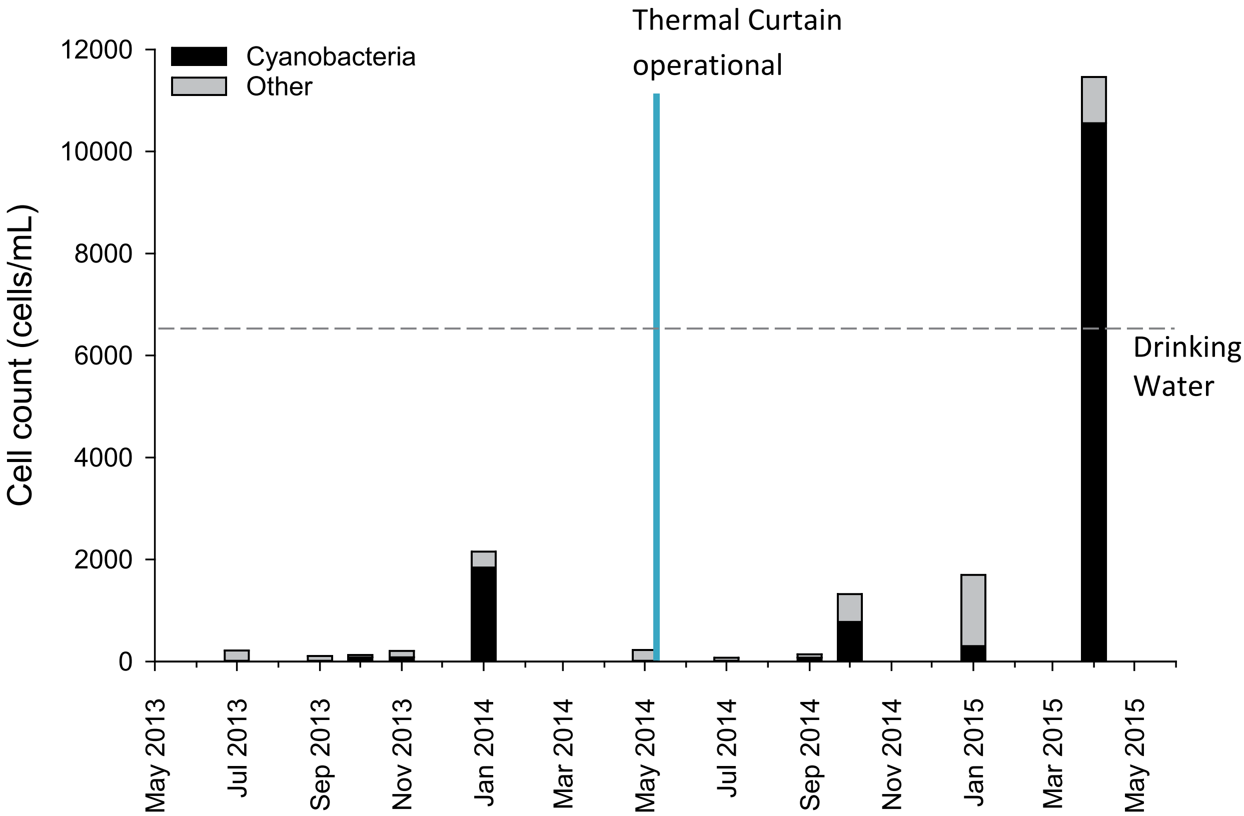


Figure 32: Concentration and proportion of cyanobacteria (dark) to "other" groups (light) in phytoplankton community. The vertical blue line indicates the point in time from when the thermal curtain was operation. The dashed horizontal line indicates the cell count for toxic cyanobacterial species considered unsafe for human consumption.

5.5 Discussion

5.5.1 Chlorophyll a *content at the outlet: Hypolimnial and epilimnial releases*

Algal concentration in an undisturbed river typically increases with distance downstream and is dependent upon flow and channel characteristics (Skidmore et al. 1998). Burrendong Dam forms a physical barrier on the Macquarie River which disrupts this progression, a phenomena described in the serial discontinuity concept (Stanford and Ward 2001). Considering this natural longitudinal pattern of phytoplankton concentration, a reference site upstream was used to assess the impact of the dam on the downstream Macquarie River. The chlorophyll *a* content observed at the outlet should ideally be of a similar value. This was not the case in both summer and winter and during epilimnial and hypolimnial releases (Figs. 29 and 30). Though the chlorophyll *a* values at the outlet did not resemble those of Long Point, there was improvement in the downstream chlorophyll *a* concentration with surface release in both seasons (Figs. 29a and 30a).

For chlorophyll *a* observed at the outlet to match upstream reference sites, the values observed at the reference site must be available within the dam water column. Depth profiles of chlorophyll *a* at intermittent intervals in the dam indicated the chlorophyll *a* values measured within the dam were too low to match those measured at Long Point (Figs. 29 and 30). Therefore it was not possible for the chlorophyll *a* concentrations at the outlet to match those of the upstream site. The concentrations measured at the outlet did correspond with the concentrations at the respective depth of withdrawal (Figs. 29 and 30)

In the cooler months, Burrendong Dam is usually in a mixed state with no thermal stratification. Therefore, chlorophyll *a* values detected at the downstream outlet are not expected to be affected by the depth of withdrawal within the dam (i.e. low-level in 2013, or surface in 2014). However, the chlorophyll *a* content of the water releases at the outlet was slightly higher in the winter month of July with surface releases (2014). The difference in phytoplankton concentration may be the result of partial or diel stratification on the day of sampling. This could have created an environment near the water surface for phytoplankton to migrate to within the photic zone (c.f. Mitrovic et al. 2001). Algal density varied minimally with depth (Fig. 29b) with a relatively small range of just less than 3.5 µg/L with depth, as expected in a mixed reservoir.

Chlorophyll *a* distribution with depth was directly affected by thermal stratification in the dam which occurred from October to early May in both years of sampling. The photic zone is located within the epilimnion, which is several degrees warmer than the hypolimnion, and this high light zone creates a good environment for phytoplankton growth and proliferation (Paerl and Huisman 2008, Talling et al. 2009, Zhou et al. 2014). The hypolimnion is physically isolated from the epilimnion and atmosphere and as a result there is usually low phytoplankton concentration present within this region. This is consistent with the chlorophyll *a* values measured in the dam depth profile during stratification. Chlorophyll *a* values measured at the outlet in July and December of 2013 and 2014 appear to coincide with the values observed at the depth of withdrawal. In December of 2013 with low-level releases, the off-take was located 18 m below the water surface. Chlorophyll *a* values at 10-20 m from the surface encompassed the value observed at the outlet. The chlorophyll *a* content of the water at the outlet was much lower in December 2013 than 2014 (Fig. 30a). In 2014, by channelling the epilimnial water for release through the thermal curtain, which was sitting at a depth of approximately 7 m from the water surface, the algal concentration measured at the outlet had increased from 2013. This is mirrored by the chlorophyll *a* values observed at the depth of the thermal curtain in the vertical profile.

5.5.2 Effect of hypolimnial and epilimnial releases on seasonal longitudinal chlorophyll a *concentrations*

Chlorophyll *a* longitudinal patterns below the dam were assessed on a seasonal basis. In all three seasons of winter, spring and summer, the chlorophyll *a* concentration immediately downstream of the dam wall were within the 95% confidence interval range of upstream sites. This indicates minimal impact of the dam on the phytoplankton density downstream of the dam, and of the thermal curtain with the surface releases, though slight increases in the seasonal mean of summer were seen.

In the summer of 2013, low chlorophyll *a* content in water sourced from the hypolimnion is likely to have played a role in the lower mean concentration in downstream waters. By changing the depth of withdrawal using the curtain to within the epilimnion, the water releases contained slightly higher concentrations of pelagic phytoplankton which may have persisted downstream. There is very little in the literature to inform on the effects of selective withdrawal in a reservoir on the longitudinal chlorophyll *a* patterns or community structure of a river.

