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Novel methods for managing freshwater refuges against climate change in southern Australia

Supporting document 1: Evaluating the utility of cold-water releases ("shandying") for enhancing the resilience of riverine species

Courtney R. Cummings, Ty G. Matthews and Rebecca E. Lester

NOVEL METHODS FOR MANAGING FRESHWATER REFUGES AGAINST CLIMATE CHANGE IN SOUTHERN AUSTRALIA

SUPPORTING DOCUMENT 1:

Evaluating the utility of cold-water releases ("shandying") for enhancing the resilience of riverine species

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Published by the National Climate Change Adaptation Research Facility 2013

ISBN: 978-1-925039-51-1   NCCARF Publication 80/13

Please cite this report as:

Cummings, CR, Matthews, TG & Lester, RE 2013, *Novel methods for managing freshwater refuges against climate change in southern Australia. Supporting Document 1: Evaluating the utility of cold-water releases ("shandying") for enhancing the resilience of riverine species,* National Climate Change Adaptation Research Facility, Gold Coast, 74 pp.

Acknowledgement

This work was carried out with financial support from the Australian Government (through the Department of Climate Change and Energy Efficiency and the National Water Commission) and the National Climate Change Adaptation Research Facility (NCCARF).

The role of NCCARF is to lead the research community in a national interdisciplinary effort to generate the information needed by decision-makers in government, business and in vulnerable sectors and communities to manage the risk of climate change impacts.

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Acknowledgements

We would like to acknowledge the support and leadership of Associate Professor Belinda Robson and the contributions of other members of the project team. We also gratefully acknowledge the support of the National Climate Change Adaptation Research Facility and National Water Commission. We are also extremely grateful to Agnes Lautenschlager who helped with the editing of the final draft and the natural resource managers and other end-users who have provided information and opinions for this study:

Marie Egerrup, Alison Howman and Carolyn Maxwell: Hydro Tasmania, Andrew Greenfield: Mallee CMA, Allan Lugg: NSW Department of Primary Industries, Scott Hardie: Department of, Primary Industries, Parks, Water & Environment, Tas., Geoff Earl and Keith Ward: Goulburn Broken CMA, Christine Hughes: Melbourne Water, Darren White: North Central CMA, Greg, Fletcher: Wimmera CMA, Jayden Woolley: Corangamite CMA, Matthew O'Connell: North East CMA, Beth Ashworth: Victorian Environmental Water Holder, Steve Gatti: AMLR NRM Board, Robert Donahue and Katherine Bennett, Department of Water, WA., Helen Arundel, Steve Wickson, Stephen Ryan, Lachlan Farrington, Adam Bester, Bryce Morden: Glenelg Hopkins CMA, Lorraine Hardwick, NSW Department of Primary Industries, Office of Water, Martin Prendergast, State Water Corporation , Simon Mitrovic and Christine Hill, Water NSW, Louise Searle, Department of Primary Industries, Vic., Wayne Koster, Arthur Rylah Institute for Environmental Research, Department of Sustainability and Environment, Vic., Jamie Pittock, Australian National University, Tom Ryan and Melanie Brennan.

ABSTRACT

As climate change progresses, mean water temperatures in rivers and streams are likely to increase and with longer and more frequent periods of extreme temperatures. As a result, water temperatures are likely to exceed tolerances of a range of taxa more often, altering aquatic ecosystems. Ameliorating these impacts may require consideration and implementation of novel, innovative management techniques. While some of these may not be feasible now, identifying their potential should be assessed now to plan and facilitate any future implementation of these techniques. Cold-water releases, where cooler bottom waters are released from stratified dams, have the potential to offset some of the effects of this increase in temperature. We reviewed the existing literature to determine the prevalence of cold-water releases for the purposes of modifying temperature regimes in rivers. We identified upper and lower temperature thresholds for fish and sensitive invertebrate taxa from literature, including for individual life-history stages where these were known, and established that many Australian rivers are already exceeding these thresholds, particularly upstream of dams. This situation is likely to deteriorate with climate change, having a substantial impact on freshwater biodiversity.

Cold-water releases downstream of dams reduced maximum water temperatures significantly in the examples that we investigated. We found that there were very few case studies, in Australia or overseas, where cold-water releases had been attempted or modelled for the purpose of actively lowering stream temperatures. In the feedback we received from southern Australian natural resource managers, there were few who were currently considering using the technique or were aware of the potential benefits of the approach. There was one exception; an agency that is planning to use planned releases to mitigate high temperature conditions to encourage recruitment in a threatened fish species.

To determine whether cold-water releases are of any value in mitigating high temperature events in the future, many knowledge gaps must be filled before the technique is likely to be feasible. The most urgent of those are the effects of cold-water releases on whole ecosystems, rather than a single species of interest, the existence and use of thermal refuges in Australian river ecosystems, a greater understanding of the temperature cues for fish migration and temperature tolerances of stream biota in general. Until these are addressed, we do not recommend using cold-water releases to ameliorate high stream temperatures. However, the approach may need to be considered for some river systems and become more feasible as climate change progresses, as knowledge gaps are filled, and if new techniques for water releases are developed.

EXECUTIVE SUMMARY

This report presents a literature review examining the current and future utility of cold-water releases (‘shandying’) from large dams as a method for providing cooler water for aquatic life that may enable them to persist as climate change progresses. We address whether targeted releases might be sufficient to sustain populations of native aquatic species and whether they could realistically be delivered to required habitats.

Under climate change, it is expected that aquatic ecosystems in southern Australia will become warmer, drier and less predictable. There are currently few models for how climate will affect in-stream temperatures and air temperatures are often used as a surrogate. However, some evidence suggests that in-stream temperatures may be increasing at a faster rate than air temperatures, potentially exposing some in-stream ecosystems to higher temperatures. Therefore, using air temperatures as a surrogate for water temperatures must be treated with caution and previous authors have emphasised the urgent need for gathering extensive in-stream temperature data.

Temperature is an integral driver of aquatic ecosystems and has both a direct and indirect role in most aspects of stream ecology, including ecological function. Direct, lethal effects can occur when maximum tolerances for a given species are exceeded. Sub-lethal effects include changes to spawning and migration cues, development of eggs, the timing of emergence of adults and body size at emergence. Other indirect effects can also include changes in water quality. For example, nutrient concentration and fluxes, organic matter decomposition rates and dissolved oxygen concentrations are all affected by temperature.

In plants, temperature affects overall production of aquatic macrophytes through altered photosynthetic rates, as well as flowering and seed germination. In invertebrates, water temperature can affect the efficiency of gills, and can change egg incubation periods, hatching success, the duration of hatching and the induction and termination of diapause. In fish, temperature can affect the efficiency of gills and overall metabolic rates, survival, disease and growth rates and reproductive processes. Rapid declines in temperature can also cause ‘cold shock’ which can lead to mass mortality events. Finally, ecosystem processes, such as decomposition and metabolism are also affected. Differences in the impact of temperature on respiration compared with the impact on production are likely to affect whole stream ecosystems. In the absence of more comprehensive data for a wide range of taxa, a proposed threshold of 21°C, which is based on the tolerances of sensitive Australian invertebrate taxa, has been utilised herein to investigate the possible impact of cold-water releases. It is important to note that individual taxa are likely to have a wide range of tolerances to high temperatures, and that this threshold is conservative, but it provides a useful starting point for a discussion regarding the feasibility of the technique. Other, region-specific thresholds could be adopted by individual management agencies in the future, should they decide to apply the technique.

In addition to changes in average stream temperatures, there is also the possibility that streams will contain a range of temperatures at any point in time, creating ‘thermally-distinct’ habitats. These may be associated with groundwater seeps or tributaries where waters of different temperatures are mixing. Overseas, fish have been observed using these thermally-distinct habitats as thermal refuges to avoid high temperature events. Other types of organisms may also use thermal refuges. Such refuges are likely to be increasingly important under climate change as in-stream temperatures approach thermal maxima for more species.

To date, there is no documentation of such thermal refuges in Australia, highlighting the need for research in that area. In the future, when managing environmental flow regimes, water temperature need to be considered, not only to prevent instances of cold shock, but also to maximise the availability of thermal refuges, assuming they are utilised in Australian freshwater environments.

In the past, the primary point of focus on cold-water releases has been the potential for negative impacts associated with the release of cold, de-oxygenated waters from the bottom of large reservoirs. However, there is also the potential for positive impacts of cold-water releases, should they be appropriately managed. Cold-water releases lower summer water temperatures, but also result in elevated winter temperatures, reduced seasonal and diel variability in temperature, and later summer peaks in the timing of maximum temperature. These changes in water temperature are known to influence biota downstream of dams, but the exact effect is difficult to separate from the effect of changes in water availability. Managers overseas are starting to use planned cold-water releases to manipulate in-stream temperatures to improve habitat for salmonid spawning and recruitment. The success of these releases depends on the ability to maintain desirable temperatures, as well as the availability of water when required, and the location of dams capable of releasing cold water relative to the location of habitat at risk of high temperature events.

Maintaining water temperatures below the upper temperature tolerances of biota is an obvious target for the use of cold-water releases. Past studies have identified upper temperature tolerances for a range of organisms with varying degrees of accuracy and temperature tolerance data for fish and invertebrates are scarce, and more research in this area is required. The thermal optimum for submerged aquatic macrophytes is thought to be up to 32°C, but floating and submerged plants tend to grow in shallow systems or the edges of deeper systems where temperatures may fluctuate widely. Upper thermal tolerances for macroinvertebrate taxa range dramatically from 11°C to 43°C. An upper tolerance of 21°C has been suggested for maintaining diversity in invertebrate assemblages, particularly sensitive taxa, which is based on data collected in Western Australia (Davies *et al.* 2004). Upper tolerances for fish also range widely, between 20 and 44°C for native Australian species but tolerances for aquatic mammals and reptiles are poorly understood. However, it is important to note that a focus on upper temperature tolerances masks the impact of the whole temperature regime on aquatic assemblages and on different life-history stages of individual taxa, as well as any potential for acclimatisation. Eggs, for example, tend to be the most sensitive life-history stage for fish and increased temperatures can result in increased egg mortality or faster development which can then lead to a smaller size at maturation. The survival and fitness of adult fish may also be influenced by elevated temperatures due to alterations to their reproductive capacity and metabolic rates. However, it is possible that fish can acclimatise to the temperature in which they develop. Therefore, fish may have some capacity to adapt to different water temperature regimes, although the extent of this capacity is largely unknown.

Researchers overseas have developed diagrams overlaying the temperature requirements of various species on thermal regime plots both upstream and downstream of dams. Diagrams of that nature highlight the overlap between temperature requirements and actual temperatures, so that any potential mismatch can be identified and potentially managed. This approach is better able to incorporate species- and region-specific differences in temperature requirements than the single threshold mentioned above.

We attempted to reproduce those figures for two native fish species; Murray cod (*Maccullochella peelii peelii*) and river blackfish (*Gadopsis marmoratus*). Temperature requirements for many life-history stages were unknown, particularly for the river blackfish, despite both species being of high conservation importance, limiting our ability to identify potential issues in the example streams used. We did identify some instances where optimal growth thresholds for the Murray cod were exceeded upstream of Hume Dam. Diagrams such as these are likely to be of use for species with adequate biological knowledge as a tool to assist managers but, in the short term, more detailed information is needed on the temperature tolerances of different life-history stages for target species.

Three case studies were identified that investigated the potential for using cold-water releases to meet the thermal tolerances of various fish species. All three case studies used mathematical models to simulate the effect of dam releases on river temperatures, and the impact on fish was then inferred or modelled. All three also focused on cold-water, salmonid species, rather than warm-water species such as most Australian natives. The first case study found that regulating dam release timing and magnitude could affect the suitability of downstream temperature regimes for trout spawning and recruitment. This suggests that multiple off-takes and shandying of water of different temperatures are not always required to influence downstream conditions. However, no single manipulation was able to maximise all thermal targets, suggesting that a compromised regime may be needed. The second case study focused on ameliorating elevated summer temperatures downstream of a hydroelectric dam to improve fish recruitment. Alterations to dam releases had the capability to influence in-stream temperatures and improve recruitment, although it was still low compared with other nearby river systems. The final case study focused on breeding conditions for introduced trout in Australia and found that raising average temperatures in summer by 2°C was likely to lead to significant decreases in growth and increased mortalities. While these case studies focus on salmonids, none of which are native to Australia, similar modelling exercises could also provide information to inform the management of native species.

We investigated existing temperature data upstream and downstream of six dams in southern Australia, including three that were identified by Ryan *et al.* (2001) as high priority for management to address cold-water release issues. Those three reservoirs already have the capacity to reduce the frequency of high temperature events which may influence sensitive taxa downstream of dams. We compared temperature data against the tentative 21°C threshold for invertebrate diversity, as an example of how the need for cold water releases could be assessed. We did not have access to data regarding release strategies at each of the dams, and so have assumed that dam operations are consistent among the three. This is unlikely to actually be the case.

At all dams, the proportion of time that the threshold of 21°C was exceeded downstream of the dam was lower than upstream of the dam, although there was less difference for lower priority dams. In some instances, the temperature further downstream had returned to similar levels to those upstream, but this effect was not consistent among dams. Maximum water temperatures downstream of dams were found to be significantly lower than maximum temperatures upstream or on adjacent tributaries (with an average of 2°C difference across the record assessed). No significant differences were identified in minimum or mean temperatures. However, the available data were sparse, provided little spatial resolution and, in some instances, included limited numbers of years.

This limits our ability to identify how often temperature thresholds are likely to be exceeded or to assess the downstream extent of these cold-water releases. Dam-specific differences in release patterns are also likely to influence downstream impacts. The distance to which cold-water releases are able to influence downstream temperatures will depend on the river system in question. Amongst others, the location and size of impoundment, the volume of each release, the storage water level and the geomorphology and channel form downstream of the dam will drive the impact of individual releases. Research from the USA indicates that any dam with a wall higher than 5 m has the capacity to stratify, meaning that colder bottom waters would be available for potential cold-water releases. Similar information regarding a threshold for stratification for Australia was not available. However, the thermal signature of dam releases has been detected more than 400 km downstream in some river systems, suggesting there is significant potential for releases to influence large tracts of river habitat.

Natural resource managers were contacted about current and future management plans to assess the use of cold-water releases as a deliberate strategy. At the time of the survey, no managers were undertaking cold-water releases to moderate high water temperatures because current management had focused on minimizing potential negative effects of cold-water releases rather than considering potential positive outcomes. To supplement the responses of the catchment managers that were contacted, seasonal watering plans for each catchment in Victoria were also evaluated.

Only one of the Victorian strategy documents referred to the use of cold-water releases in this context. In the Yarra River, releases between 9 and 11°C are planned when summer temperatures approach a threshold of 25°C. This is intended to encourage spawning of Australian grayling (*Prototroctes maraena*). However, at the time of writing, this strategy had not been implemented. Furthermore, none of the other managers were considering implementing cold-water releases in the future, although several expressed interest once the concept had been explained. Most suggested that demonstrating a successful implementation elsewhere would be needed before they would consider incorporating the strategy in their own planning.