5.5.3 Effect of hypolimnial and epilimnial releases on the phytoplankton community at the outlet

Seasonal succession

The seasonal succession of pelagic algae in lakes typically follows a pattern of domination of small centric diatoms in spring, green algae and large diatoms in early summer, followed by dinoflagellates then cyanobacteria in mid-late summer, and the take-over of diatoms again in autumn (Sommer et al. 1986). Diatoms cannot regulate their location within the water column and are prone to sinking in stagnant environments, making the winter period of mixing the optimal environment for their persistence (Pinckney et al. 1998, Ptacnik et al. 2003). Green algae take over in early summer before cyanobacteria set in when nutrients are depleted in the epilimnion (Lopes et al. 2007). Some cyanobacterial species can control buoyancy with gas vesicles and mucilage and do well under a stable water column such as under stratified conditions (Mitrovic et al. 2003). Further, some also have nitrogen-fixing capabilities (heterocysts). These features can allow cyanobacteria to dominate the phytoplankton community in summer (Huisman et al. 2005) though this does not always occur.

When considering the cyanobacterial domination of the phytoplankton community, the findings in this study support the typical seasonal succession of phytoplankton reported in the literature (Kohler 1994, Wehr and Descy 1998) (Fig. 32). With minimal cyanobacteria detected immediately downstream of Burrendong outlet in the autumn and winter months, the increase in chlorophyll *a* is mainly attributed to other algae groups including diatoms and green algae. This could be seen as beneficial to downstream river stretches during this period as diatom groups are a naturally occurring plankton group of lotic systems, supporting the riverine food web. Similarly, Hartman and Himes (1961) found the proportion of diatoms in the phytoplankton community structure downstream of the Pymatuning Reservoir to increase with distance from the dam wall within the 16.9 km distance of sampling. Planktonic diatoms commonly occur in lotic environments due to their fast growth rates overcoming poor light environments (Wehr and Descy 1998) and have been measured to contribute up to 95% of phytoplankton biomass (Montesanto et al. 2000). They also typically dominate the algal community of reservoirs in winter, and when released downstream, have been found to persist and increase their biomass with distance from the outlet (Kohler 1994). Alternatively, common summer species (including cyanobacteria) can decrease substantially over the course of the downstream river (Kohler 1994) though this is not always the case as witnessed by the Macquarie River bloom of the 1990's. The percentage of the phytoplankton community occupied by cyanobacteria recorded at Burrendong outlet was low (< 10%) in winter of both 2013 and 2014, and increased in summer (Fig. 32).

Cyanobacteria

While increased chlorophyll *a* may be considered a beneficial outcome of surface releases in respect to planktonic food resources, it is essential to also consider any possible adverse consequences of cyanobacteria in release waters. Cyanobacterial blooms are potentially damaging to the ecological health of the Macquarie River, as well as to the downstream communities that rely on the river for drinking water supply, agricultural irrigation and recreation. Cyanotoxins are released from both actively growing and lysed cells and are produced in up to 75% of cyanobacterial blooms (Singh and Dhar 2013). The toxins may be neurotoxic, hepatotoxic, and dermatotoxic and can harm organisms including zooplankton, fish, birds, and mammals (Singh and Dhar 2013). There are risks for human use as drinking water, for example the town of Wellington uses the Macquarie River as its drinking water source. As a result, the impact of dam releases on downstream cyanobacterial concentrations is a principal management concern. The World Health Organisation (WHO) recommends a safe water guideline of 1 μg/L of the hepatotoxin microcytsin (Chorus and Bartram 1999) and this has been translated into a guideline of 6,500 cells/mL of the cyanobacteria *Microcystis aeruginosa* (NHMRC 2011).

Reservoir releases introduce lentic phytoplankton species into a lotic environment and potentially toxic cyanobacteria are common in the epilimnion of a thermally stratified reservoir. As stratified dams can create an environment suitable for the proliferation of cyanobacteria leading to blooms (Jones and Poplawski 1998, Mitrovic et al. 2011, Bittencourt-Oliveira et al. 2012) epilimnial releases may source and seed cyanobacteria downstream of the dam. This could be a contributing factor to the increased phytoplankton chlorophyll *a* biomass that was observed below the dam in the post-curtain stratification period of 2014 and epilimnial releases. Residence time was high in the river as a result of low discharge volume, meaning the downstream Macquarie River may have been more favourable environment for lentic phytoplankton species including cyanobacteria. Cell counts confirmed the community structure to be dominated by the potentially toxic cyanobacterial species *Anabaena*. In April the cell count increased to 11460 cells/mL, with 10555 cells/mL being cyanobacteria (*Anabaena*). This is above the limit for the safe consumption of drinking water set by NHMRC (NHMRC 2011).

The probability of cyanobacterial blooms being released downstream of the dam during periods of thermal stratification within the reservoir are substantially higher with surface releases compared to hypolimnial releases. The percentage of the phytoplankton community in Burrendong Dam occupied by cyanobacteria increased with surface releases during the summer months. Though cell counts had increased in January 2014 with low-level releases, the cell count had increased to levels that surpass the safe drinking water guidelines in April of 2015 with surface releases.

5.6 Management options

There are a few options for preventing cyanobacterial blooms within the reservoir being transported downstream. One of these could be dropping the depth of the thermal curtain to below the thermocline, or potentially the installation of an algal curtain. Lowering the thermal curtain to a greater depth from the water surface, such as near or below the thermocline, could prevent or reduce the toxic algal cells from being released downstream. However, this would again result in cold water pollution. Depending on the speed by which the curtain is lowered, cold shock may become a risk. Cold shock in aquatic organisms occurs as a result of sudden temperature change (Donaldson et al. 2008), usually caused by the sudden change in release depth from surface to low-level (Ryan and Preece 2003). An array of symptoms can occur, ranging from impeded foraging success to the organism entering into a coma-like state, rendering them at risk of predation or physical damage from striking the substrate from in-stream turbulence (Donaldson et al. 2008, Lyon et al. 2008). This could be avoided with slower lowering of the thermal curtain. The appropriate depth of the curtain may be dictated by the distribution of cyanobacteria through depth, as during stratified conditions they may not be homogenous within the epilimnion (Mitrovic et al. 2001, Medrano et al. 2013). If most cyanobacteria are within the top 2 to 4 m then a curtain at 7 or 8 m may miss the bulk of them. Sinking dead cyanobacterial cells could also be an issue as they may release more cyanotoxins into the water column than live cells (Berg et al. 1987, Watanabe et al. 1992). However, the concentration of toxins present at varying depths within a reservoir is currently not tested.

A secondary option may be an algal curtain submerged from the water surface to an appropriate depth, for example, 3 - 5 m could physical obstruct floating algal cells from accessing the outlet of the thermal curtain. This would depend on the depth distribution of algae and how this changes on a daily time scale and with environmental factors such as wind. Also the physical ability of a floating curtain to withstand the draw of the water through the curtain opening will need to be assessed. Research should be done to identify if this is feasible from the physical perspective, and if so, whether the distribution of cyanobacteria with depth would be practical for the use of the algal curtain and to determine the best depth to set it at, depending on meteorological conditions and diel cycles. If most cyanobacteria are found to occur in the surface few metres of water, it may be possible that a large percentage of the bloom would be impeded from being released downstream, decreasing the risk from releases and the likelihood of a bloom establishing in the Macquarie River.

5.7 Conclusions

Epilimnial releases resulted in increased chlorophyll *a* concentrations from hypolimnial releases, measured at the outlet. The epilimnial releases also appeared to improve the longitudinal recovery of chlorophyll *a* in summer compared to hypolimnial releases. With the change from hypolimnial to epilimnial releases, cell counts of cyanobacteria, and the proportion of cyanobacteria to other alga genera increased substantially. These findings support our hypotheses 3a and 3b.

Chapter 6: Conclusions & Recommendations

6.1 Summary of findings

This research thesis examined the occurrence and cause of historical CWP in the Macquarie River below Burrendong Dam and the effectiveness of an innovative thermal control structure, the thermal curtain, in mitigating CWP. This was achieved through analysis of historical temperature data, and through comparisons of thermal patterns in 2013 and 2014 summer periods. This included comparing temperatures above and below the dam as well as the longitudinal distance of thermal disruption downstream of the dam. The research also examined the impacts of the dam on nutrient concentrations and algal concentration (as chlorophyll *a*) and cyanobacteria concentration before and after thermal curtain operation.

This research is highly important for assessing the effectiveness of the thermal curtain in mitigating CWP. It is also important as a means to gather information and to advise for the effective management of the structure. Prior studies have suggested releases from Burrendong Dam had resulted in degraded downstream water quality, likely to be affecting downstream ecological health (Astles et al. 2003, Lugg and Copeland 2014). Improvement in water quality downstream of the dam is desirable for this purpose, that is, the improvement of downstream ecological health, particularly with the important role of environmental flows delivered from the dam. Water of warmer temperatures would be beneficial for both environmental and anthropogenic uses. The thermal curtain has been developed as an affordable alternative to the more expensive options of multiple level off-takes for the mitigation of CWP and improvement of downstream water quality.

6.1.1 Historic CWP below Burrendong Dam

CWP has been a long term problem occurring in the Macquarie River below Burrendong Dam. Findings from the analysis of historical data indicated the CWP caused thermal suppression of around 13.5°C below the dam. CWP was a result of hypolimnial releases during periods of thermal stratification, with the months of December and January exhibiting the greatest thermal depression. Three factors were identified as major contributors to historical CWP: thermal stratification of the water body, large residence level of the dam and large discharge volumes.

6.1.2 Effect of the thermal curtain: Temperature

Operation of the thermal curtain resulted in the improvement of the thermal regime downstream of the dam. By sourcing water from the epilimnion of the stratified water body for release instead of from the hypolimnion, the downstream water temperatures were more similar to the natural thermal regime of the Macquarie River. The thermal regime improved by up to 3°C in daily mean, and 2.5°C in monthly mean. These improvements are modest when considering the low residence level of the reservoir for the duration of the study, where it rarely exceeded 20% capacity. With a lower residence level the thermocline is located closer to the off-take, meaning warmer temperatures are likely to be released downstream than with greater residence level when the thermocline sits at a higher altitude from the outlet. Though substantial improvements were identified in the thermal regime as a result of the thermal curtain use with low residence level, it is likely that more profound improvement will occur with a deeper reservoir, when the capacity is closer to 100%.

The mean daily temperatures in the year before operation of the thermal curtain were compared to the thermal requirements for the spawning of native fish. This indicated spawning temperatures were not met for 3 common species, golden perch, Murray cod and Silver Perch and that temperature is a factor in the reduction of native species downstream of Burrendong Dam. With improved temperatures using the curtain, these minimum spawning temperatures were met for all there fish for at some periods directly below the dam.

Diel variation is also an important component of a rivers thermal regime. Cold water releases reduced the diel variation. With warm surface releases, diel variation improved markedly, where the maximum value of the diel range rose by a factor of 1.5. The study also found that thermal stratification occurs within the thermal curtain itself. When discharge is less than 800 ML/day, thermal stratification within the curtain reduces the effectiveness of the thermal curtain in mitigating cold water pollution. Flows above this threshold lead to the break-down of thermal stratification. It was during these periods when the thermal curtain was found to be most effective.

6.1.3 *Effect of the thermal curtain: Nutrients*

The nutrient concentrations within the reservoir throughout the duration of the study were frequently greater than ANZECC trigger values. Thermal stratification led to the development of an oxycline and DO levels below 2 mg/L in the hypolimnion, and an increase in the concentration of most analytes in the hypolimnion. Some of these nutrients (ammonia, NO_x and TFN) were increased in discharge waters from the hypolimnion from upstream values. The concentrations were reduced slightly with the operation of the thermal curtain. However, the strength of stratification and duration of anoxic conditions in the hypolimnion may have increased. This may be amplified with greater residence level and lead to a more profound impact on nutrient concentrations.

6.1.4 Effect of the thermal curtain: Phytoplankton

By changing the release depth from below the thermocline to above it and within the epilimnion, there was an increase in downstream phytoplankton concentration, measured as chlorophyll *a* (μg/L). The longitudinal recovery of phytoplankton density to upstream values was improved in summer. While it is useful to have phytoplankton concentrations that are similar to upstream of the dam from surface releases, there is the risk of toxic cyanobacteria. Late in the summer of 2015 cell counts indicated that the majority of the phytoplankton composition below the dam consisted of potentially toxic strains of cyanobacteria, and cell counts increased with surface releases. Surface releases may be detrimental to downstream riverine health and investigation of management control options are necessary to reduce the risk of toxic blooms developing in the Macquarie River seeded by the dam, which could endanger the health of livestock, wildlife and humans.

6.2 Management recommendations

6.2.1 Thermal curtain depth: is 7 m effective?

The position of the thermal curtain should be located above the thermocline. This will allow the warm surface waters to be targeted, lowering the possibility for cooler, hypolimnetic water to be drawn. This project identified that the optimum temperature range to closely match upstream reference site thermal regime was available in the epilimnion. If water was sourced from the hypolimnion, these temperatures would be reduced. This would result in the downstream thermal regime being colder than required, and the thermal curtain not being used to its fullest potential.

For the duration of this study, 7 m was an appropriate height for the functionality of the thermal curtain. However, the study did not extend to include the end of the stratification period of the first year of thermal curtain operation. Surface releases strengthen stratification and raise the thermocline with respect to depth in the reservoir. The depth of the thermal curtain for current use was decided according to the thermal stratification pattern within Burrendong Dam according to data collected when low-level release valve was operational. Therefore, it is suggested that the thermal profile be monitored to ensure the epilimnion is being targeted most effectively. Furthermore, the stratification patterns may also change with increased volume of the reservoir. For this study, the dam remained at around 20% of full capacity.

6.2.2 Problematic cyanobacteria

A secondary curtain could be submerged from the water surface to an appropriate depth surrounding the current thermal curtain, for example, 3-5 m. This could physically obstruct algal cells from accessing the top of the thermal curtain. Some cyanobacteria may still be below this depth depending on mixing conditions, and dead cyanobacterial cells may still be directed to the outlet through the thermal curtain. Such a system may reduce a large percentage of cyanobacterial blooms from within the reservoir from being released downstream. Thus the likelihood of high concentrations or blooms being seeded in the downstream stretch of the Macquarie River would be reduced.

Alternatively, the thermal curtain could be dropped to below the thermocline to prevent the release of harmful cyanobacteria downstream as is currently suggested practice. In this case, the potential for cold shock would need to be investigated. If identified to be a problem, reducing the lowering speed of the curtain is suggested.

6.2.3 Nutrients

Nutrient and metals analysis completed in this study indicate that the dynamics within the reservoir itself has impacted on their concentrations, with levels exceeding ANZECC guidelines for the protection of ecosystems. The nutrient concentrations observed at the outlet were reduced with the use of the thermal curtain, though still greater than the ANZECC trigger values. The eutrophication of the Macquarie River is likely to be as a result of external nutrient loading, with the land use of the catchment largely agricultural. It is recommended nutrients are monitored closely with the monitoring of cyanobacteria populations as the concentrations may exacerbate or support blooms.