There are three potential strategies that that could be implemented by natural resource managers. Dam releases could be modified in temperature (i.e. by mixing cooler bottom waters with warmer summer waters in different proportions, ‘shandying’) to mimic the natural temperature of downstream waters. This strategy is likely to require multiple dam off-takes and regular reconfiguration of those off-takes to change water temperatures, which may be prohibitively expensive, particularly for frequent temperature changes. Cold-water releases could also be used to limit the frequency and extent of high temperature events, as a part of an environmental flow strategy. This matches the strategy being implemented in the Yarra River system, and may be the most feasible, as it would require less water, particularly in systems that suffer from large extractions. Uncertainty remains regarding the volumes of water required to redress high temperature events and any potential impacts of high flows to ameliorate temperature when rivers would naturally have low flows. It is also likely that environmental water will be of short supply during these critical periods. The final strategy that could be implemented is to facilitate movement of fish between existing thermal refuges. Given the lack of knowledge on the location and use of thermal refuges by stream biota in Australia, this strategy is not currently feasible.

In conclusion, cold-water releases have the potential to ameliorate high in-stream temperatures in a substantial number of southern Australian river systems, and potentially for long distances.

However there are substantial knowledge gaps that need to be filled before the strategy can be seriously considered by managers, other than as a test case. The impact of releases on whole aquatic assemblages must be monitored, rather than focusing on a single species. Before and during any release, dissolved oxygen concentrations, and the location of the cold-water lens, as well as release temperature must be monitored. Fundamental knowledge gaps regarding the existence and use of thermal refuges by Australian species must be filled, particularly for warm-water species, and the prevalence of these refuges downstream of dams. All of these factors, as well as channel topography and hydrology, have to be known for each dam individually, before cold-water releases can be considered for a specific context, and in many instances there may be a mismatch between the location of suitable dams and the location of streams suffering from high temperature events.

We recommend three key steps to start filling important knowledge gaps: collect fine-scale temperature data in a range of Australian stream systems to identify the location and use of thermal refuges; develop in-stream temperature models to enable the potential impact of releases to be simulated and validate these with experimental releases; and collect information on the thermal migration cues of Australian fish as a starting point for considering the release of cold water to encourage fish migration. Until these and other key knowledge gaps are filled, the technique of releasing cold-water to influence in-stream temperatures is unlikely to be feasible, although this may change as climate change progresses.

1. INTRODUCTION

In order to meet the challenges of human-induced climate change, natural resource managers will need novel methods of enhancing refuge function in order to protect biodiversity and ecosystem function. There are many possible strategies that individual species can employ to find refuge from adverse conditions, and it is likely that there are direct links between refuge qualities and their ability to support biodiversity.

This report forms the output for one of four sub-projects in a National Climate Change Adaptation Research Facility (NCCARF)-funded project entitled 'Novel methods for managing freshwater refuges against climate change in southern Australia'. The four sub-projects investigate different types of refuge and how each may be able to be exploited to maintain biodiversity under climate change. The subject of this report, sub-project 1 ('Evaluating the utility of cold-water releases ['shandying']') addresses the provision of cooler water for aquatic life (including life-history stages such as spawning) and whether dam releases may be able to be manipulated as a tool to avoid exceeding maximum temperature thresholds.

Within the project, we draw a distinction between refugia, which exist over evolutionary time scales (i.e. those that are effectively permanent and defined geographically so that species can adapt and persist), and refuges, which are defined functionally as providing opportunities for parts, and not necessarily all, of a species' life-history stages to be completed. These definitions are consistent with the primary and secondary uses of the term "refugium" in the Freshwater Biodiversity NARP.

We present a literature review examining whether targeted releases could be used to sustain populations of native aquatic species and whether they could realistically be delivered to the required habitats. This review is focused on the role of temperature in regulating freshwater ecosystems and whether cold-water releases have been used elsewhere to enhance those ecosystems. We do not provide a comprehensive review of the climate change literature, but give a brief introduction to the likely impacts of climate change on freshwater environments and the potential impacts of temperature on freshwater biota. We then provide an outline of the use of cold-water releases to enhance freshwater ecosystems, based on Australian and overseas published literature, review individual case studies where this technique has been investigated and provide a summary of the current views of natural resource managers in eastern Australia regarding cold-water releases. We also present possible future scenarios by which managers in Australia could attempt a cold-water release, and make recommendations regarding the monitoring that should accompany such a trial. In undertaking this review, we utilised the scientific literature and grey literature wherever possible. It should be noted that some potential sources in the grey literature is difficult to access and the suite included here may not be comprehensive.

1.1 Changes in water temperatures under climate change

Observed and predicted impacts of human-induced climate change are now widely reported in numerous technical reports and the published literature, including Australia (e.g. Koehn 2011). Generally, the outlook is not positive for many areas of Australia, with both sea-surface temperatures and air temperatures increasing faster than the global average (Lough and Hobday 2011). It is widely acknowledged that reduced stream flows and increasing stream temperatures associated with human-induced climate change pose two of the greatest threats to the ecology and maintenance of biodiversity in lotic ecosystems (e.g. Chessman 2009, Caissie 2006, Webb *et al.* 2008).

Environmental flows may be able to address the former threat. Here, we focus on the latter.

The impacts of human-induced climate change appear to be more widely publicised for marine environments than freshwater ecosystems, possibly owing to the presence of iconic ecosystems such as the Great Barrier Reef (Koehn 2011). Air temperatures are warming faster than ocean temperatures (Hobday and Lough 2011), which potentially places freshwater ecosystems, which tend to be much smaller in size and thermal capacity than marine systems, under greater threat from the predicted impacts of climate change than their marine counterparts (Koehn 2011). In addition, descriptions of longer-term changes and predictive models for freshwater ecosystems are less numerous than those for marine ecosystems because continuous physico-chemical data are not as widely available for lakes and rivers. As a result, air temperatures are often used as a proxy for freshwater systems, where models exist (Lough and Hobday 2011).

However, in-stream temperatures may be increasing faster than air temperatures, implying that caution is required when using air temperatures as a surrogate. Therefore, assessments of changes in water temperatures over the longer term are not possible for nearly all Australian freshwater ecosystems. However, examples of increasing water freshwater temperatures are evident in parts of the Northern Hemisphere (e.g. US and Europe, Kaushal *et al.* 2010, Webb and Nobilis 2010). Chessman (2009) found that water temperatures in some NSW streams have actually increased at a higher rate (0.12°C per annum) than air temperature (0.09°C per annum) over a recent 13-year period (1994 – 2007). Average air temperatures across Australia have increased by nearly 1°C during the last 60 years (CSIRO 2007), which is likely to have corresponded with an approximate 0.6 - 0.8°C increase in stream temperatures (Morrill *et al.* 2005). However, the example by Chessman (2009) highlights some of the uncertainty associated with these estimates and actual warming rates for freshwater environments may be higher than previously expected. For inland Australia, average temperatures are predicted to rise by up to 1.2°C by 2030 (CSIRO 2007), but predicted temperatures after 2030 diverge widely depending on the scenario modelled, and it is likely that temperatures will follow the medium- to high-emission scenarios rather than the low-emission scenario (Hobday and Lough 2011). The combination of predicted increases in average stream temperatures and variability will place greater stress on lotic biota. The tolerance limits of sensitive stream taxa are already being exceeded in many streams in south-western Australia (Davies 2010) and this may also be true elsewhere. Therefore, innovative human intervention measures will be required to mitigate increasing stream temperatures in order to conserve stream biodiversity. Riparian revegetation (see Cook *et al.* 2013) and environmental flow releases appear to be the two most commonly proposed methods for ameliorating predicted increases in stream temperatures. Another potential option is to consider the careful management and incorporation of hypolimnetic releases as part of environmental flow releases to ameliorate rising stream temperatures downstream of large water storages. Therefore, the aim of this review is to identify the potential feasibility of using hypolimnetic releases to provide a refuge against increasing summer temperatures for stream biota in southern Australia.

1.2 Importance of temperature for the maintenance of structure and function in lotic ecosystems

Temperature is an integral part of the ecological function in lotic systems, having both direct and indirect effects on most aspects of stream ecology (Caissie 2006, Webb *et al.* 2008). Direct effects include the exceedance of tolerances including those required for successful reproduction. Indirect effects include the exclusion of species based on thermal preference or alteration of resilience to other potential stressors, alteration of metabolic and physiological abilities and alteration of overall productivity. For example, a 1°C rise in stream temperatures increases the rate of metabolism in cold-blooded animals by about 10% (Gordon *et al.* 1997). Higher temperatures also enhance chemical reactions and biological activities (Bergfur and Friberg 2012, Brown *et al.* 2004). Temperature is also a key factor determining the activity of organisms in ecosystems (Friberg *et al.* 2009), with higher temperatures stimulating biological activities at least within physiological limits (Bergfur and Friberg 2012).

Water temperature influences many different life stages of stream fauna. Development cues for freshwater fish and insects can include a summation of thermal units as well as absolute temperatures (Olden and Naiman 2010). This means that short-term high temperatures may play at least as big a role as smaller longer-term increases in average temperatures. Modification of the triggers for migration, spawning, egg development and hatching of many fish species can occur with alteration to temperature regimes (Cook *et al.* 2013). This can then lead to premature emergence of adults, possibly at times when climatic conditions in the terrestrial environment are unsuitable for adult survival or when few mates from adjacent sites are present (Cook *et al.* 2013, Ryan *et al.* 2003 and references therein). The overall fecundity of many organisms can also be reduced by temperature changes because larvae mature at smaller sizes in warmer water and smaller insects produce fewer eggs (Vannote and Sweeney 1980).

One potential consequence of increased temperatures is a shift in the distribution of a range of organisms, including aquatic species. Plants and animals are beginning to respond to temperature changes associated with human-induced climate change by exhibiting large shifts in their geographic range (e.g. Walther *et al.* 2002, Root *et al.* 2003, Sorte *et al.* 2010). A review of published literature identified 129 marine species exhibiting shifts in range boundaries (from 55 studies), with 75% shifting in a pole ward direction (Sorte *et al.* 2010). These species ranged from primary producers, invertebrates and fish to bird species. In a global meta-analysis of range shifts for both terrestrial and aquatic species, Root *et al.* (2003) found 80% of nearly 1500 species had shifted in a pole ward direction. Shiel *et al.* (2004) has also shown large shifts in the range boundaries of many species in an 18-year sampling period on the coastline of California. A 3.5°C increase in water temperature over ten years caused significant community-wide changes in 150 species of algae and invertebrates, with many cold water species being replaced with warm-water species (Shiel *et al.* 2004). Many latitudinal shifts for a range of species have also been documented, including shifts in distribution to high altitudes for tree coverage in both New Zealand and Europe (Walther *et al.* 2002). This is particularly evident for temperate organisms, which tend to be shifting further toward the poles than tropical species. In cases where global warming results in thermal tolerances being approached or surpassed, range shifts, range expansions and assemblage turnovers are often observed (Morrongiello *et al.* 2011). Southward shifts of climatic zones have already been observed for Australian coastal waters between 10.6°S and 29.6°S. Warming is leading to southward shifts of over 200 km for the eastern coast and approximately 100 km for the western coast (see Lough 2008). Although these trends have mainly been described in marine and terrestrial species to date, it is likely that similar pole ward shifts in distribution are possible in freshwater species in the future in response to changes in temperatures, assuming that such a shift is possible.

1.3 Water Quality

Temperature has a pronounced effect on the water quality in lotic systems. Many aspects of water quality are affected, including solute and pollutant fluxes, nutrient concentrations, organic matter and sediments and dissolved oxygen concentrations (Webb *et al.* 2008, Olden and Naiman 2010).

Generally, gas solubility decreases and mineral solubility increases with increasing water temperature, potentially leading to lower levels of dissolved oxygen combined with higher concentrations of nutrients and contaminants, including heavy metals (Webb *et al.* 2008, Olden and Naiman 2010). The toxicity of contaminants and the efficacy of water treatment, as well as taste and odour are also affected by water temperature. Warm water and nutrient enrichment can lead to cyanobacteria blooms which can produce toxins and decrease water quality (Bormans *et al.* 1997).

1.4 Plants and phytoplankton

Temperature (along with light availability) can affect aquatic plants and phytoplankton, either directly or indirectly, by determining their distribution and productivity, the timing of reproductive events, such as the induction of flowering and the germination of seeds (Barko *et al.* 1986, Santamaria and Van Vierssen 1997). These changes to primary productivity can then lead to subsequent changes in both community and foodweb structure (Shurin *et al.* 2012). For example, pond experiments showed that increased temperatures can indirectly cause declines in periphyton and phytoplankton, which were attributed to a shift towards top-heavy food webs with lower biomass of benthic and pelagic primary producers (Shurin *et al.* 2012).

In contrast to the indirect effect of increased temperature described directly above, higher temperatures tend to directly influence plants by promoting greater productivity, with concomitant increases in both shoot length and shoot number (Barko *et al.* 1986), providing that temperatures fall within their thermal tolerance range. Shoot-to-root ratios commonly increase as temperatures rise within species-specific tolerances, and decreases once tolerances have been exceeded (Wilson 1988). Overall, the temperature response in angiosperm growth follows an optimum curve, with optimal growth occurring at species-specific optimal temperatures. Higher temperatures can also lead to increased chlorophyll concentration in some plants (Barko *et al.* 1986), but there are often species-specific responses in photosynthesis rates (Santamaria and Van Vierssen 1997, Pilon and Santamaria 2002).

Aquatic plants appear to be more resilient to changes in thermal regimes than terrestrial plants, with adaptations and different morphological characteristics apparent at different temperatures. Some plant species have adapted their photosynthetic process to be enhanced at the temperature of which they have been subjected. For example, populations of the terrestrial sea pea *Lathyrus japonicus* exhibit more pronounced respiration at high latitudes compared with low latitude populations, which may reflect a higher growth rate being advantageous during short high-altitude growing seasons (Lechowicz *et al.* 1980).

Overall, plants can be influenced by temperature directly, through increased growth and production, or indirectly, by increasing competition or shifts in food webs. It is expected that plant distribution will shift in the future in response to increased temperatures, due to changes in local patterns of reproduction and increased stress

(Short and Neckles 1999). These changes will likely include altered plant community structure, which can have a large impact on food resources and habitat for biota, and thereby whole ecosystems (Barko *et al.* 1986).

1.5 Invertebrates

In general, aquatic invertebrates respond to the entire thermal regime, which includes absolute temperature levels, seasonal and diel ranges, rates, and the timing and duration of thermal events, rather than just to average temperature (Ward 1992). Invertebrates are affected by temperature directly via their metabolic processes (e.g. respiration in gilled animals is limited above 30°C, due to lower solubility of oxygen in warm water), fecundity, egg incubation period, hatching success, duration of hatching, and the induction and termination of diapause (Ward 1992). The effects of temperature on fecundity and the responses of eggs to thermal conditions influence the distribution patterns of aquatic insects and the competitive position of a species at a given locale (Ward 1992).