6.2.4 Future use of the thermal curtain

Overall, the thermal regime improved with the use of the thermal curtain despite relatively low dam levels. It is anticipated the downstream ecological health and water quality will benefit from this. The data from this study indicates that the thermal curtain appears to be a successful and more affordable alternative to other current options for CWP mitigation, such as MLO's. Therefore, its future use in other dams should be considered. It is recommended to continue thermal monitoring at Burrendong Dam, and to test the effectiveness of the curtain during a period of higher dam levels such as above 50% capacity.

Some factors that directly affect the success of the thermal curtain structure in mitigating CWP should be taken into account when contemplating installation at other dams. For example, the discharge thresholds as a means of weakening or preventing thermal stratification within the thermal curtain itself. The threshold for Burrendong Dam was identified to be discharge volumes of greater than 800ML/day. At Burrendong, this should not be an issue due to high water demands during the stratification period. However, the thermal curtain in a dam with smaller release volumes may not be as successful in improving the downstream thermal regime as these volumes may not be sufficient to break down stratification within the curtain. Therefore dam purpose and subsequent release volumes should be taken into consideration when considering the installation of a thermal curtain.

6.3 Research recommendations

6.3.1 Reservoir residence level

1- Ongoing monitoring under a deeper reservoir

The results of this study assess the effects of the dam and thermal curtain on temperature, nutrients and algae when the reservoir rarely exceeded 20% capacity for the duration of the study and the thermocline was closer to the off-take. It is highly recommended that future research continues to assess these factors under a range of operating conditions including when the dam is closer to full capacity. It is anticipated that increased storage volume will affect the development and depth of the thermocline and oxycline during thermal stratification. This warrants further investigation of the effectiveness of the thermal curtain in mitigating CWP and the effects on algae and nutrients within and below the dam.

2- Stratification within the thermal curtain

Residence level may affect the strength of stratification within the thermal curtain, effectively altering the discharge threshold. An increase in the residence level can affected the development and depth of a thermocline within a body of water. This should be taken into consideration in future. This may also influence the discharge threshold required to breakdown the stratification. Further study is required to understand the full extent of the impacts of residence level on the development of thermal stratification within the thermal curtain, and how different capacities of a dam may affect this and subsequently the effectiveness of the thermal curtain in mitigating CWP.

6.3.2 Phytoplankton

3- Continued monitoring of chlorophyll *a* and algal community structure for long term analysis and to account for annual variation in results

Ongoing monitoring of phytoplankton dynamics in terms of chlorophyll *a* and community structure is recommended. The findings of this study were based on information gathered over a period of one year prior to, and one year after the operation of the thermal curtain. While trends and patterns were identified, this data pool does not account for the possibility of annual variation. Further monitoring to gather more data is recommended to further support and strengthen the findings in this study and to encompass different algal community dynamics including cyanobacterial dominance and blooms.

4- Management options for prevention of releasing cyanobacterial blooms downstream:

    a. Development of an algal curtain

This study has found that the release of surface waters can increase the cell count of cyanobacteria occurring downstream. This is a primary management concern as the occurrence of potentially toxic blooms downstream of the dam would have major impact on downstream towns and drinking water supply, recreation, agricultural stock and wildlife. It has been recommended in this study that an algal curtain be considered to form a physical barrier to buoyant algal cells, preventing their release downstream through the thermal curtain. Therefore, future research is required to determine if cyanobacterial vertical distributions are suitable for a curtain, and if so, the most effective depth for it to go down to.

b. Lowering the thermal curtain: determine the most appropriate and effective depth

Lowering the thermal curtain is an alternative management option in order to prevent the release of potentially harmful cyanobacterial blooms to downstream stretches of the Macquarie River. An appropriate depth should be identified for the most effective management. It is likely that the curtain will be required to be lowered to a point near or below the thermocline. A depth to balance cyanobacterial discharge downstream and cold water pollution should be found. Further, the speed by which the curtain should be dropped needs to be determined. If the curtain is lowered too quickly, there is a risk of causing cold shock in downstream biota. Cold shock occurs as a result of sudden temperature change and can manifest through an array of symptoms in fish, ranging from discoordination to fatality.

6.3.3 Biological measurements for recovery

There is very little in the literature on the effect of the thermal regime on biota. The most common information available is on fish, and generally focuses on the effects of thermal suppression and elevation on younglings in laboratory environments. Further research should include the effects of the thermal curtain and dam operation on a range of biota including zooplankton, macroinvertebrates, mussels and fish. The findings of this study are encouraging, whereby the improvements in the thermal regime downstream of Burrendong Dam as a result of the use of the thermal curtain are likely to support the breeding of native fish. To include the effects of CWP mitigation on aquatic organisms in future studies would be an important for understanding how the thermal improvements may have improved river health.

References

Acaba, Z., H. Jones, R. Preece, S. Rish, D. Ross, and H. Daly. 2000. The Effects of Large Reservoirs on Water Temperature in Three NSW Rivers Based on the Analysis of Historical Data. Centre for Natural Resources, NSW Department of Land and Water Conservation: Sydney.

Amirbahman, A., A. Pearce, R. Bouchard, S. Norton, and J. Steven Kahl. 2003. Relationship between hypolimnetic phosphorus and iron release from eleven lakes in Maine, USA. Biogeochemistry **65**:369-386.

AMOG Consulting. 2010. Burrendong Dam Conceptual Design Of Floating Curtain System: stage 1 curtain system investigation and development. Australia.

ANCOLD. 2012. Australian National Committee on Large Dams Register of large dams in Australia.

ANZECC, and ARMCANZ. 2000. Australian and New Zealand guidelines for fresh and marine water quality. Australian and New Zealand Environment and Conservation Council & Agriculture and Resource Management Council of Australia and New Zealand, Canberra.

APHA. 2005. Standard methods for the examination of water and wastewater. 21 edition. American Public Health Association, Washington, D.C.

Asaeda, T., H. S. Pham, D. G. N. Priyantha, J. Manatunge, and G. C. Hocking. 2001. Control of algal blooms in reservoirs with a curtain: a numerical analysis. Ecological Engineering **16**:395-404.

Asaeda, T., D. G. N. Priyantha, S. Saitoh, and K. Gotoh. 1996. A new technique for controlling algal blooms in the withdrawal zone of reservoirs using vertical curtains. Ecological Engineering **7**:95-104.

Astles, K. L., R. K. Winstanley, J. H. Harris, and P. C. Gehrke. 2003. Experimental study of the effects of cold water pollution on native fish. 44, NSW Fisheries Research Institute, Australia.

Australian National Committee on Large Dams. 1982. Register of Large Dams in Australia.

Baldwin, D. S., and J. Williams. 2007. Differential release of nitrogen and phosphorus from anoxic sediments. Chemistry and Ecology **23**:243-249.

Baldwin, D. S., J. Wilson, H. Gigney, and A. Boulding. 2010. Influence of extreme drawdown on water quality downstream of a large water storage reservoir. River Research and Applications **26**:194-206.

Bates, D. M., and J. M. Chambers. 1992. Nonlinear Models. Pages 421-454 *in* J. M. Chambers and T. J. Hastie, editors. Statistical Models in S. Wadsworth, Pacific Grove, CA.

Baxter, R. M. 1977. Environmental effects of dams and impoundments. Annual Review of Ecology and Systematics **8**:255-283.

Berg, K., O. M. Skulberg, and R. Skulberg. 1987. Effects of decaying toxic blue-green-algae on water-quality - A laboratory study. Archiv Fur Hydrobiologie **108**:549-563.

Beutel, M. W. 2001. Oxygen consumption and ammonia accumulation in the hypolimnion of Walker Lake, Nevada. Hydrobiologia **466**:107-117.

Beutel, M. W. 2006. Inhibition of ammonia release from anoxic profundal sediments in lakes using hypolimnetic oxygenation. Ecological Engineering **28**:271-279.

Bino, G., C. Steinfeld, and R. T. Kingsford. 2014. Maximizing colonial waterbirds' breeding events using identified ecological thresholds. and environmental flow management. Ecological Applications **24**:142-157.

Bittencourt-Oliveira, M. C., S. N. Dias, A. N. Moura, M. K. Cordeiro-Araujo, and E. W. Dantas. 2012. Seasonal dynamics of cyanobacteria in a eutrophic reservoir (Arcoverde) in a semi-arid region of Brazil. Brazilian Journal of Biology **72**:533-544.

Blumberg, A. F., and D. M. Ditoro. 1990. Effects of climate warming on dissolved-oxygen concentrations in Lake Erie. Transactions of the American Fisheries Society **119**:210-223.

Bormans, M., H. Maier, M. Burch, and P. Baker. 1997. Temperature stratification in the lower River Murray, Australia: implication for cyanobacterial bloom development. Marine and Freshwater Research **48**:647-654.

Bormans, M., and I. T. Webster. 1997. A mixing criterion for turbid rivers. Environmental modelling & software **12**:329-333.

Brown, G. W. 1969. Predicting temperatures of small streams. Water Resources Research **5**:68-&.

Burford, M. A., S. A. Green, A. J. Cook, S. A. Johnson, J. G. Kerr, and K. R. O'Brien. 2012. Sources and fate of nutrients in a subtropical reservoir. Aquatic Sciences **74**:179-190.

Burger, D. F., D. P. Hamilton, and C. A. Pilditch. 2008. Modelling the relative importance of internal and external nutrient loads on water column nutrient concentrations and phytoplankton biomass in a shallow polymictic lake. Ecological Modelling **211**:411-423.

Burton, C. 2000a. Assessment of the water temperature regime of the Cudgegong River, Central West, New South Wales. NSW Department of Land and Water Conservation.

Burton, C. 2000b. Assessment of the water temperature regime of the Macquarie River, Central West, New South Wales. NSW Department of Land and Water Conservation.

Byren, B. A., and B. R. Davies. 1989. The effect of stream regulation on the physico-chemical properties of the palmiet river, South Africa. Regulated Rivers: Research & Management **3**:107-121.

Caissie, D. 2006. The thermal regime of rivers: a review. Freshwater Biology **51**:1389-1406.

Caliskan, A., and S. Elci. 2009. Effects of Selective Withdrawal on Hydrodynamics of a Stratified Reservoir. Water Resources Management **23**:1257-1273.

Camargo, J. A., K. Alonso, and M. de la Puente. 2005. Eutrophication downstream from small reservoirs in mountain rivers of Central Spain. Water Research **39**:3376-3384.

Camargo, J. A., and D. G. de Jalon. 1990. The downstream impacts of the Burgomillodo reservoir, Spain. Regulated Rivers: Research & Management **5**:305-317.

Camargo, J. A., and D. G. De Jalon. 1995. Assessing the influence of altitude and temperature on biological monitoring of freshwater quality: A preliminary investigation. Environmental monitoring and assessment **35**:227-238.

Carpenter, S. R., J. F. Kitchell, and J. R. Hodgson. 1985. Cascading Trophic Interactions and Lake Productivity. Bioscience **35**:634-639.

Chorus, I., and J. Bartram. 1999. Toxic cyanobacteria in water: A guide to public health significance, monitoring and management. Spon/Chapman & Hall, London.

Clarkson, R. W., and M. R. Childs. 2000. Temperature effects of hypolimnial-release dams on early lire stages of Colorado River Basin big-river fishes. Copeia:402-412.

Coles, J. F., and R. C. Jones. 2000. Effect of temperature on photosynthesis-light response and growth of four phytoplankton species isolated from a tidal freshwater river. Journal of Phycology **36**:7-16.

Conley, D., P. Stalnacke, H. Pitkanen, and A. Wilander. 2000. The transport and retention of dissolved silicate by rivers in Sweden and Finland. Limnology and Oceanography **45**:1850-1853.

Conover, W. 1999. Practical nonparametric statistics. 3rd edition. Wiley, 592 p, New York.

Coutant, C. 1963. Stream plankton above and below Green Lane Reservoir. Pages 22-126 *in* Proceedings of the Pennsylvania Academy of Science.

Cowx, I., W. Young, and J. Booth. 1987. Thermal characteristics of two regulated rivers in mid-wales, UK. Regulated Rivers: Research & Management **1**:85-91.

Dent, S. R., M. W. Beutel, P. Gantzer, and B. C. Moore. 2014. Response of methylmercury, total mercury, iron and manganese to oxygenation of an anoxic hypolimnion in North Twin Lake, Washington. Lake and Reservoir Management **30**:119-130.

Department of Public Works and Services. 1996. Modification of Outlet Works at DLWC Dams: Value Management Study Report. Department of Public Works and Services, Sydney.
Diana, J. S. 1984. The growth of largemouth bass, Micropterus-salmoides (Lacepede), under constant and fluctuating temperatures. Journal of Fish Biology **24**:165-172.
Donaldson, M. R., S. J. Cooke, D. A. Patterson, and J. S. Macdonald. 2008. Cold shock and fish. Journal of Fish Biology **73**:1491-1530.
Driver, P., J. Harris, R. Norris, and G. Closs. 1997. The role of the natural environment and human impacts in determining biomass densities of common carp in New South Wales rivers. Fish and Rivers in Stress. The NSW Rivers Survey'.(Eds PC Gehrke and JH Harris) pp:225-250.
Edinger, J. E., D. W. Duttweiler, and J. C. Geyer. 1968. The response of water temperatures to meteorological conditions. Water Resources Research **4**:1137-1143.
Egge, J. K., and D. L. Aksnes. 1992. Silicate as regulating nutrient in phytoplankton competition. Marine Ecology Progress Series **83**:281-289.
Elliott, J. M. 1978. Effect of temperature on the hatching time of eggs of Ephemerella ignita (Poda) (Ephemeroptera:Ephemerellidae). Freshwater Biology **8**:51-58.
EPA Victoria. 2004. Cold water discharges from impoundments and impacts on aquatic biota. EPA Victoria.
Evans, E. C., G. R. McGregor, and G. E. Petts. 1998. River energy budgets with special reference to river bed processes. Hydrological Processes **12**:575-595.
Fernandez, R. L., M. Bonansea, A. Cosavella, F. Monarde, M. Ferreyra, and J. Bresciano. 2012. Effects of bubbling operations on a thermally stratified reservoir: implications for water quality amelioration. Water Science and Technology **66**:2722-2730.
Fontane, D. G., J. W. Labadie, and B. Loftis. 1981. Optimal control of reservoir discharge quality through selective withdrawal. Water Resources Research **17**:1594-1604.
Fox, J., and S. Weisberg. 2010. Nonlinear Regression and Nonlinear Least Squares in R. An R Companion to Applied Regression. Sage Publications, Los Angeles.
Fraisse, S., M. Bormans, and Y. Lagadeuc. 2013. Morphofunctional traits reflect differences in phytoplankton community between rivers of contrasting flow regime. Aquatic Ecology **47**:315-327.
Friesen, M. K., J. F. Flannagan, and S. G. Lawrence. 1979. Effects of temperature and cold-storage on development time and viability of eggs of the burrowing mayfly Hexagenia-rigida (Ephemeroptera, Ephemeridae). Canadian Entomologist **111**:665-673.
Garnier, J., G. Billen, and M. Coste. 1995. Seasonal succession of diatoms and chlorophyceae in the drainage network of the Seine River - Observations and modeling. Limnology and Oceanography **40**:750-765.
Gassama, N., C. Cocirta, and H. U. Kasper. 2012. Use of major and selected trace elements to describe mixing processes in a water reservoir. Comptes Rendus Geoscience **344**:25-32.
Gleick, P. H. 2003. Global freshwater resources: soft-path solutions for the 21st century. Science (New York, N.Y.) **302**:1524-1528.
Gleick, P. H., and M. Palaniappan. 2010. Peak water limits to freshwater withdrawal and use. Proceedings of the National Academy of Sciences **107**:11155-11162.
Gordon, N. D., T. A. McMahon, B. L. Finlayson, C. J. Gippel, and R. J. Nathan. 2004. Stream hydrology: an introduction for ecologists. John Wiley & Sons.
Green, D., J. Petrovic, P. Moss, and M. Burrell. 2011. Water resources and management overview: Macquarie-Bogan catchment. NSW Office of Water, Sydney.
Gu, R. C., S. Montgomery, and T. Al Austin. 1998. Quantifying the effects of stream discharge on summer river temperature. Hydrological Sciences Journal-Journal Des Sciences Hydrologiques **43**:885-904.