Data are limited for the thermal tolerances of many Australian stream invertebrates (but see Cook *et al.* 2013). In New Zealand, studies of the distribution of invertebrates in volcanic thermal streams suggest the presence of taxa with a high tolerance for elevated temperatures but other taxa were more sensitive and exhibited lower maximum temperature tolerances. For example, some dipterans (flies) were present in thermal streams between 34 and 43°C (Winterbourn 1969), while specific coleopterans (beetles) and other dipteran species (flies) have been found in streams with temperatures over 44°C (James 1985). In contrast, particularly sensitive plecopteran taxa (stoneflies) were uncommon in streams where summer temperatures exceeded 19°C (Quinn and Hickey 1990).

The upper thermal tolerance of 12 New Zealand stream invertebrates has been assessed in laboratory trials (Quinn *et al.* 1994). A wide range of upper thermal tolerances were found, with lethal temperatures (LT_{50}) ranging from 24.5 to 34.0°C after 48 hours and 22.6 to 32.6°C after 96 hours, following acclimation at 15°C (Quinn *et al.* 1994). The taxa most sensitive to higher thermal conditions belonged to the Ephemeroptera (mayflies) and Plecoptera (stoneflies), while the most tolerant taxa belonged to the trichopteran family Conoesucidae (caddisflies), coleopterans and two molluscs (clams and snails), each with LT_{50} values above 30°C (Quinn *et al.* 1994). Two crustaceans, an atyid shrimp and an amphipod, were moderately tolerant.

These results suggest that many New Zealand invertebrate taxa may already be exposed to their upper lethal temperatures, which may be limiting the distribution and abundance of invertebrate taxa in New Zealand streams. It is possible that a similar situation is occurring in Australia (Cook *et al.* 2013), however further laboratory trials and field sampling are needed for many Australian species.

1.6 Fish

The importance of water temperature for fish in lotic systems is generally well known. Water temperature is a critical parameter for survival, growth and all reproductive processes in fish, particularly for river reaches directly below large water storages (Cadwallader 1978). Temperature influences metabolic rate and the speed of muscle contractions, which then influence activity and behaviour, and thus swimming speed, prey-capture ability and food assimilation (Krause *et al.* 2005). Regular and prolonged periods above upper and lower temperature tolerances cause increased stress, increased susceptibility to disease (Cairns *et al.* 2005), potential reductions in fitness of the population (Krause *et al.* 2005) and impaired growth, foraging and competitiveness (Pörtner and Farrell 2008). Rapid and sudden decreases in temperature can lead to a condition known as a 'cold coma' (Berry 1988), which can result in mass mortality events. Mass mortality of fish associated with changes in temperature has been observed in tropical Australian streams (Townsend *et al.* 1992), but we are unaware of such events in temperate regions.

Water temperature is critical for all reproductive processes in fish including gamete development and maturation, ovulation and spermination, spawning, embryogenesis, hatching and larval and juvenile development and survival (Pankhurst and Munday 2011). Mature adults rely on temperature increases to cue maturation in spring and early summer spawners (see Pankhurst and King 2010). The embryos and larvae of fish species tend to be the life-history stages most sensitive to the effects of temperature (Gillianders *et al.* 2011). Changes in the age-structure of fish populations can also occur, through shifts in the thermal windows present for spawning adults, eggs and larvae (Gillianders *et al.* 2011). Reductions in the maximum sizes of fish also occur with increased temperatures, which may affect the overall fecundity of individuals.

Many species seek thermal refuge during events of higher temperature (Caissie 2006). Ebersole *et al.* (2001) showed that in northern American streams, approximately 10–40% of fish were observed close to thermal refuges at midday and such an aggregation of fish resulted in higher densities than those observed elsewhere in the stream. Cold-water areas or patches can include cold-water tributaries, lateral seeps, deep pools and cold backwater areas (Ebersole *et al.* 2003a). Many fish species have been observed seeking refuge in colder tributaries during warmer temperature events, particularly salmonids (i.e. salmon and trout) (Kaya *et al.* 1977, Cunjak *et al.* 1993, Ebersole *et al.* 2001, 2003a). Deep pools (with a cold-water source) have been shown to be important thermal refuges for trout, even preferring such habitats when dissolved oxygen content is low (Elliott 2000, Bilby 1984; Matthews *et al.* 1994, Matthews and Berg 1997). Overall, there are numerous publications associated with the use of thermal refuges by North American fishes. In contrast, very little is known about thermal refuge use by warm-water native species in Australia, highlighting a need for similar research here in Australia.

1.7 Ecosystem Processes

Changes in stream temperatures will affect ecosystem processes, such as leaf litter processing, respiration, primary production and denitrification (Acuna *et al.* 2008). The specific effect of temperature on these processes is summarised in this section.

1.7.1 Leaf litter processing

Temperature and the concentration and degradation of organic carbon have strong impacts on decomposition (Allan 1995). Research in Portugal and France has shown that temperature affects the distribution, diversity, growth and reproduction of aquatic fungi (Geraldes *et al.* 2012), with subsequent effects on plant litter decomposition. For instance, a temperature increase by 5-10°C can stimulate fungal growth and reproduction, leading to a twofold increase in plant litter decomposition (Fernandes *et al.* 2009, Ferreira and Chauvet 2011, Geraldes *et al.* 2012). However, fungal species diversity generally decreases with increasing temperature (Fernandes *et al.* 2012), but leaf-litter decomposition can be more affected by temperature than by fungal diversity (Geraldes *et al.* 2012). Furthermore, temperature can influence the reproduction of fungal species, with species that produced spores at lower temperatures failing to sporulate at higher temperatures (Fernandes *et al.* 2012). Yet again, effects of temperature on plant litter decomposition by fungi seem to be dependent on species-specific tolerances, emphasising a need for research on local species. Fungal species in Australia are likely to be quite specialised give the specific properties of eucalypt leaves, which make up large proportions of leaf litter in Australian streams. Therefore, fungal species in Australia may exhibit different responses to altered temperatures to fungi described elsewhere.

1.7.2 Metabolism

Metabolism is an essential ecosystem process by which energy and materials are transformed within an organism and exchanged between organisms and the environment (Gillooly *et al.* 2001). Temperature influences the metabolic rate of organisms through effects on the rates of biochemical reactions. Simple reaction kinetics models describe the effect of varying temperatures as a function of activation energy, and such models have been used to fit the metabolic rates of microbes, ectotherms, endotherms and plants for temperatures ranging between 0 and 40°C (Gillooly *et al.* 2001). Generally, whole organism metabolic rates increase exponentially with temperature (Hemmingsen 1960, Kleiber 1932 in Gillooly *et al.* 2001).

Elevated temperatures generally raise ecosystem respiration and consequently oxygen consumption. The Boltzmann factor (relating activation energy to metabolic rate) also applies to ecosystem processes, such as respiration and primary productivity (Acuna *et al.* 2008). Primary production and respiration each respond differently to changes in temperature because of their different activation energies. Thus, with a temperature increase from 0 to 30°C, primary production increases fourfold, while respiration increases 16-fold (Acuna *et al.* 2008) leading to potential oxygen depletion. Such differences in the response of primary production compared to respiration will influence the metabolism of the ecosystem as a whole.

An increase in oxygen consumption rates of 30-40% has been predicted in response to a theoretical temperature increase of 2-3°C in the absence of resource limitation in three lowland streams during summer in North Zealand, Denmark (Sand-Jensen and Pedersen 2005). However, several marine and terrestrial studies have stressed that predicted changes in respiration associated with climate change predictions may be overestimated due to a lack of consideration of resource availability (e.g. Gifford 1995). For example, long-term experiments in forest and grassland ecosystems have demonstrated that the production-to-respiration ratio is relatively insensitive to temperature because respiration is effectively limited by the supply of carbohydrate fixed through photosynthesis (Gifford 1995, Hartley *et al.* 2006).

In freshwater systems, organic matter is not usually a limiting resource because many river systems have input from terrestrial systems, which exceeds respiration rates (Acuna *et al.* 2008). Nutrient and organic carbon availability will, however, influence both the respiration rate and activation energy (Acuna *et al.* 2008), thus affecting system metabolism. These complex interactions make predicting the effect of changes in stream temperature on in-stream metabolism difficult, and the impact is likely to vary at different locations.

2. THERMAL REFUGES FOR STREAM BIOTA

In addition to averaged stream temperatures having the ability to shape aquatic ecosystems, there is also the potential for 'thermally-distinct habitats' to create refuges for biota during periods of temperature stress (e.g. during short-lived high temperature events). Such thermally-distinct habitats incorporate localised areas that have a different temperature from the surrounding waters. They are likely to occur around groundwater seeps and tributaries (Magnuson *et al.* 1979) as well as lateral seeps, deep pools and cold backwater areas (Ebersole *et al.* 2003a), as waters from different sources mix with existing stream waters. Channel morphology and heterogeneity plays a critical role in the maintenance of stream temperatures, by buffering or insulating against rapid changes in temperature (Pool and Berman 2001). For example, shallow systems with greater surface areas are likely to warm up more quickly than deep channels or systems with deep pools that contain larger thermal mass (Webb and Zhang 1997).

There are examples where cold-water refuges are critical for the survival of some overseas fish species. For example, as indicated above, adult salmonids are known to utilise cooler patches of water to avoid high in-stream temperatures (Kaya *et al.* 1977; Cunjak *et al.* 1993; Elliott 2000; Ebersole *et al.* 2003a) and trout appear to move to cooler areas when temperatures exceed approximately 22°C (Ebersole *et al.* 2001). Ebersole *et al.* (2001) showed that in northern American streams, approximately 10–40% of fish were observed close to thermal refuges at midday. These aggregations of fish resulted in higher densities than those observed elsewhere in the stream. Deep pools (with a cold-water source), for example, have been shown to be vital thermal refuges for trout, even preferring such habitats when dissolved oxygen content was low (Bilby 1984; Matthews *et al.* 1994; Matthews and Berg 1997, Elliott 2000). It is also likely that other life-history stages (e.g. spawning, larvae), as well as other in-stream biota (e.g. invertebrates and plants) are able to take advantage of heterogeneous stream temperatures. Overall, there is much information about northern American fish utilising thermal refuges, but little is known about warm-water native species in Australia, highlighting a great need for similar research here.

These thermal habitat patches are likely to become more important with increasing temperatures associated with climate changes. To date, temperature, along with the delivery of sediment, nutrients and organic material, are rarely considered in the planning and implementation of environmental flows (Lake *et al.* 2007, Olden and Naiman 2010). Carefully-managed cold-water releases could be considered for future releases from regulated river systems as part of environmental flow releases to create ‘artificial’ cold-water refuges (e.g. possibly by mimicking the natural thermal regime in a manner analogous to the hydrologic regime as much as is possible, Poff *et al.* 2010) for fish and other lotic flora and fauna. In order to do this, temperatures would need to be altered gradually to prevent ‘cold shock’ and a loss of equilibrium in aquatic organisms, particularly during releases from hydroelectric dams (Krause *et al.* 2005).

Currently, there are few examples where refuges have been the active target of restoration works, although there are examples of where artificial habitats have subsequently been used as a sanctuary or refuge from disturbance (Lake *et al.* 2007). Focusing on the dispersal ability of aquatic organisms may enable managers to target areas within that dispersal range for restoration or maintenance, therefore providing regular areas of appropriate habitat that may act as ‘stepping stone’ refuges between remnant high-quality habitats (Robson *et al.* 2008, 2012). While this report focuses on the role of temperature, other factors are also of importance, such as maintaining dissolved oxygen levels, maintenance of physical habitat, food resources and management of nutrient levels. The following section focuses on some of the potential effects of cold-water or hypolimnetic releases.

3. THERMAL IMPACTS OF HYPOLIMNETIC RELEASES

Most of the ecological research associated with hypolimnetic (or cold-water) releases has focused on negative rather than positive impacts. These negative impacts can include large changes in the ambient temperature of waters downstream of dams, changes in the dissolved oxygen concentration, impacts on the cues for spawning and migration of biota within the system (Olden and Naiman 2010), and changes in the availability of contaminants and the uptake of heavy metals due to interactions with temperature and dissolved oxygen concentrations (e.g. Couillard and Chartier 1993, Noyes *et al.* 2009, Graney *et al.* 1984). In many instances, dam releases are now actively managed to mitigate 'thermal pollution' and its associated effects downstream (e.g. by shandying of hypolimnetic and epilimnetic waters or by only releasing epilimnetic water). Negative effects, including those associated with low oxygen levels, have been extensively reviewed (see Ryan *et al.* 2001 and Olden and Naiman 2010), thus this review will concentrate on the potential positives effects.

Lowering of summer temperatures is the most commonly reported effect of hypolimnetic releases, but other effects include elevated winter temperatures, reduced seasonal and diel amplitude, delay of the summer peak and winter temperatures, rapid temperature reductions (Ryan *et al.* 2001). Hypolimnetic releases are known to affect biota downstream of dams (Olden and Naiman 2010) but these effects are not always clear. Vinson (2001) documented large changes in invertebrate richness and abundance following dam construction, but did not find improvements in macroinvertebrate assemblages following the installation of multiple off-takes to reduce thermal effects. Jackson *et al.* (2007) compared unregulated and regulated streams and was unable to differentiate between effects associated with hydrology and those associated with temperature. The effect of elevated winter temperatures, which occur in cold-winter climates where large dams store water at temperatures higher than ambient water temperatures upstream, is not well understood.

Despite the knowledge gaps, scientists and managers are starting to consider using cold-water releases as a means to alter the ecological character of rivers below a dam to produce more suitable conditions. Cold-water releases are being used in some locations worldwide to reduce water temperatures to provide environments for desirable cold-water fishing communities, predominantly for trout, salmon and walleye (Olden and Naiman 2010). In some instances this takes advantage of the releases from hydroelectric dams which can fluctuate to produce a highly variable and frequently lower summer water temperature than the downstream environment (Olden and Naiman 2010). Krause *et al.* (2005) investigated many different flow regime scenarios with varying thermal attributes to determine whether a more suitable environment could be developed downstream by altering the current management operations.

The ultimate utility of these cold-water releases will depend on whether they are sufficient to maintain temperatures that will support the desired freshwater biota and also whether releases will be able to be made in priority areas of risk under climate change. In the following sections, we explore the role of temperature in structuring biotic assemblages downstream of a dam, and then outline case studies where cold-water releases are being considered to take advantage of the temperature effects to enhance downstream biotic assemblages.