Hanjra, M. A., T. Ferede, and D. G. Gutta. 2009. Reducing poverty in sub-Saharan Africa through investments in water and other priorities. Agricultural Water Management **96**:1062-1070.

Hanjra, M. A., and M. E. Qureshi. 2010. Global water crisis and future food security in an era of climate change. Food Policy **35**:365-377.

Harleman, D. R. F. 1982. Hydrothermal Analysis of Lakes and Reservoirs. Pages 27-62 *in* J. A. Gore and G. E. Petts, editors. Alternatives in Regulated River Management. CRC Press, Florida.

Harris, J. H. 1997. Environmental Rehabilitation and Carp Control. Pages 22-24 *in* J. ROBERTS and R. TIZLEY, editors. Controlling Carp: Exploring the Options for Australia, Albury. CSIRO Land and Water.

Hartman, R. T., and C. L. Himes. 1961. Phytoplankton from Pymatuning Reservoir in downstream areas of the Shenanga River. Ecology **42**:180-&.

Hochman, Z., P. S. Carberry, M. J. Robertson, D. S. Gaydon, L. W. Bell, and P. C. McIntosh. 2013. Prospects for ecological intensification of Australian agriculture. European Journal of Agronomy **44**:109-123.

Hotzel, G., and R. Croome. 1994. Long-term phytoplankton monitoring of the Darling River at Burtundy, New South Wales - Incidence and significance of cyanobacterial blooms. Australian Journal of Marine and Freshwater Research **45**:747-759.

Huisman, J., H. C. P. Matthijs, and P. M. Visser. 2005. Harmful cyanobacteria. Springer, Dordrecht, The Netherlands.

Humborg, C., D. J. Conley, L. Rahm, F. Wulff, A. Cociasu, and V. Ittekkot. 2000. Silicon retention in river basins: Far-reaching effects on biogeochemistry and aquatic food webs in coastal marine environments. Ambio **29**:45-50.

Hutchinson, G. E. 1953. The Concept of Pattern in Ecology. Proceedings of the Academy of Natural Sciences of Philadelphia **105**:1-12.

Ingleton, T., T. Kobayashi, B. Sanderson, R. Patra, C. M. O. Macinnis-Ng, B. Hindmarsh, and L. C. Bowling. 2008. Investigations of the temporal variation of cyanobacterial and other phytoplanktonic cells at the offtake of a large reservoir, and their survival following passage through it. Hydrobiologia **603**:221-240.

Jones, G. J., and W. Poplawski. 1998. Understanding and management of cyanobacterial blooms in sub-tropical reservoirs of Queensland, Australia. Water Science and Technology **37**:161-168.

Jorgensen, S. E., H. Loffler, W. Rast, and M. Straskraba. 2005. Lake and reservoir management. Elsevier Science.

Jowsey, E. 2012. The changing status of water as a natural resource. International Journal of Sustainable Development and World Ecology **19**:433-441.

Jung, N.-c. 2009. Eco-hydraulic modelling of eutrophication for reservoir management. Delft University of Technology, The Netherlands.

Kantoush, S. A., J. L. Boillat, and A. J. Schleiss. 2008. Evolution of sediment deposition and flow patterns in a rectangular shallow reservoir under suspended sediment load.

Kingsford, R. T., and K. M. Auld. 2005. Waterbird breeding and environmental flow management in the Macquarie Marshes, Arid Australia. River Research and Applications **21**:187-200.

Kobayashi, T., D. S. Ryder, T. J. Ralph, D. Mazumder, N. Saintilan, J. Iles, L. Knowles, R. Thomas, and S. Hunter. 2011. Longitudinal spatial variation in ecological conditions in an in-channel floodplain river system during flow pulses. River Research and Applications **27**:461-472.

Kohler, J. 1994. Origin and succession of phytoplankton in a river-lake system (Spree, Germany). Hydrobiologia **289**:73-83.

Kunz, M. J., F. S. Anselmetti, A. Wueest, B. Wehrli, A. Vollenweider, S. Thuering, and D. B. Senn. 2011a. Sediment accumulation and carbon, nitrogen, and phosphorus deposition in

the large tropical reservoir Lake Kariba (Zambia/Zimbabwe). Journal of Geophysical Research-Biogeosciences **116**.
Kunz, M. J., A. Wuest, B. Wehrli, J. Landert, and D. B. Senn. 2011b. Impact of a large tropical reservoir on riverine transport of sediment, carbon, and nutrients to downstream wetlands. Water Resources Research **47**.
Lake, J. 1967. Principal fishes of the Murray-Darling River system.
Lehmkuhl, D. M. 1972. Change in thermal regime as a cause of reduction of benthic fauna downstream of a reservoir. Journal of the Fisheries Research Board of Canada **29**:1329-&.
Lemly, A. D., R. T. Kingsford, and J. R. Thompson. 2000. Irrigated agriculture and wildlife conservation: Conflict on a global scale. Environmental Management **25**:485-512.
Lewin, J. C. 1957. Silicon metabolism in diatoms. 4. Growth and frustule formation in Navicula-pelliculosa Canadian Journal of Microbiology **3**:427-&.
Lobell, D. B., M. B. Burke, C. Tebaldi, M. D. Mastrandrea, W. P. Falcon, and R. L. Naylor. 2008. Prioritizing climate change adaptation needs for food security in 2030. Science **319**:607-610.
Lopes, C. B., A. I. Lillebo, J. M. Dias, E. Pereira, C. Vale, and A. C. Duarte. 2007. Nutrient dynamics and seasonal succession of phytoplankton assemblages in a Southern European Estuary: Ria de Aveiro, Portugal. Estuarine Coastal and Shelf Science **71**:480-490.
Lugg, A. 1999. Eternal winter in our rivers: addressing the issue of cold water pollution. NSW Fisheries.
Lugg, A., and C. Copeland. 2014. Review of cold water pollution in the Murray–Darling Basin and the impacts on fish communities. Ecological Management & Restoration **15**:71-79.
Lund, J. W. G. 1950. Studies on Asterionella-formosa hass .2. Nutrient depletion and the spring maximum. Journal of Ecology **38**:15-35.
Lyon, J. P., T. J. Ryan, and M. P. Scroggie. 2008. Effects of temperature on the fast-start swimming performance of an Australian freshwater fish. Ecology of Freshwater Fish **17**:184-188.
Lyytikainen, T., and M. Jobling. 1998. The effect of temperature fluctuations on oxygen consumption and ammonia excretion of underyearling Lake Inari Arctic charr. Journal of Fish Biology **52**:1186-1198.
Maheshwari, B. L., K. F. Walker, and T. A. McMahon. 1995. Effects of regulation on the flow regime of the river murray, Australia. Regulated Rivers-Research & Management **10**:15-38.
Marshall, D. W., M. Otto, J. C. Panuska, S. R. Jaeger, D. Sefton, and T. R. Baumberger. 2006. Effects of hypolimnetic releases on two impoundments and their receiving streams in Southwest Wisconsin. Lake and Reservoir Management **22**:223-232.
Matzinger, A., B. Mueller, P. Niederhauser, M. Schmid, and A. Wueest. 2010. Hypolimnetic oxygen consumption by sediment-based reduced substances in former eutrophic lakes. Limnology and Oceanography **55**:2073-2084.
Medrano, E. A., R. E. Uittenbogaard, L. M. D. Pires, B. J. H. van de Wiel, and H. J. H. Clercx. 2013. Coupling hydrodynamics and buoyancy regulation in Microcystis aeruginosa for its vertical distribution in lakes. Ecological Modelling **248**:41-56.
Merrick, J. R., and G. n. E. Schmida. 1984. Australian freshwater fishes. JR Merrick.
Miles, N. G., and R. J. West. 2011. The use of an aeration system to prevent thermal stratification of a freshwater impoundment and its effect on downstream fish assemblages. Journal of Fish Biology **78**:945-952.
Mitrovic, S. M., L. C. Bowling, and R. T. Buckney. 2001. Vertical disentrainment of Anabaena circinalis in the turbid, freshwater Darling River, Australia: quantifying potential benefits from buoyancy. Journal of Plankton Research **23**:47-55.