4. TEMPERATURE TOLERANCE LIMITS OF FRESHWATER BIOTA

The upper temperature tolerance for southern Australian fish, reptiles, amphibians, invertebrates and plants are not well known. Furthermore, in many instances, the basis of the tolerance (e.g. maximum observed field temperature, LD_{50}) is not specified, so it is very difficult to ensure that comparisons among species are appropriate (Table 1). For some sensitive stream invertebrates, including mayflies, stoneflies and caddisflies, the upper tolerance limit is considered to occur at 21°C (Davies *et al.* 2004). In the present report we use this value as a starting point that infers maximum invertebrate diversity. This is then used to illustrate how such a threshold could be applied to assess the utility of a technique such as cold-water release. When future research is able to provide more detailed, region-specific temperature thresholds for sensitive aquatic communities, this should be applied instead. The threshold suggested by Davies *et al.* (2004) corresponds with literature available for cold-water fish in North America, where temperatures above 19 to 21°C have been shown to have a negative impact on salmonid behaviour, migration and survival (Krause *et al.* 2005).

Mitigation of cold-water releases in Australia may in fact favour invasive cold water species over Australian natives given that Australian native fish tend to be adapted to a warmer climate than fish introduced from colder climates in the Northern Hemisphere (e.g. salmonids) (Brown 2004). It might be possible for targeted cold-water releases to be used to control the distribution and abundance of exotic warm water fish species (e.g. carp), but the success of this technique is likely to vary. For example, some authors suggest that targeted cold water releases that are used to exceed lower temperature tolerances are unlikely to be successful, but controlled releases that target spawning and development of exotics may have potential (Rutherford *et al.* 2009). However, little empirical evidence appears to support that notion to date.

While increasing temperatures are likely to have a negative impact on all biota, past reviews associated with impacts of cold-water releases have focused on fish (e.g. Ryan *et al.* 2003, Rutherford *et al.* 2009), largely because there is more biological information for fish than other aquatic biota. Due to a lack of information on the thermal requirements of reptiles and mammals, they have not been covered in this report. Briefly, the predicted impacts associated with increasing stream temperatures have been recently described for platypus, *Ornithorhynchus anatinus* (Klamt *et al.* 2011). Although the upper thermal tolerance for the platypus is unknown, the optimal thermal range for the platypus is between 0.5 and 25°C (Smyth 2003).

The upper thermal tolerance (> 40°C) of the water rat, *Hydromys chrysogaster*, is unlikely to be reached in the near future, but the optimal range of 25°C to 30°C (Fanning and Dawson 1980) may be exceeded on a regular basis, and the impacts of this on reproductive output and fitness remain largely unknown. Recent thermal tolerance information for Australian freshwater turtles appears to be more common for optimal incubation temperatures (e.g. Eiby and Booth 2011, Micheli-Campbell *et al.* 2011) rather than adult tolerances.

4.1 Plants and Algae

Freshwater macrophytes generally have a thermal tolerance intermediate between that of terrestrial plants and marine plants (Santamaria and Van Vierssen 1997). The thermal optimum for many submerged freshwater macrophytes appears to be between 28 and 32°C (Barko and Smart 1981, Barko *et al.* 1982 however *c.f.* Tobiessen and Snow 1984). Inland aquatic habitats are generally well buffered against extreme fluctuations in temperature, but strong differences in thermal regime can be expected both among different geographical regions, and among seasons in temperate climates (Santamaria and Van Vierssen 1997). Habitat type can also affect thermal regimes. For example, floating plants and submerged plants that usually grow in shallow waterbodies are particularly exposed to contrasting temperature ranges during the course of the growth season compared to those found in deeper water (Santamaria and Van Vierssen 1997).

4.2 Invertebrates

Some of the upper thermal tolerances of freshwater invertebrates have been documented from Australian streams. Some of these tolerances are mentioned here, but see Cook *et al.* (2013) for more detail.

The upper thermal tolerances for a range of crustaceans have been tested. For example, upper thermal tolerances for Parastacidae (yabbies and crayfish) range from 28.6°C for *Cherax cainii* to 35.5°C for *Cherax destructor* (Bryant and Papas 2007). Members of the Cambaridae and Astacidae (Northern Hemisphere crayfish) have similar tolerances to the Parastacidae, with upper thermal tolerances of 27°C for *Camaroides japonicas* (Davies *et al.* 2006), 38.5°C for *Orconectes rusticus* (both Cambaridae, Claussen 1980) and 31.3°C for *Pacificastacus leniusculus* (Astacidae, Davies *et al.* 2006). The thermal tolerance of freshwater shrimp (Atyidae) is lower than that of the freshwater yabbies and crayfish, with an upper thermal tolerance of 25.7°C (Davies *et al.* 2006). Smaller crustaceans have a range of thermal tolerances, such as amphipods, which in Australia have tolerances between 14.6°C and 34.1°C (Quinn *et al.* 1994, Buchanan *et al.* 1988).

There is also a large range in the upper thermal tolerances for aquatic insects, with some dipterans (flies) ranging from 25.1°C (Silmuliidae blackflies, Gaufin and Herd 1971) and 32.4°C (Athericidae watersnipe flies, Davis *et al.* 2006). Ephemeroptera (mayflies), Plecoptera (stoneflies) and Trichoptera (caddisflies) are families known to be more sensitive to water quality than other families, and have upper thermal tolerances ranging from 11.7°C for *Hexagenia limbata* (Ephemeroptera, Gaufin and Hern 1971) to 31.4°C *Chimarra obscura* (Trichoptera, Davies *et al.* 2006). Some odonates (dragonflies and damselflies) with have upper thermal ranges from 32.5°C for *Boyeria vinosa* (Aeshnidae dragonfly) to 43.1°C for *Macromia illinoiensis* (Macromiidae dragonfly, Davies *et al.* 2006).

4.3 Fish

The upper thermal tolerances are known for many freshwater fish in southern Australia. Of the 20 native species listed in Table 1, the upper thermal tolerances range from 20°C for congolli (*Pseudaphritis urvillii)* to 44°C for spangled perch (*Leiopotherapon unicolor*). Known thermal tolerances for common invasive species span a similar range, with the lowest maximum temperature tolerance occurring for brown trout (*Salmo trutta*) at 22 to 26°C and the highest (up to 44°C) occurring for eastern gambusia (*Gambusia holbrooki*). However, there are numerous fish species that have a very wide temperature tolerance range (i.e. 2 to > 40°C, Rutherford *et al.* 2009).

Descriptions such as those provided in Table 1 have a tendency, however, to mask the complex effect that temperature can play on the various life-history stages of freshwater fish. Adults may have very different thermal tolerances to larval and egg stages, and these are addressed in the following sections.

Table 1. Known upper thermal tolerances of freshwater fish in southern Australia. # denotes tolerances from overseas studies.

Species	Common name	Upper thermal tolerance (°C)	Tolerance basis	Status	Reference
Natives					
Galaxias truttaceus	Spotted galaxias	< 20	Unspecified		Merrick and Schmida 1984
Pseudaphritis urvillii	Congolli	20	Unspecified		Bice 2010
Afurcagobius tamarensis	Tamar River goby	23	Unspecified	Restricted	Bice 2010
Aldrichetta forsteri	Yellow-eyed mullet	24	Expert opinion		Bice 2010
Gadopsis marmoratus	River blackfish	25	Unspecified	Threatened	Allen *et al.* 2003
Galaxias brevipinnis	Climbing galaxias	27 – 28$^{\#}$	Critical thermal maximum		Richardson *et al.* 1994
Craterocephalus fluviatilis	Murray hardyhead	28	Unspecified	Threatened	Bice 2010
Pseudogobius olorum	Swan River goby	28	Unspecified		Bice 2010
Melanotaenia fluviatilis	Murray-Darling rainbowfish	28	Unspecified	Uncommon	Ralph *et al.* 2010
Pseudomugil signifier	Pacific blue-eye	28	Unspecified		Allen *et al.* 2003
Retropinna semoni	Smelt	28	Unspecified		Ralph *et al.* 2010
Gadopsis bispinosus	Two-spined blackfish	28	Unspecified		Rutherford *et al.* 2009
Craterocephalus stercusmuscarum fulvus	Unspecked hardyhead	29 - 38	Unspecified	Threatened	Allen *et al.* 2003, McNeil and Hammer 2007
Geotria australis	Pouched lamprey	30	Unspecified	Rare	McNeil and Hammer 2007
Nannoperca obscura	Yarra pygmy perch	30	Unspecified	Restricted	Bice 2010
Galaxias olidus	Mountain galaxias	32	Unspecified		Cadwallader and Backhouse 1983
Mogurnda adspersa	Southern purple-spotted gudgeon	34	Unspecified	Threatened	Bice 2010
Galaxias maculatus	Common galaxias	35.4$^{\#}$	Critical thermal maximum		Richardson *et al.* 1994
Maccullochella peelii peelii	Murray cod	37	Unspecified		Ralph *et al.* 2010
Macquaria ambigua ambigua	Golden perch	37	Unspecified		Ralph *et al.* 2010
Bidyanus bidyanus	Silver perch	37	Unspecified	Vulnerable - Endangered	Ralph *et al.* 2010
Nematalosa erebi	Bony herring	38	Unspecified		McNeil and Hammer 2007

Species	**Common name**	**Upper thermal tolerance (°C)**	**Tolerance basis**	**Status**	**Reference**
Nannoperca australis	Southern pygmy perch	38	Unspecified	Threatened	Bice 2010
Tandanus tandanus	Freshwater catfish	38	Unspecified	Declining	Ralph *et al.* 2010
Leiopotherapon unicolor	Spangled perch	44	Unspecified		Ralph *et al.* 2010
Invasives					
Carassius auratus	Goldfish	42	Absolute thermal limits (avoidance response)		Hoyland *et al.* 1979
Cyprinus carpio	Common carp	40.6	Lethal temperature		Horosziewicz 1973
Salmo trutta	Brown trout	22-26	Upper critical range		Krause *et al.* 2005
Salvelinus fontinalis	Brook char; brook trout	$30^{\#}$	Upper incipient lethal temperature		McCullough 1999
Perca fluviatilis	Redfin perch	31	Upper lethal temperature		Lintermans 2007; Weatherley 1963
Oncorhynchus mykiss	Rainbow trout	31	Critical thermal maximum		Patra *et al.* 2007
Salmo salar	Atlantic salmon	$32^{\#}$	Upper incipient lethal temperature		McCullough 1999
Gambusia holbrooki	Eastern gambusia	38 - 44	Unspecified		Clarke *et al.* 2000; Swanson and Cech 1996; Bice 2010
Misgurnus anguillicaudatus	Oriental weatherloach	> 42	Recorded in field at 42°C		Lintermans 2007

4.3.1 Adults

Temperature can suppress reproduction across many freshwater fish species. In sub-Arctic species, the inhibitory effects typically appear at temperatures of between 10 and 11°C, among cold-temperate species at around 18°C, between 18 and 21°C for temperate species and above 24°C for warm temperate species (Pankhurst and Munday 2011). The threshold used in the USA as the maximum temperature for stockable trout waters is 21°C (DEQ 1997). Temperature has a large influence on metabolic reactions and physiological processes (e.g. oxygen consumption, Weatherley and Gill 1987, digestive and assimilation efficiencies, growth rate and reproduction, Reyjol *et al.* 2009) and so can act in varied and complex ways on individual fish and fish species. For example, increasing water temperatures by 1°C also has the potential to modify the life-history strategy of a range of species, by accelerating the development of gonads and affecting the timing of spawning (Gillet and Quétin 2006).

4.3.2 Eggs

Eggs are one of the most thermally-sensitive life stages in fishes. Tolerance limits appear to be within ±6°C of the spawning temperature for many species (Rombough 1997). Embryonic development is significantly affected by temperature changes and, for many species, the rate of embryonic development more than triples for each 10°C increase in temperature (Rombough 1997). This increased rate of development results in reduced incubation periods which may advance hatching, particularly in cold-water species with long incubation periods (Rombough 1997). Shorter incubation periods may influence the fitness of the species of interest and are likely to be dependent on whether environmental conditions for larval survival are suitable at the time of hatching (Pankhurst and Munday 2011).

Small increases in temperature can also dramatically increase egg mortality, but gametogenesis is highly temperature sensitive in many fish species and breeding may cease before critical thermal limits for egg survival are reached (Pankhurst and Munday 2011). In contrast, other species of fish will spawn at suboptimal temperatures and thus may suffer reduced survivorship due to increased temperatures both reducing gamete viability (Van Der Kraak and Pankhurst 1997) and increasing mortality during embryogenesis (Pankhurst and Thomas 1998).

4.3.3 Larvae

The metabolism, growth and development of larval fish can all be affected by temperature (Blaxter 1992). The metabolic rate of larval fishes varies amongst species, but generally increases sharply with increased temperature (Rombough 1997). Higher metabolic rates result in higher demands of basal energy sources at higher temperatures (Pankhurst and Munday 2011). Larval fish also experience increased growth rates with increased temperatures (Rombough 1997). The growth rates of larval fish have shown to be linearly related to increased temperatures until the upper thermal limit is reached. This is unlike other life stages which usually decline before the lethal thermal limit is reached (Pankhurst and Munday 2011). Increased larval development rates may result in reduced durations of each life stage (Pankhurst and Munday 2011). In marine fishes, the larval phase usually has a high mortality rate so faster developmental and reduced duration of larval stages may be advantageous for some fish species (O'Connor *et al.* 2007).

It has been suggested that fish species are able to acclimate and adapt to increasing water temperatures over short periods of time.

The thermal exposure history of fish species can influence the response to increasing temperatures, with a varied thermal exposure resulting in higher tolerances and thus a greater capacity to cope with thermal stress for many species (Pankhurst and Munday 2011). The thermal history of the parent fish can also be important for embryonic and larval life stages, with an increased thermal history of parents increasing the thermal tolerance of juveniles (Rombough 1997).

4.3.4 Relationships between in-stream temperatures and key aquatic organisms

In addition to upper thermal tolerances, whole thermal regimes can be of importance to fish life-history and success. This means that, as well as considering average temperatures through time, we can also look at seasonal requirements or temperatures on even shorter time-scales. Olden and Naiman (2010) provided an example of how stream thermal regimes could be plotted and described as a series of metrics describing that regime, similar to those that are also used to describe flow regimes. These metrics can then be related to ecological communities (e.g. individual species or functional groups) by overlaying corresponding life-history stages.

For native and exotic fish in Australia, there are several technical reports and texts that provide critical temperature requirements (e.g. Koehn and O’Connor 1990; McDowall 1996; Drew 2008; Rutherford *et al.* 2009). The information from these references is summarised in Appendix A. Most of this information is provided in summary tables rather than using the type of pictorial representation presented by Olden and Naiman (2010) to link life-cycle information to a typical thermal regime from the streams and rivers that the fish inhabit. However, we have used the method outlined by Olden and Naiman (2010) to provide examples for two Australian native fish, Murray cod (*Maccullochella peelii peelii*) and river blackfish (*Gadopsis marmoratus*). These two fish species were chosen over invertebrates, because data for fish in Australia are more readily available than for insects and fish are often considered to be more charismatic or iconic than stream invertebrates.