Mitrovic, S. M., L. Hardwick, and F. Dorani. 2011. Use of flow management to mitigate cyanobacterial blooms in the Lower Darling River, Australia. Journal of Plankton Research **33**:229-241.

Mitrovic, S. M., R. L. Oliver, C. Rees, L. C. Bowling, and R. T. Buckney. 2003. Critical flow velocities for the growth and dominance of Anabaena circinalis in some turbid freshwater rivers. Freshwater Biology **48**:164-174.

Montesanto, B., S. Ziller, D. Danielidis, and A. Economou-Amilli. 2000. Phytoplankton community structure in the lower reaches of a Mediterranean river (Aliakmon, Greece). Archiv Fur Hydrobiologie **147**:171-191.

Mortimer, C. H. 1941. The exchange of dissolved substances between mud and water in lakes. Journal of Ecology **29**:280-329.

Mortimer, C. H. 1971. Chemical exchanges between sediments and water in the Great Lakes-speculations on probable regulatory mechanisms. Limnol. Oceanogr **16**:387-404.

NHMRC, N. 2011. Australian drinking water guidelines paper 6 national water quality management strategy. National Health and Medical Research Council, National Resource Management Ministerial Council, Commonwealth of Australia, Canberra.

Nicholls, K. H., and P. J. Dillon. 1978. An Evaluation of Phosphorus-Chlorophyll-Phytoplankton Relationships for Lakes. Internationale Revue der gesamten Hydrobiologie und Hydrographie **63**:141-154.

NSW DPI. 2015. Central West Regional Algal Coordinating Committee. URL: http://www.water.nsw.gov.au/water-management/water-quality/algal-information/central-west-regional-algal-coordinating-committee.

Nurnberg, G. K. 2007. Lake responses to long-term hypolimnetic withdrawal treatments. Lake and Reservoir Management **23**:388-409.

Nyholm, N. 1978. A simulation model for phytoplankton growth and nutrient cycling in eutrophic, shallow lakes. Ecological Modelling **4**:279-310.

Olden, J. D., and R. J. Naiman. 2010. Incorporating thermal regimes into environmental flows assessments: modifying dam operations to restore freshwater ecosystem integrity. Freshwater Biology **55**:86-107.

Oliver, R. L., and C. J. Merrick. 2006. Partitioning of river metabolism identifies phytoplankton as a major contributor in the regulated Murray River (Australia). Freshwater Biology **51**:1131-1148.

OneTemp Pty Ltd. 2013. Measure, control, record. From www.Onetemp.com.au.

Paerl, H. W., R. S. Fulton, 3rd, P. H. Moisander, and J. Dyble. 2001. Harmful freshwater algal blooms, with an emphasis on cyanobacteria. TheScientificWorldJournal **1**:76-113.

Paerl, H. W., and J. Huisman. 2008. Climate - Blooms like it hot. Science **320**:57-58.

Petts, G. E. 1985. Impounded rivers: perspectives for ecological management.

Pinckney, J. L., H. W. Paerl, M. B. Harrington, and K. E. Howe. 1998. Annual cycles of phytoplankton community-structure and bloom dynamics in the Neuse River Estuary, North Carolina. Marine Biology **131**:371-381.

Preece, R., and N. S. Wales. 2004. Cold water pollution below dams in New South Wales: a desktop assessment. Water management Division, Department of Infrastructure, Planning and Natural Resources.

Preece, R. M., and H. A. Jones. 2002. The effect of Keepit Dam on the temperature regime of the Namoi River, Australia. River Research and Applications **18**:397-414.

Priyantha, D. G. N., T. Asaeda, S. Saitoh, and K. Gotoh. 1997. Modelling effects of curtain method on algal blooming in reservoirs. Ecological Modelling **98**:89-104.

Ptacnik, R., S. Diehl, and S. Berger. 2003. Performance of sinking and nonsinking phytoplankton taxa in a gradient of mixing depths. Limnology and Oceanography **48**:1903-1912.

R Core Team. 2015. R: A language and environment for statistical computing [Internet]. Vienna, Austria: R Foundation for Statistical Computing; 2013. URL: http://www.r-project.org/.

Raupach, M., J. Kirby, D. Barrett, and P. Briggs. 2001. Balances of water, carbon, nitrogen and phosphorus in Australian landscapes:(1) Project description and results. Canberra: CSIRO Land and Water.

Reinfelds, I., and S. Williams. 2012. Threshold flows for the breakdown of seasonally persistent thermal stratification: Shoalhave River below Tallowa Dam, New South Wales, Australia. River Research and Applications **28**:893-907.

Reynolds, C., and A. Walsby. 1975. Water-blooms. Biological reviews **50**:437-481.

Rolls, R. J., I. O. Growns, T. A. Khan, G. G. Wilson, T. L. Ellison, A. Prior, and C. C. Waring. 2013. Fish recruitment in rivers with modified discharge depends on the interacting effects of flow and thermal regimes. Freshwater Biology **58**:1804-1819.

Rowland, S. J. 1983. Spawning of the Australian fresh-water fish Murray cod, Maccullochella-peeli (Mitchell), in earthen ponds. Journal of Fish Biology **23**:525-534.

Rutherford, J., M. Lintermans, J. Groves, P. Liston, C. Sellens, and H. Chester. 2009. Effects of cold water releases in an upland stream. eWater Cooperative Research Centre, Canberra.

Ryan, T. 2001. Status of cold water releases from Victorian dams. 0731149718, Department of Natural Resources and Environment, Victoria.

Ryan, T., and R. M. Preece. 2003. Potential for cold water shock in the Murray-Darling Basin: A scoping study for the Murray-Darling Basin Commission.

Sadati, M. A. Y., M. Pourkazemi, M. Shakurian, M. H. S. Hasani, H. R. Pourali, M. Pourasaadi, and A. Yousefi. 2011. Effects of daily temperature fluctuations on growth and hematology of juvenile Acipenser baerii. Journal of Applied Ichthyology **27**:591-594.

Sahoo, G. B., and D. Luketina. 2003. Modeling of bubble plume design and oxygen transfer for reservoir restoration. Water Research **37**:393-401.

Sartory, D., and J. Grobbelaar. 1984. Extraction of chlorophyll a from freshwater phytoplankton for spectrophotometric analysis. Hydrobiologia **114**:177-187.

Sattari, M. T., H. Apaydin, and F. Ozturk. 2009. Operation analysis of Eleviyan irrigation reservoir dam by optimization and stochastic simulation. Stochastic Environmental Research and Risk Assessment **23**:1187-1201.

Selig, U., and G. Schlungbaum. 2002. Longitudinal patterns of phosphorus and phosphorus binding in sediment of a lowland lake-river system. Hydrobiologia **472**:67-76.

Sherman, B. 2000. Scoping options for mitigating cold water discharges from dams. CSIRO Land and Water Canberra.

Sherman, B. 2005. Hume Reservoir Thermal Monitoring and Modelling: Final Report: Prepared for State Water as Agent for the Murray-Darling Basin Commission. CSIRO Land and Water.

Sherman, B., C. R. Todd, J. D. Koehn, and T. Ryan. 2007. Modelling the impact and potential mitigation of cold water pollution on Murray cod populations downstream of Hume Dam, Australia. River Research and Applications **23**:377-389.

Sherman, B. S., I. T. Webster, G. J. Jones, and R. L. Oliver. 1998. Transitions between Aulacoseira and Anabaena dominance in a turbid river Weir pool. Limnology and Oceanography **43**:1902-1915.

Shiklomanov, I. A. 2000. Appraisal and assessment of world water resources. Water International **25**:11-32.

Singh, N. K., and D. W. Dhar. 2013. Cyanotoxins, related health hazards on animals and their management: A Review. Indian Journal of Animal Sciences **83**:1111-1127.

Sinokrot, B. A., and H. G. Stefan. 1994. Stream water-temperature sensitivity to weather and bed parameters. Journal of Hydraulic Engineering-Asce **120**:722-736.

Skidmore, R. E., S. C. Maberly, and B. A. Whitton. 1998. Patterns of spatial and temporal variation in phytoplankton chlorophyll a in the River Trent and its tributaries. Science of the Total Environment **210**:357-365.

Sommer, U., Z. M. Gliwicz, W. Lampert, and A. Duncan. 1986. The PEG-model of seasonal succession of planktonic events in fresh waters. Archiv Fur Hydrobiologie **106**:433-471.

Spence, J., and H. Hynes. 1971. Differences in benthos upstream and downstream of an impoundment. Journal of the Fisheries Board of Canada **28**:35-43.

Srinivasan, V., E. F. Lambin, S. M. Gorelick, B. H. Thompson, and S. Rozelle. 2012. The nature and causes of the global water crisis: Syndromes from a meta-analysis of coupled human-water studies. Water Resources Research **48**.

Stanford, J. A., and J. V. Ward. 2001. Revisiting the serial discontinuity concept. Regulated Rivers-Research & Management **17**:303-310.

State Water Corporation. 2009. Dam facts and figures: Burrendong Dam. URL: http://www.statewater.com.au/_Documents/Dam%20brochures/Burrendong%20Dam%20brochure.pdf.

Stringfellow, W., J. Herr, G. Litton, M. Brunell, S. Borglin, J. Hanlon, C. Chen, J. Graham, R. Burks, R. Dahlgren, C. Kendall, R. Brown, and N. Quinn. 2009. Investigation of river eutrophication as part of a low dissolved oxygen total maximum daily load implementation. Water Science and Technology **59**:9-14.

Stumm, W., and J. J. Morgan. 1996. Aquatic chemistry: chemical equilibria and rates in natural waters. Third edition. John Wiley & Sons.

Sulser, T. B., B. Nestorova, M. W. Rosegrant, and T. van Rheenen. 2011. The future role of agriculture in the Arab region's food security. Food Security **3**:S23-S48.

Sweeney, B. W. 1978. Bioenergetic and developmental response of a mayfly to thermal variation. Limnology and Oceanography **23**:461-477.

Sweeney, B. W., and J. A. Schnack. 1977. Egg development, growth, and metabolism of Sigara-alternata (SAY) (Hemiptera-corixidae) in fluctuating thermal environments. Ecology **58**:265-277.

Talling, J., F. Sinada, O. Taha, and E. H. Sobhy. 2009. Phytoplankton: Composition, Development and Productivity. Pages 431-462 *in* H. Dumont, editor. The Nile. Springer Netherlands.

Threlkeld, S. T. 1990. Reservoir limnology; Ecological perspectives (K. W. Thornton. B.L. Kimmel, and F. E. Payne [eds.]). Limnology and Oceanography **35**:1411-1412.

Tisdell, J. G., J. Ward, and T. Grudzinski. 2002. The development of water reform in Australia. CRC For Catchment Hydrology.

Todd, C. R., T. Ryan, S. J. Nicol, and A. R. Bearlin. 2005. The impact of cold water releases on the critical period of post-spawning survival and its implications for Murray cod (Maccullochella peelii peelii): A case study of the Mitta Mitta River, southeastern Australia. River Research and Applications **21**:1035-1052.

Turner, L., and W. Erskine. 1996. Morphometry and stratification of the Bents Basin scour pool, Nepean River, NSW. Wetlands (Australia) **17**:14-28.

Turner, L., and W. D. Erskine. 2005. Variability in the development, persistence and breakdown of thermal, oxygen and salt stratification on regulated rivers of southeastern Australia. River Research and Applications **21**:151-168.

United Nations, Department of Economic and Social Affairs, Population Division. 2014. World Population Prospects: The 2012 Revision, Methodology of the United Nations Population Estimates and Projections, Working Paper No. ESA/P/WP.235.

Vermeyen, T. B. 1997. Modifying reservoir release temperatures using temperature control curtains.

Walker, K. 1980. The downstream influence of Lake Hume on the River Murray. Pages 182-191 *in* W. Williams, editor. An Ecological Basis for Water Resource Management. Australian National University Press, Canberra, Australia.

Ward, J. 1976. Effects of thermal constancy and seasonal temperature displacement on community structure of stream macroinvertebrates. Pages 302-307 *in* G. W. E. a. R. W. McFarlane, editor. Thermal Ecology II. ERDA Symposium Series CONF-750425.

Ward, J. V. 1974. A temperature-stressed stream ecosystem below a hypolimnial release mountain reservoir. Archiv Fur Hydrobiologie **74**:247-275.

Ward, J. V., and J. Stanford. 1983. The serial discontinuity concept of lotic ecosystems. Dynamics of lotic ecosystems **10**:29-42.

Watanabe, M. F., K. Tsuji, Y. Watanabe, K. Harada, and M. Suzuki. 1992. Release of heptapeptide toxin (microcystin) during the decomposition process of Microcystis aeruginosa. Natural toxins **1**:48-53.

Wehr, J. D., and J. P. Descy. 1998. Use of phytoplankton in large river management. Journal of Phycology **34**:741-749.

Wells, K. 2015. Australian farming and agriculture – grazing and cropping.

Welsh, B. L., and F. C. Eller. 1991. Mechanisms controlling summertime oxygen depletion in western Long Island Sound. Estuaries **14**:265-278.

WHO (World Health Organization). 2003. Guidelines for Safe Recreational Water Environments : Volume 1, Coastal and Fresh Waters. World Health Organization (WHO), Albany, NY, USA.

Zalocar de Domitrovic, Y., A. S. G. Poi de Neiff, and S. L. Casco. 2007. Abundance and diversity of phytoplankton in the Parana River (Argentina) 220 km downstream of the Yacyreta reservoir. Brazilian journal of biology = Revista brasleira de biologia **67**:53-63.

Zhou, Q., W. Chen, K. Shan, L. Zheng, and L. Song. 2014. Influence of sunlight on the proliferation of cyanobacterial blooms and its potential applications in Lake Taihu, China. Journal of Environmental Sciences **26**:626-635.

Zouabi-Aloui, B., S. M. Adelana, and M. Gueddari. 2015. Effects of selective withdrawal on hydrodynamics and water quality of a thermally stratified reservoir in the southern side of the Mediterranean Sea: a simulation approach. Environmental monitoring and assessment **187**.