In order to apply the method, data were required for temperatures in a specific river or stream. For the Murray cod example (Figure 1), we used thermal regimes for locations above and below the Hume Dam, which are situated on the main stem of the Murray River, and are within the known distribution of the species. These data were sourced from the Victorian Water Data Warehouse (www.vicwaterdata.net). These locations provide an example of how critical life-history stages of Murray cod correspond with a thermal regime that is directly influenced by the cold-water releases below the Hume Dam compared with one upstream. For the Murray cod example, there was some differentiation in the thermal profiles upstream and downstream of Lake Hume. Downstream waters tended to be warmer in winter and spring and cooler in summer and autumn than waters upstream of the dam.

The coolest temperatures both upstream and downstream were within the range of minimum temperatures able to be tolerated by Murray cod. However, summer temperatures upstream of the dam exceeded the range of temperatures for upstream migration, spawning and incubation on several occasions during the breeding season, and also exceed the maximum threshold for optimal growth for the best part of a month, although the temperature remained within the ‘preferred’ temperatures described by Ryan *et al.* (2003). The time period used for this analysis was in a drought period, but the differences in temperature above and below the dam illustrate the possible impact of such a storage, and the utility of this approach to overlaying a species’ known thermal tolerances on stream temperature data to assess thermal suitability.

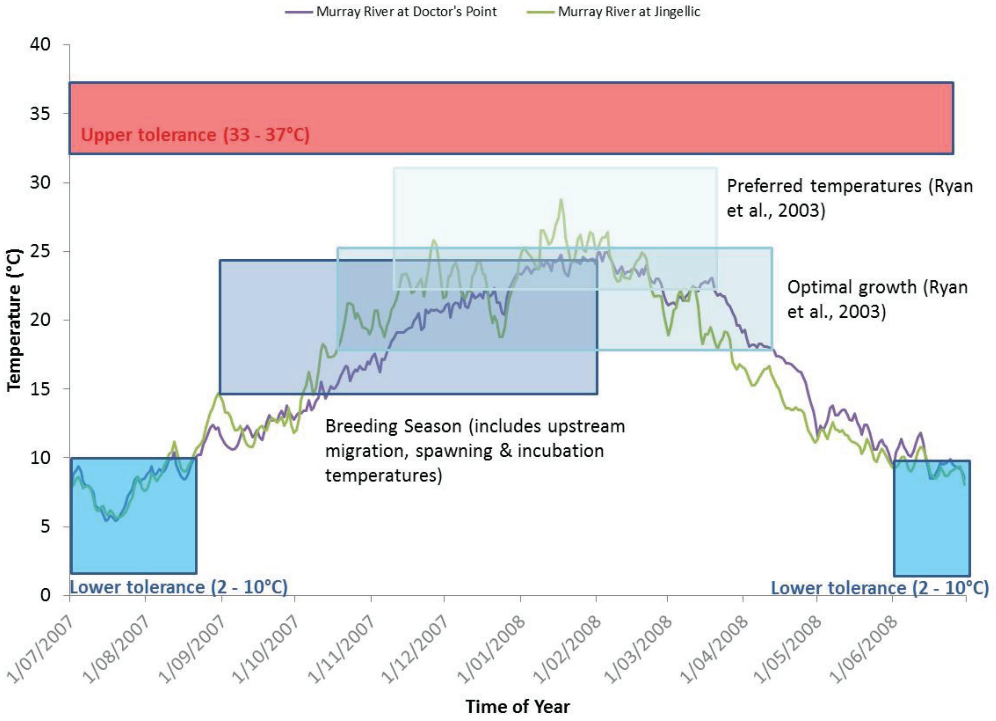

Figure 1. Biological information and thermal regimes for Murray Cod (*Maccullochella peelii peelii*) overlying two thermal regimes that represent temperature profiles above (green line) and below (purple line) Lake Hume Dam.

For river blackfish (Figure 2) we have used the thermal regime in the unregulated Lardners Creek as a comparison. The thermal regime is well within the upper and lower temperature limits for this species for the entire year shown, including the optimal temperatures for spawning and incubation during the breeding season. However, biological information for river blackfish (*Gadopsis marmoratus*) is limited, particularly temperature preference and optimal temperatures for growth.

Therefore, much of the species-specific ranges included in Olden and Naiman (2010) remains undefined for river blackfish, and so therefore not represented on Figure 2. Similar gaps in preferred temperature ranges and tolerances also exist for other Australian native fish (see Appendix A).

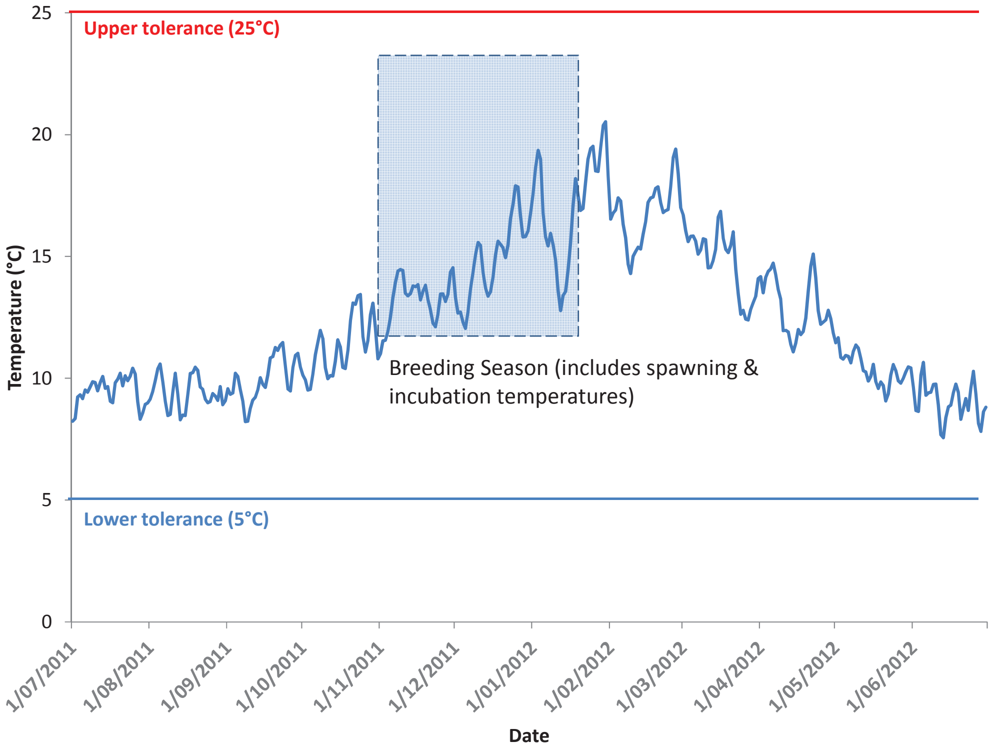

Figure 2. Biological information and typical thermal regime for river blackfish (*Gadopsis marmoratus*) overlying the thermal regimes for Lardners Creek, which is an unregulated tributary on the Gellibrand River, south-west Victoria.

It is evident from these two examples, which represent two of the better-studied native fish, that many of the thermal requirements for different life-history stages remain unknown. For example, there is limited information relating to the effect of falling spring temperatures on Murray cod. Where available, more detailed information regarding the range of optimal temperatures for feeding and growth could be included on these diagrams, which would increase their utility for managers concerned with environmental flow releases and concomitant changes in-stream temperatures below the release point (including both the timing as well as the range of temperatures).

Most of the information outlining thermal optimal for key life-history stages tends to overlap, rather than forming discrete compartments spread out along the thermal regime (which is more similar to the examples given in Olden and Naiman 2010). In some respects, this makes it easier to manage flows, as similar flows will be suitable for more than one life-history stage (i.e. similar flows will maintain a critical range of temperatures that facilitate spawning, egg development and juvenile rearing) rather than needing to manage flows at several different stages during the year. Interestingly, these examples highlight that the upper tolerances for native fish are far from being exceeded in these two systems which is significant, as it suggests that native fish in many systems may not be regularly experiencing temperatures near their tolerances, but other sub-lethal impacts may still be occurring (e.g. the duration of temperatures for optimal growth are potentially shortened upstream).

We are unaware of any Australian study that has tried to correlate a suite of quantitative metrics (e.g. frequency of low thermal events) such as those described by Olden and Naiman (2010), but this may be a worthy goal for future research to enhance our understanding of the possible future effects of climate change and the efficacy of mitigation strategies such as cold-water releases. Diagrams of this nature are likely to be useful for species with adequate biological information, as a tool for managers to assess alternative flow regimes. In particular, they may assist in identifying critical aspects of the thermal regime for a particular system which are in danger of exceeding (or falling below) thermal tolerances for species of interest. However, in the short term, without more-detailed information on the thermal tolerances of the various life-history stages, they may be of limited use for the majority of Australian species.

4.4 Ecosystem Processes

Our knowledge of the role of temperature on ecosystem processes is extremely limited. In most cases, thresholds and other relationships with temperature are unknown. Metabolism is an exception, in that we know that the metabolic reactions of most organisms (other than those with specialised adaptations to live at hotter temperatures, such as perthermaphiles) occur between 0 and 40°C. Metabolic reactions cease near 0°C due to the phase transition associated with freezing water and above approximately 40°C, metabolic rates are reduced by the increasing influence of catabolism (Gillooly *et al.* 2001). Beyond this, additional research into the role of thermal effects on ecosystem processes is urgently required.

5. CASE STUDIES USING COLD-WATER RELEASES TO PROTECT FRESHWATER FISH

In the following section, we briefly describe and summarise three case studies that have assessed whether cold-water releases could be managed to benefit in-stream fauna in reaches downstream of the dam wall. It should be noted that the manipulation of water releases from different off-take levels has also been considered for ameliorating increasing water temperatures inside large water storages (Saito and Koski 2006), but we have focussed on river reaches downstream of water storages only. As indicated previously, the negative impacts of thermal pollution on rivers has been well documented in the past (e.g. Ryan *et al.* 2001 and Olden and Naiman 2010), so we have specifically targeted case studies investigating the use of cold-water releases to improve at least one aspect of freshwater ecosystems. The three case studies presented span work that occurred in a single Australian and two overseas rivers. Few case studies of this nature exist, and those that do are based on mathematical models associated with how varying flow regimes influence in-stream temperature, rather than direct assessments of how biota respond to manipulations of hypolimnetic releases.

Furthermore, all three examples focus on managing temperature for cold-water fish species, predominantly introduced salmonids rather than warm-water native Australian species. The purpose in presenting these case studies was to summarise the knowledge base and to illustrate methods for evaluating possible benefits for cold-water releases, rather than to focus on the individual tolerances highlighted by each. We contend that the principles illustrated by each case study should be relevant to warm-water species if future ambient in-stream temperatures begin to reach or exceed the upper tolerance limits of warm-water species residing in river reaches below large water storages, noting that those tolerances are likely to be higher than the values mentioned here.

5.1 Potential use of planned cold-water releases to support a tailwater trout fishery in the Smith River, USA

Krause *et al.* (2005) assessed the thermal regime downstream of Philpott Dam, Virginia, USA, to determine whether flow management strategies could be used to benefit an important brown trout (*Salmo trutta*) fishery within a 24-km reach. A dynamic flow and stream temperature model was used to evaluate alternative flow scenarios that would be required to: 1) increase the occurrence of optimal growth temperatures (12 – 19°C); 2) reduce the occurrence and magnitude of hourly temperature fluctuations; and 3) reduce the frequency and duration of periods that exceeded the optimal temperature (21°C) for brown trout growth and survival. A total of 16 flow scenarios were assessed, including variations in the timing (e.g. morning versus late afternoon) and duration of releases.

The authors suggested that there was no single manipulation of the flow regime that provided optimal improvement for all three criteria. For example, the upper optimum temperature was exceeded 1.3% of time during normal release operations, but this was reduced to less than 0.1% by ceasing all non-release periods and by increasing baseflow and the duration of releases during the morning rather than evening. In contrast, the maximum increase in the number of ideal temperature days was achieved by releasing in the morning, decreasing the duration of releases and by not increasing baseflow. A combination of continual release rather than intermittent releases, together with a one-hour release in the morning provided the best improvement for all three criteria.

The study by Krause *et al.* (2005) demonstrated that it may be possible to manipulate temperature without the need for expensive retrofitting of dams to include multiple off-take infrastructure that takes water from a range of heights within the dam. However, it should also be noted that the model is based on hydro-peaking flows, so the model parameters tested are likely to vary more widely for the majority of dams on mainland Australia because hydroelectricity production is uncommon other than the Snowy River scheme. These models may be directly useful for some of the Tasmanian storages, but would require further investigation to assess their applicability for releases from non-hydropower dams.

5.2 Modelling potential effects of dam removal and retrofitting hypolimnetic off-takes on steelhead recruitment in the Manistee River, USA

Horne *et al.* (2004) presented an example where in-stream temperatures directly below two hydroelectric dams were warmer during summer, rather than cooler, as is usual, due to the release of warm surface waters from each dam to produce hydroelectricity. This increase in temperature was thought to impede the recruitment of steelhead (or rainbow trout, *Oncorhynchus mykiss*), which form part of a highly successful recreational fishery that was established after their introduction to the Michigan area in the late 1800s. These elevated temperatures were thought to increase the susceptibility of young-of-year fish to mortality during the summer period, leading to reduced recruitment of this species.

There were four specific objectives of this study, which all involved the use of mathematical models and statistical tests to develop overall predictions for how in-stream water temperatures would vary given two dam alteration scenarios: dam removal and retrofitting with a bottom-withdrawal mechanism to provide greater control on the temperature of water released. The outcomes of these scenarios were then used to predict improvements in steelhead recruitment. The study focused on a 3-km reach below Tippy Dam, which was targeted because most of the steelhead recruitment occurs within this particular section of stream. Two different models, *SNTEMP* and *CE-THERM-R1,* were used to test the dam removal and off-take retrofitting scenarios.

The models suggested that both dam alteration scenarios would improve steelhead recruitment through reduced in-stream temperatures over the summer period. Recruitment rates were predicted to increase by between 59 and 129%, but were still low compared to other nearby streams and tributaries. The study also concluded that hypolimnetic releases demonstrated the greatest potential to reduce in-stream temperatures, particularly during very warm summers. However, the authors also suggested that further estimates for the dam removal model were required because steelhead recruitment in upstream river reaches was possible following dam removal, and this was not considered in the model used for this study.

5.3 Planned mitigation of cold-water releases and the potential impacts on a recreational brown trout fishery - an Australian example

Brown (2004) presented a combination of mathematic models that predicted the potential impacts of increasing water temperatures on growth and mortality of brown trout (*Salmo trutta*) in the Goulburn River downstream of Lake Eildon, Australia.

This example consisted of a situation where a dam was targeted for mitigation measures to ameliorate the impacts of cold-water releases on Australian native fish. The study presented a potentially adverse outcome for recreational fishers because these mitigation measures are likely to be detrimental to the trout fishery.

Brown (2004) used a combination of published models to estimate growth and mortality of brown trout. This first model consisted of an adapted bioenergetics-based growth model which incorporated foraging of drift-feeding brown trout in the Maruia River in New Zealand (Hayes *et al.* 2000). This bioenergetics model was used to predict the effects of raising average daily water temperature by 2 and 4°C on brown trout growth, which were within the range of predicted increases associated with the proposed mitigation measures. A second model was then used to estimate mortality, which was based on an equation constructed from the relationship between the growth parameters and natural mortality that was developed from multiple fish stocks and species by Pauly (1980). The combination of these two models then enabled predictions of growth and mortality for brown trout by incorporating environmental water temperature and biological data (i.e. size-at-maturity, length-at-age data and length-mass relationships) taken directly from the Goulburn River.

The environmental water temperatures used in the modeling consisted of average daily water temperatures recorded over three years, from October 1999 to September 2002. Temperatures during that period ranged from 7 to 11°C during the coldest parts of the year (in August) and 14 to 21°C during summer (in March). A substantially lower average daily water temperature was experienced during the summer of 2001/02 than occurred for either of the two preceding years due to releases of cold irrigation water when Lake Eildon was at approximately 50% storage capacity in late November, keeping water temperatures below 17°C for most of the summer period (approximately 4°C lower than other years). This provided temperatures that were closer to optimal for brown trout growth (i.e. 13°C) than the two preceding summers.

Modeled results showed that raising the average daily temperature by 2°C could reduce the potential maximum size of brown trout by 4–38% and an increase of 4°C could reduce growth by 30–43%. This would result in an increased time taken to reach 300 g (the reference size of typical quality trout harvested by anglers) by up to 31% and 144%, at increases of 2 and 4°C, respectively. The cooler summer temperatures during the 2000/01 year resulted in yearlings reaching approximately twice the predicted mass for yearlings in other years. The natural mortality model showed that increases in water temperature resulted in increased mortality of brown trout, due to changes in fish size, growth rate and water temperature. These results must be considered cautiously though, given the confidence intervals surrounding the estimates for mortality were large.

While the study by Brown (2004) did not specifically focus on the use of cold-water releases for the benefit of in-stream biota, these sorts of models could be useful for future scenarios of increasing temperatures where selective cold-water releases could be used to provide optimum temperatures for growth and survival of both native and non-native species. One potential limitation of this study is that a pre-existing model specifically developed for brown trout was required, which was based on drift feeding by hatchery-reared trout based in New Zealand (Hayes *et al.* 2000). Therefore, it is debatable whether these parameters would also be relevant to wild trout populations in Australia and similar models for use on Australian natives are unlikely to exist at present. Providing that similar models to Hayes *et al.* (2000) are available or can be readily developed, other data requirements for the scenarios provided by Brown (2004) should be more accessible, which include in-stream temperature data and basic biological information such as size at maturity, length-at-age and length-mass relationships.

6. THERMAL REGIMES OF SOUTHERN AUSTRALIA: EXAMPLES USING VICTORIAN DAMS

To illustrate the potential for tolerance limits to be exceeded for sensitive taxa in Victoria, we compiled temperature data from the Victorian Water Data Warehouse for five water storages in Victoria. Cases were chosen where sufficient water temperature data were available downstream of a dam and either above the dam or on a comparable adjacent stream. The five example water storages consist of three high priority dams and a single medium and low priority dam as outlined in the report by Ryan *et al.* (2001). Ryan *et al.* (2001) considered priority dams as those that were discharging into natural streams and had no surface releases. Farm dams were excluded, and maximum priority dams were assigned to those that had releases from greater than 10 m below full supply level (FSL) and had regular releases. Medium priority dams were categorised by having releases from 6 to 10 m below FSL or by having only occasional releases. The minimum priority dams were those that had either destratification or releases from less than 6 m or shallow regulating pondages. No data were available regarding the operation of dams and the timing and pattern of releases, so differences among dams may exist.

The upper temperature tolerance limit of 21°C used in this section is based on the upper tolerance limit for sensitive macroinvertebrate taxa in Australia and New Zealand, particularly Ephemeroptera (mayflies), Plecoptera (stoneflies) and Trichoptera (caddisflies) (see Cook *et al.* 2013; Quinn *et al.* 1994), and is used as an example of how the method may be able to be applied for a known threshold. Upper thermal tolerances of temperature ranges between 19-21°C also appear to be commonly used as upper temperature tolerances or as the upper end of an optimal temperature range in a number of studies (e.g. Guillemette *et al.* 2011) most of which focus on salmonids. We acknowledge that some sensitive taxa will be either less or more tolerant of this upper temperature, depending on region and potential for acclimation (e.g. leptocerid caddisflies in the Grampians National Park in Victoria survive in pools with mean in-stream summer temperatures above 23°C, Matthews *et al.* unpublished data). Species- and regional-specific tolerances should be used in any serious consideration of the impact of cold-water releases in a given catchment. Where appropriate thresholds are not available, we advise extreme caution in applying this approach, as a 21°C thermal threshold may be inappropriate due to regional differences of thermal tolerances in aquatic biota.

Olden and Naiman (2010) stated that most assessments (if not all) of the impacts of cold-water releases on downstream reaches are based on visual inspection of thermal regimes rather than statistics. Therefore, we provide an example of a simple statistical test that could be used to compare thermal regimes from upstream, or adjacent tributaries, and downstream of the five example reservoirs in Victoria. These were developed by processing daily stream temperature data in the River Assessment Package (Marsh 2004) and were tested for significant differences in the annual maximum, annual minimum and average temperature upstream, compared to downstream or adjacent tributary, temperatures using PERMANOVA (Anderson *et al.* 2008) in PRIMER v6. with the PERMANOVA+ add-on. The analysis used a two-factor design including location relative to the dam (i.e. upstream or downstream) as a fixed factor and dam identity as a random factor. Sites on adjacent tributaries were classified as ‘upstream’ for the purposes of this analysis where other suitable sites were not available. The annual maximum, minimum and average temperatures were analysed in separate univariate tests.

6.1 *High Priority Dams in Victoria*

6.1.1 *Dartmouth Dam*

The Dartmouth Dam is situated on the Mitta Mitta River and has two outlet structures, situated at 62 m and 122 m below FSL (Koehn *et al.* 1995). Releases of up to 15000 ML are possible (Ryan *et al.* 2001). Daily temperature data are available for a site 50 km upstream of the dam (ID: 401203; *n* = 2857) and at 7 km (ID: 401211; *n* = 3017) and 70 km downstream of the dam wall (ID: 401204; *n* = 2942) downstream between 2003 and 2011 (Figure 3). Stream temperatures exceeded the example threshold of 21°C for 17% of the time at 50 km upstream of Dartmouth Dam during an eight year period between September 2003 and September 2011 (Figure 3). During the same time period, the number of days that exceeded 21°C was greatly reduced (2% of the time) at 7 km downstream of the dam. The influence of cold-water releases had dissipated at 70 km downstream to the point where stream temperatures exceeded 21°C for 20% of the time. So in this system, temperatures exceeding 21°C may occur naturally, but the influence of the dam on water temperatures can clearly be seen, and the likelihood of crossing a regionally-relevant threshold could be assessed using this method.

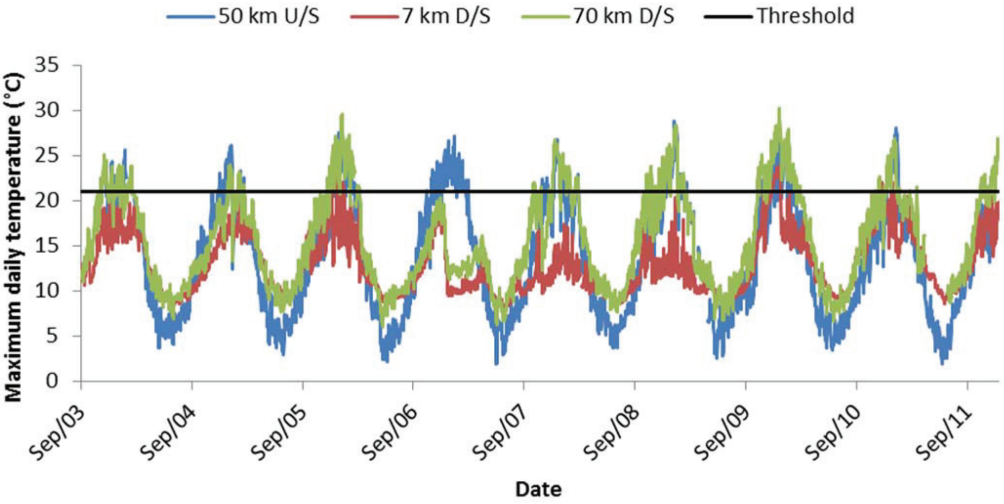

Figure 3. Thermal regime showing the number of days when in-stream temperatures exceeded 21°C at stations 50km upstream, and at 7 km and 70 km downstream of Dartmouth Dam. The Dartmouth Dam is situated on the Mitta Mitta River, Victoria

6.1.2 *Malmsbury Reservoir*

Malmsbury Reservoir is one of three dams situated along a 10-km stretch of the Coliban River. The Upper Coliban and Lauriston reservoirs are situated above the Malmsbury Reservoir, but no thermal data were available for either reservoir. Daily thermal data is available at 50 km downstream of the Malmsbury Reservoir (ID: 406215; *n* = 1630) and at an unregulated site on the Campaspe River, adjacent to the Coliban River (ID: 406213; *n* = 1106) for a four-year period between 2007 and 2011 (Figure 4). During that four-year period, temperatures exceeded 21°C for 26% and 18% of the time at locations adjacent and downstream of Malmsbury Reservoir, respectively (Figure 4).

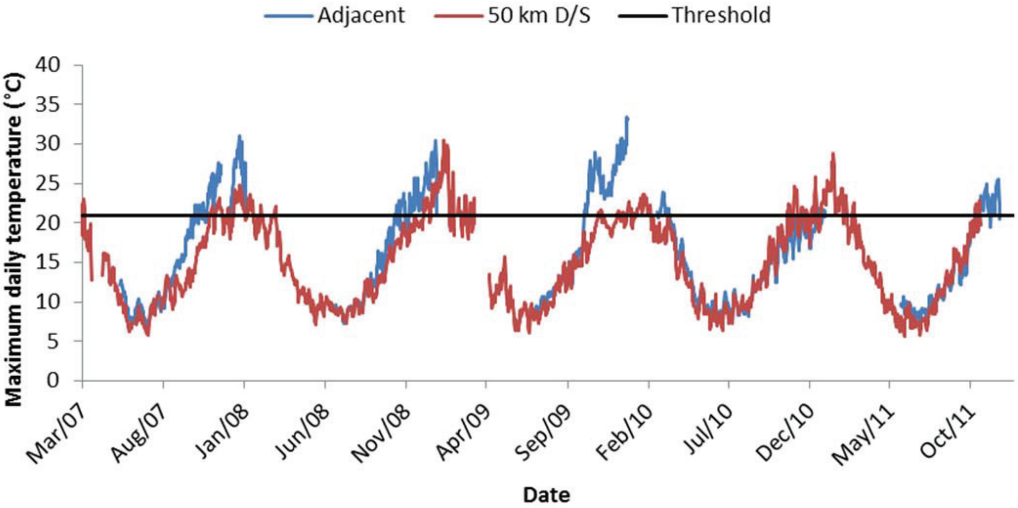

Figure 4. Thermal regime showing the number of days when in-stream temperatures exceeded 21°C at a location situated 50 km downstream of Malmsbury reservoir and at an adjacent unregulated location on the Campaspe River

6.1.3 Tullaroop, Cairn Curran and Laanecoorie Reservoirs

The upper section of the Loddon River has three major reservoirs including Cairn Curran, Tullaroop and Laanecoorie, and together these three dams were categorised as a high priority scenario (Ryan *et al.* 2001). Daily temperature data in the Loddon River is limited to one site upstream of all reservoirs (ID: 407222; *n* = 318), downstream of the Laanecoorie Reservoir (ID: 407203; *n* = 569) and much further downstream (ID: 407229 *n* =865) from October 2008 – October 2011 (Figure 5). Stream temperatures exceeded 21°C for 34% and 28% of the time at the upstream location and directly downstream of Laanecoorie Reservoir, respectively. At the site further downstream, stream temperatures exceeded 21°C for a greater period of time (38%) than the two upstream locations, although long stretches of missing data from each gauge may have influenced these values.

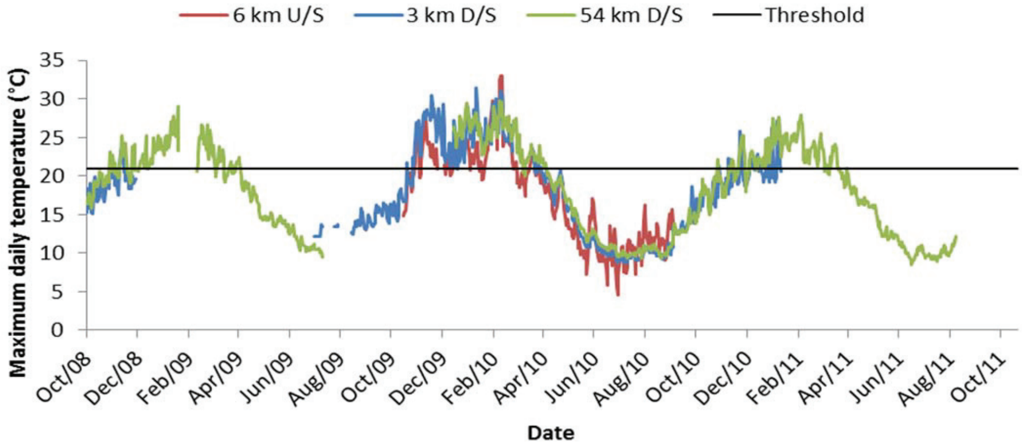

Figure 5. Thermal regime showing the number of days when in-stream temperatures exceeded 21°C at stations associated with Tullaroop, Cairn Curran and Laanecoorie reservoir system on the Loddon River, Victoria

6.2 Lower Priority Dams in Victoria

6.2.1 Lal Lal Reservoir

Lal Lal Reservoir is categorised as a medium priority dam on the Moorabool River. It has a dam wall of 48 m high and a capacity of 59,549 ML, with releases made from a multi-level tower with a greatest depth of 8 m below FSL (Ryan *et al.* 2001). There are two sites with daily temperature data available close to the Lal Lal Reservoir; one 4 km upstream of Lal Lal Reservoir (ID: 232210; *n* = 734) and one situated 20 km downstream (ID: 232204; *n* = 596). Data for the downstream site are only available between December 2009 and December 2011, and thus comparisons have been made over this two-year period only. Upstream of the reservoir, temperatures exceeded the 21°C threshold for 10% of the time (Figure 6) and 12% at 20 km downstream. These data suggest that cold-water releases between December 2009 and 2011 have dissipated before 7 km downstream of the wall in most instances (e.g. see the beginning of the sequence in 2009 for an exception).

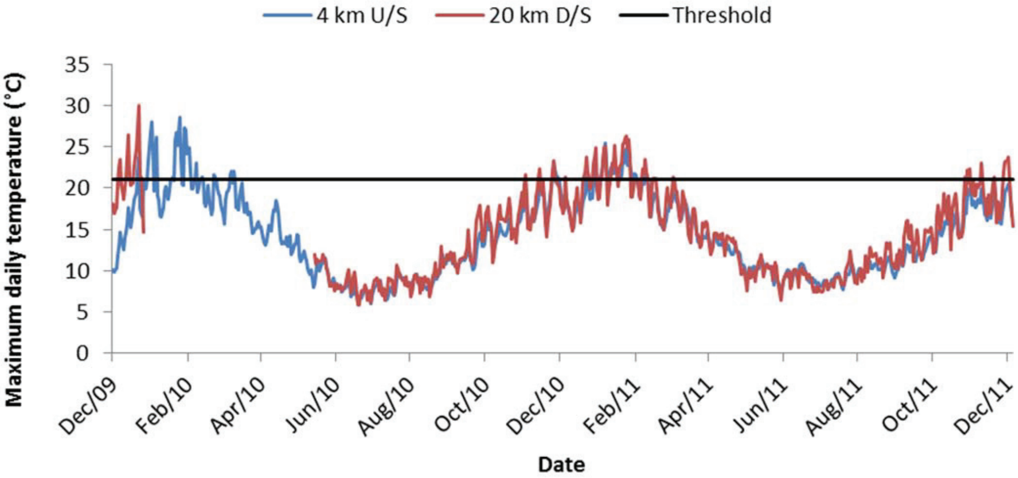

Figure 6. Thermal regime showing the number of days when in-stream temperatures exceeded 21°C at stations 4 km upstream and at 20 km downstream of Lal Lal reservoir situated on the Moorabool River, Victoria

6.2.2 Thomson Dam

Thomson Dam is situated on Thomson River and has a capacity of 1,068 GL. Daily water temperature records were available 1 km upstream (ID: 225114, *n* = 161) and 5 km downstream of Thomson Dam (ID: 225210; *n* = 161) from December 1994 to June 2009. Stream temperatures exceeded 21°C on only two occasions at the downstream location during that period (Figure 7). The results illustrated in Figure 7 are in contrast to observations made by Gippel *et al.* (1992) where lower stream temperatures were apparent 5 km downstream of the dam.

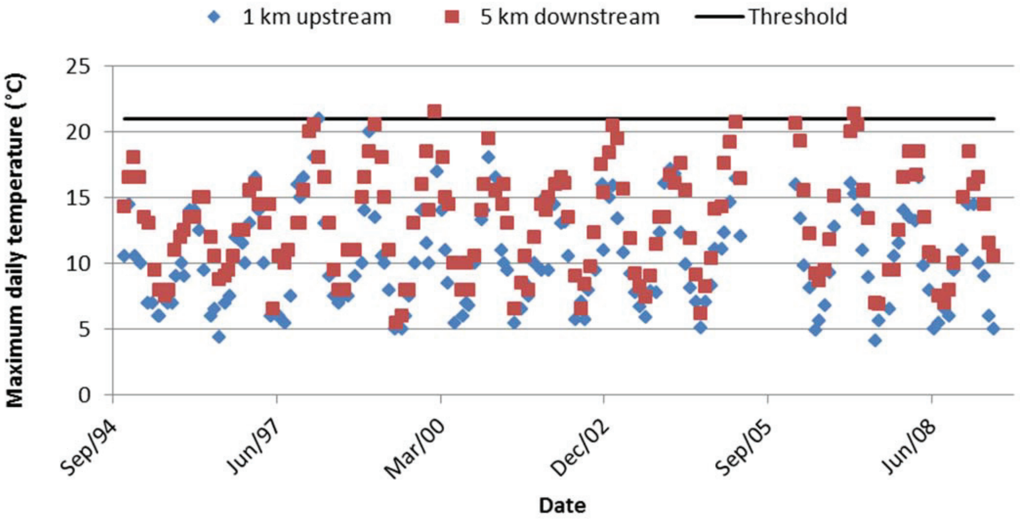

Figure 7. Thermal regime showing the number of days when in-stream temperatures exceeded 21°C at stations 1 km upstream and at 5 km downstream of Thomson Dam situated on the Thomson River, Victoria

6.3 Summary of findings

When data from four of the five examples presented here were analysed (excluding Thompson Dam), we detected significant differences in the maximum water temperatures upstream and downstream of dams (pseudo-$F_{1,5}$ = 24.0, P = 0.025). This analysis incorporated both sites immediately downstream and further downstream of dams together. Maximum water temperatures were significantly and consistently lower downstream of dams compared with upstream, with an average across all rivers of 29.0°C downstream, compared with 31.3°C upstream. No significant differences were detected in the minimum or mean water temperatures upstream and downstream of these dams.

Differences in mean maximum monthly water temperatures generally appeared to be greater between the upstream and downstream sites over the summer period (Figure 8). This trend was more apparent for the Dartmouth system between the upstream site and 7 km downstream of Dartmouth Dam, but stream temperatures appear to have returned to previous temperatures 70 km downstream of the dam (Figure 8a).

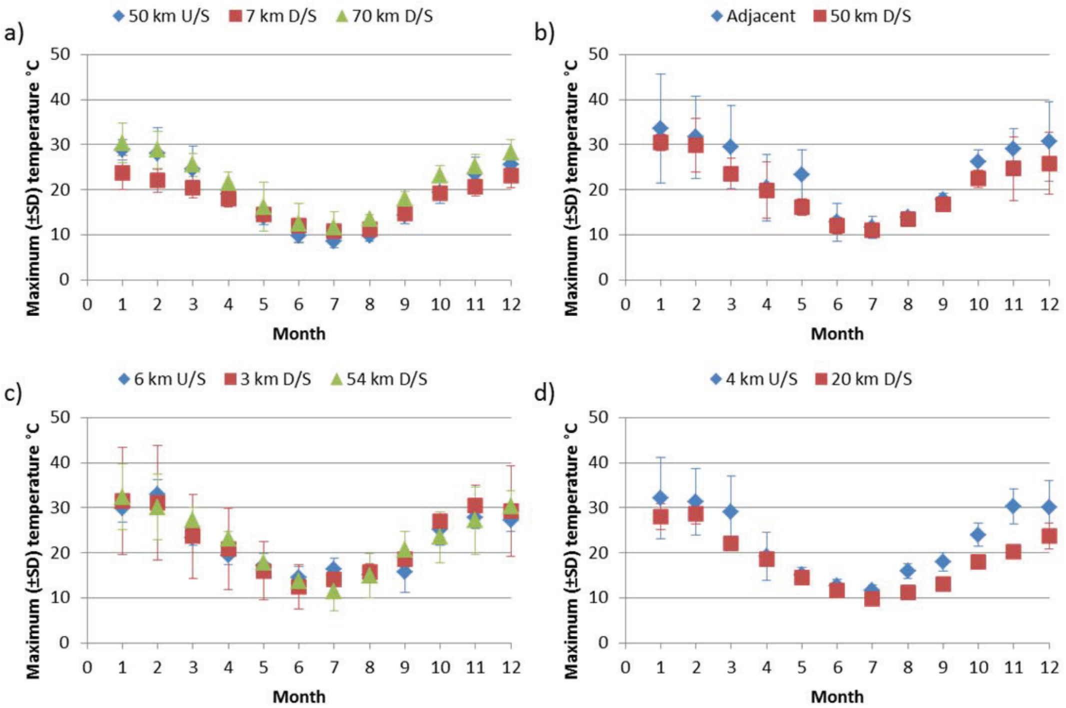

Figure 8. Mean (±SD) maximum monthly water temperature for a) Dartmouth Dam, b) Malmsbury Reservoir, c) Cairn Curran, Tullaroop and Laaencoorie Reservoirs and d) Lal Lal Reservoir, over various time frames (see text above). Month 1 is January, and so on

Variability in water temperatures is another important aspect to consider when assessing stream thermal regimes. The annual variability of water temperature for the sites downstream of the dams was generally lower than that of upstream or adjacent sites (Figure 9). For Dartmouth Dam and Malmsbury Reservoir, the thermal variation in upstream sites was noticeable higher in the first few years that data were available, but this variation has reduced to a level similar to that of the downstream and adjacent sites in later years (Figures 9a, b). The variation observed in the Cairn Curran, Tullaroop and Laanecoorie system was variable in itself, but few data were available for the upstream site (Figure 9c).

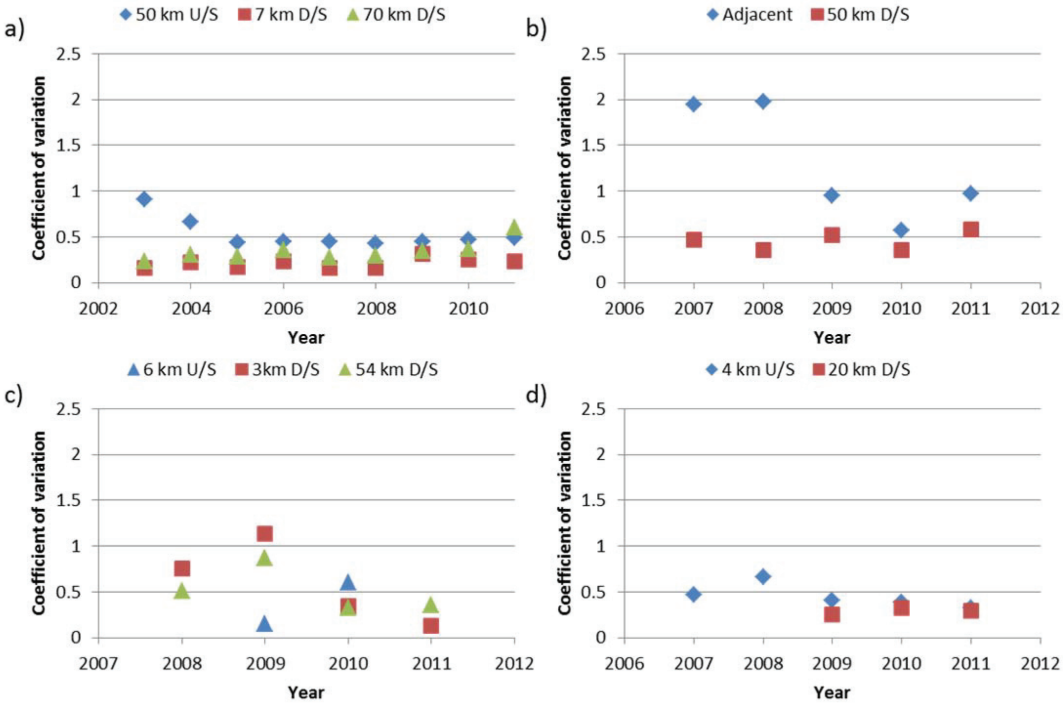

Figure 9. Annual coefficient of variation in temperature over time periods were data was available for a) Dartmouth Dam, b) Malmsbury Reservoir, c) Cairn Curran, Tullaroop and Laaencoorie Reservoirs and d) Lal Lal Reservoir

The three high priority dams showed a reduction of stream temperatures below the dam and a reduction in the number of days where the maximum temperature exceeded 21°C. This was particularly evident at Dartmouth Dam, where stream temperatures downstream of the impoundment were reduced below the 21°C threshold 98% of the time compared to 24% above the dam. This reduction in stream temperatures may be due to the infrastructure in place at Dartmouth Dam (off-take structures at 62 m and 122 m below FSL) and the volumes of water that are released by the dam (up to 15000 ML).

The medium and low priority examples, Lal Lal Reservoir and Thomson Dam, respectively, both showed no obvious difference in stream temperatures during the periods that water temperature data was available. This supports the conclusions drawn by Ryan *et al.* (2001) on the varying impact on stream temperatures caused by dams, based on release and de-stratification strategies. These two impoundments may not have the capacity for cold-water releases.

The three high priority reservoirs illustrate that cold-water releases from southern Australian reservoirs already have the capability of reducing the frequency of the upper temperature tolerances of sensitive macroinvertebrate taxa and cold-water fish such as salmonids during summer period. Whether attempting to use targeted cold-water releases to achieve such a reduction is appropriate or not will vary among river systems and the specific taxa within each. However, these temperature data also illustrate the poor spatial resolution of temperature data for Victorian reservoirs, which we also suspect is the case for other dams and reservoirs in southern Australia. The extent to which these cold-water releases penetrate downstream of the dam wall is difficult to ascertain given the paucity of temperature data for within regulated systems and also on adjacent unregulated rivers. Increasing the spatial resolution of in-stream temperatures should be a high priority for natural resource management agencies on both regulated and unregulated rivers.

7. CAPACITY FOR HYPOLIMNETIC RELEASES IN SOUTHERN AUSTRALIA

The downstream penetration of cold-water releases is influenced by a number of factors. These include the timing and volume of releases, the distribution and size of impoundments, the number of tributaries entering the system downstream of the dam, catchment and stream geomorphology, channel form of the downstream reaches and air temperature (Crisp 1987). Riparian vegetation cover and stream orientation directly downstream of the dam wall are also likely to influence penetration (Davies 2010). Cold-water releases from large dams in south eastern Australia cause the greatest impacts on stream temperature, however, larger dams are more expensive to construct, which can leave less money for the additional infrastructure required to regulate the temperature of dam releases (McMahon and Finlayson 1995) enabling cold-water releases to be managed. Furthermore, many dams were constructed before the impact of thermal pollution was recognised, so may not currently have the capacity to shandy water from across the water column within the dam.

Research in the USA suggests that any impounded water body with a dam wall of over 5 m in depth can undergo thermal stratification, except those reservoirs where average yearly inflow volume exceeds the reservoir volume by a factor of 10 or more (Harlemann 1982) but similar information is unavailable for Australia. However, it is known that thermal signatures associated with releases have been detected up to 200 km downstream of some large water storages in Australia (Ryan *et al.* 2003), illustrating the potential for controlled cold-water releases from some dams at least. It is likely that other large storages have similar potential.

The number of reservoirs that release hypolimnetic waters in southern Australia has been reviewed for Victoria (Ryan *et al.* 2001, 2003), the Murray Darling Basin (Ryan and Preece 2003); NSW (Lugg 1999, Preece 2004) and Queensland (Brennan 2002). Five major changes associated with cold-water releases have been identified: reduced seasonal variability in temperatures; reduced daily variability; the steepness of rise in spring temperatures tend to flatten out with cold-water release; delays in summer peak temperatures, sometime by months rather than weeks (which has the capacity to affect timing of important life history stages); and sudden and dramatic drops in temperature following large releases, particularly at times when natural flows would be low or much warmer (e.g. particularly with releases from hydroelectric dams; Ryan *et al.* 2003). We were unable to identify any reviews for Tasmanian dams, but a mass mortality event of fish was observed in the Pieman River, which was attributed to supersaturation of oxygen below the Pieman dam following a release of water (Carolyn Maxwell and Julie Mondon, pers. comm., March 2013).

Table 2. Some examples of past studies that have looked at impacts of hypolimnetic releases in southern Australia. See also Ryan *et al.* (2003)

Reference	Target species/process	Impact variable and notes
Lugg 1999	NA	Issues associated with cold-water pollution in NSW
Ryan *et al.* 2001	NA	Potential for cold-water pollution in Victoria
Ryan and Preece 2003	NA	Potential for cold-water pollution in Murray Darling Basin. Incorporates parts of Vic, SA, NSW and QLD
Rutherford *et al.* 2009	NA	Potential for cold-water pollution in ACT, specifically Bendora Reservoir, Cotter River
Ryan *et al.* 2003	Murray cod, golden perch, silver perch, freshwater catfish, crimson-spotted rainbow fish, rainbow trout	Laboratory trials testing preferences, growth, swimming behaviour and egg development in response to varying temperatures
Brennan 2002	NA	Potential for cold-water pollution in Queensland
Preece 2003	NA	Potential for cold-water pollution in NSW
Phillips *et al.* 2001	NA	Workshop consisting of a series of papers investigating potential impacts of cold-water releases and potential mitigation measures
Sherman *et al.* 2007	CWR mitigation measures	Hume Dam
Preece and Jones 2002	NA	Temperature regime below Keepit Dam, NSW
Acaba *et al.* 2000	NA	Assessed historical NSW data
Todd *et al.* 2005	Murray Cod	Mitta Mitta River

8. CURRENT MANAGEMENT AWARENESS OF HYPOLIMNETIC RELEASES

To assess the feasibility of implementing cold-water releases as a future management strategy to ameliorate the effects of climate change, 12 catchment managers from Victoria, Western Australia, Tasmania and South Australia were contacted. Each contact was asked about their organisation’s current management strategy regarding hypolimnetic releases and whether cold-water releases were being considered as a future management strategy, to complement existing irrigation, water supply and environmental flows strategies.

To supplement the responses of the catchment managers that were contacted, seasonal watering plans published by the Victorian Environmental Water Holder (VEWH) for each catchment in Victoria were also evaluated. The VEWH was developed to achieve better environmental outcomes for a variety of water availability scenarios. This goal was to be achieved by improved cooperation between catchment management authorities and Melbourne Water. Any references to cold-water releases that were listed from the VEWH management plans for each catchment are summarised below.

8.1 What are the current attitudes to/awareness of hypolimnetic releases?

Management of cold-water releases was common for many of the catchment managers who were contacted. Most natural resource management (NRM) agencies were aware of the potential negative issues associated with cold-water releases and some had implemented strategies to reduce those effects on streams. Managers were often familiar with some of the common deleterious effects associated with unplanned cold-water releases, such as reduced variation in in-stream thermal regimes and the potential for reduced oxygen concentrations downstream of the impoundment.

Management of hypolimnetic releases was rarely mentioned or considered in the VEWH reports. Of the 14 VEWH reports, four made mention of cold-water releases, which were for the Tarago, Yarra, Werribee and Goulburn systems. For three of these reports (Tarago, Werribee and Goulburn), the reference to cold-water releases focused on negative impacts to lotic ecosystems that are associated with the provision of hypolimnetic releases for environmental flow allocations. In the Tarago Catchment, the release of cold water into the Tarago and Bunyip Rivers was incorporated in the risk contingency plan, with a medium threat and likelihood of occurrence of reduced temperatures occurring in the catchment stated (Victorian Environmental Water Holder 2011a).

To prevent the identified consequences of such releases (i.e. potential stress, or loss of biota and public concern/outcry of these releases) monitoring and preventative adaptive management were suggested actions. For the Yarra River, cold-water releases were recognised and temperature was a consideration in the timing of summer and winter freshes (Victorian Environmental Water Holder 2011b). Managers from this system had a more positive approach to releasing flows to manage downstream water temperatures (see below). Thus, only one NRM agency was actively incorporating temperature as a potential positive aspect of their environmental flow release strategy, rather than treating it solely as a risk.

8.2 Is anyone considering the technique?

None of the catchment managers contacted were considering controlled cold-water releases as a future management strategy to combat rising stream temperatures. Cold-water releases were predominately viewed as a potential threat to the health of reaches downstream of major dams and the thought of using the cooler water to benefit the thermal regime downstream of the impoundment was new to many of the managers. Despite this, some could see the potential for using cold-water releases as a management strategy for ameliorating increasing stream temperatures, but several expressed concern about the likely difficultly surrounding implementation of the strategy and how these releases could be managed in an effective manner. For example, there was concern expressed regarding who would provide continual and ongoing monitoring of stream temperatures downstream from the release point, together with assessment of the longitudinal changes associated with increasing distances downstream. Several managers suggested that consideration of this strategy would be made if successful implementation elsewhere could be demonstrated.

Managers of the Yarra River catchment have been considering temperature as part of their environmental flow releases (Victorian Environmental Water Holder 2011b). During dry summer periods, priority reaches were planned to be monitored for declining dissolved oxygen and increasing water temperatures. If temperatures are found to approach 25°C, summer freshes have been planned to be released from banked volume to prevent downstream temperatures from rising above 25^{o}C. Additional summer releases had also been considered to protect spawning of Australian grayling (*Prototroctes maraena*), by releasing waters in April that were within the range of 9 to 11°C, which are ideal for grayling spawning (Koster and Dawson 2010). During winter months, winter releases have been considered when temperatures are >17°C (Victorian Environmental Water Holder 2011b). Within the Yarra River system, several impoundments are present, but only two have the capacity for hypolimnetic releases (Upper Yarra Dam and Maroondah Reservoir; Ryan *et al.* 2001). As yet, none of these procedures have been implemented (Anna Lucas, Melbourne Water, pers. comm., December 2012).

9. POTENTIAL SCENARIOS FOR MANAGEMENT

1. To mimic natural thermal regimes

Using cold-water releases as a management strategy to alleviate the potential increases in stream temperature due to climate change would be most successful if the natural thermal regime could be preserved. The most common negative effects documented by cold-water releases are a result of depressed summer temperatures, elevated winter temperatures, seasonality displacement by either time or duration and loss of diel variation (NSW Office of Water 2011). In NSW, temperatures of lowland rivers exhibit temperatures of 5 to 10°C in winter and 20 to 30°C in summer and the natural diel variation can be up to 5°C per day (NSW Office of Water 2011).

In the summer months, dams are often highly stratified with warmer water sitting upon deeper, colder waters. During these times, water can be selectively withdrawn from an appropriate depth (providing the appropriate infrastructure exists) to best match the natural thermal regime expected in the river or by releasing water from multiple off-takes, effectively 'shandying' water from different levels to achieve the desired temperature. However, releasing water from multiple off-takes is likely to be complex and will require appropriate modelling and direct trials to be undertaken before this method could be used. It may be necessary to incorporate multiple off-takes in situations where dissolved oxygen concentrations within bottom waters of the dam are lower than those at the surface. In the winter months, dams are not usually stratified, but the water stored is typically warmer than those of upstream reaches or in similar nearby rivers, so the selective withdrawal of water during winter is unlikely to be a viable option.

The NSW Office of Water (2011) has suggested that the natural diel variation downstream of high priority dams is currently unachievable because of the timeframes required to change infrastructure settings that can deliver water from varying depths and temperatures. Managing the thermal regime on a weekly basis is possible, but the management costs of such constant changes to infrastructure settings would be high. The natural temperature increase in lowland rivers is about 1°C per week, on average during autumn and spring (most pronounced temperature increase of the year; NSW Office of Water) meaning that large changes to the mix of depths from which water is drawn would be needed regularly to match that rise. Matching the natural thermal regime on a monthly basis (i.e. rather than weekly or more frequently) may be a workable and reasonable compromise between ecological outcomes and manageability (NSW Office of Water 2011), assuming this can be done to complement other environmental (and water supply) objectives.

2. To reduce frequency and duration of high thermal temperature events

Short-range weather forecasts could be used for the preparation and planning of targeted cold-water releases to prevent the frequency and duration of extreme high temperature events. This technique would require continual monitoring of water temperatures both above and below a dam wall. Planned environmental flow releases to prevent summer stream temperatures exceeding 25°C have already been considered for the Yarra River in Victoria (see above), but this technique is yet to be directly implemented (Anna Lucas, Melbourne Water, pers. comm., December 2012).
Currently, this technique appears to be most feasible option because it should require less water than that needed to mimic natural thermal regimes, and that water could be specifically targeted over short period of time.

We are unaware of any Australian studies that have modelled how the quantity and timing of cold-water releases might influence the extent and duration of temperatures downstream of large water storages, similar to those described by two of the three case studies (Horne *et al.* 2004; Krause *et al.* 2005), but this information would be highly valuable to assist in planning.

Currently, however, the volume of water required to prevent extremely high downstream water temperatures is unknown. There is also a possibility that such a volume of water may result in higher flows than those typically experienced in targeted streams for the time of year.

3. To facilitate movement and migration of fish between thermal refuges (or tributaries) during critical life-cycle stages

Carefully-planned cold-water releases might be considered for facilitating migration patterns of fish by re-establishing suitable thermal connectivity between naturally occurring thermal refuges or to allow migration of fish between tributaries that provide necessary habitat to complete crucial stages of their life cycle (e.g. spawning habitat), assuming there are no physical barriers to migration. Cold-water releases could be used to facilitate fish migrations that are cued by temperature.

While this scenario focuses on the facilitation of fish migration and habitat provision, it is possible that other components of the ecosystem (e.g. sensitive macroinvertebrates) would also benefit from the releases if their temperature tolerances were also exceeded. This scenario is based on descriptions provided by two of the three case studies (Horne *et al.* 2004; Krause *et al.* 2005) that discuss the potential of using this technique to facilitate the persistence of cold-water fish (i.e. salmonids) in the Northern Hemisphere. However, there are several significant knowledge gaps that prevent current use of this technique in Australia, such as whether there is a similar issue for native species.

10. CONCLUSIONS AND RECOMMENDATIONS FOR AUSTRALIAN STREAMS

The case studies outlined above are all species-specific and aim to maintain ideal temperature ranges for a specific target species. While this may provide optimal thermal conditions for growth and serve to minimize mortalities, the manipulation of thermal regimes to suit a cold-water target species in Australia is likely to have an impact on the food resources for these species (i.e. in-stream macroinvertebrate assemblages). This has been acknowledged in the case studies (Krause *et al.* 2005). Also, there might be an effect on the dynamic relationship between temperature and other ecosystem processes and function (e.g. Magnuson *et al.* 1979). Therefore, for hypolimnetic releases to be effective, preliminary trials should include more than monitoring of a single target species. Ignoring other ecosystem links, such as macroinvertebrate assemblages, could result in other indirect effects by altering the abundance and supply of food at critical times during the life-cycle of species.

It is apparent that cold-water releases from some of the larger dams in Australia are capable of reducing in-stream temperature by as much as 8-12°C (Inland River Network News 2000) and for quite some distance downstream (e.g. up to 400 km for the Macquarie River below Burrendong Dam; Walker 1979). Given these figures, it appears that carefully managed cold-water releases may in some cases be viable for ameliorating predicted increases in stream temperatures associated with climate change.

However, we emphasise the ‘carefully-managed releases’ because a multitude of parameters will need to be measured before, during and after any planned releases. For example, dissolved oxygen concentrations, depth and position the cold-water lens relative to the off-take and the extent of mixing between the upper and lower stratified layers while water is being released, will need to be measured within the hypolimnetic waters above the dam wall prior to any release .Continual monitoring of conditions downstream of the dam will also be required during the release to ensure that flows and temperatures are being maintained within the target bounds, particularly for the first applications of this method. Releases will need to be managed to avoid temperature shock, low dissolved oxygen levels and sudden increases in flow particularly in naturally low-flow seasons, so gradual releases may also be required to provide organisms with time to adapt to these changes. Gradual releases as a management strategy for environmental flows are recognised by the VEWH (e.g. VEWH 2012).

The persistence of some cold-water fish species in warm-water streams may be dependent on the presence of thermal refuges, particularly during summer. Use of thermal refuges by fish has been documented by numerous studies, particularly for salmonids in the Northern Hemisphere (e.g. Berman and Quinn 1991; Ebersole *et al.* 2003a) but only one example was found from the Southern Hemisphere (Olsen and Young 2009). Cold-water thermal refuges can occur at sites of cooler, groundwater recharge (Matthews and Berg 1997; Baird and Krueger 2003), mixing of cooler subsurface and warmer surface layers (Bilby 1984; Ozaki 1988; Burkholder *et al.* 2008; Acuna and Tochner 2009), thermally-stratified pools (Matthews *et al.* 1994; Nielsen *et al.* 1994; Matthews and Berg 1997; Tate *et al.* 2007) or by cooler waters entering from tributaries (Kaeding 1996; Baird and Krueger 2003; Goniea *et al.* 2006; Sutton *et al.* 2007; Keefer *et al.* 2009; Sutton and Soto 2012; Petty *et al.* 2012) and side channels (Stephens and Dupont, 2011).

The size and frequency of these thermal refuges have been associated with increased abundances of salmonids in some streams of the USA, but it is rare for the distribution of thermal refuges to be mapped (Ebersole *et al.* 2003a). For southern Australian streams, we could not find any examples associated with the description of thermal refuges or their use by in-stream biota. Since a lot of Australian native species are warm climate adapted, their usage of thermal refuges is likely to be different from cold water adapted salmonids in the Northern Hemisphere, but this remains a knowledge gap. Thermal refuge is potentially available for a range of plants and animals that can move to higher altitudes, but this is limited for fish in Australia due to the largely flat topography (Koehn 2011) and multiple, natural barriers that may be present (e.g. waterfalls).

Few studies have described use of thermal refuges by non-salmonid species (e.g. leopard darter *Percina pantherina*, Schaefer *et al.* 2003; striped bass, Jackson *et al.* 2002). However, while we do not advocate the position, the conservation of salmonids is still likely to be relevant and a priority for southern Australian streams given the high socio-economic value of trout fisheries in Victoria, NSW and Tasmania, despite the fact that the species are introduced. For example, recreational anglers in Victoria spend an estimated $170 million a year pursuing trout, redfin and other native species (Department of Primary Industries 2011), estimates for Tasmania are around $40 million per annum (Inland fisheries Service 2008) and $70 million per annum are associated with recreational trout fishing in the Snowy River region of NSW (Dominion Consulting 2001). It should be noted that some of the thermal refuges described above occur in very large river systems (e.g. stream width > 50m, Columbia and Snake Rivers, Keefer *et al.* 2009; Tagliamento River, Acuna and Tochner 2009) so it is possible that thermal refuges such as those described above do not exist in Australia because the major regulated rivers are much smaller than their Northern Hemisphere counterparts.

Given that current cold-water releases already appear to be reducing in-stream temperatures by up to 8-12°C and can penetrate hundreds of kilometres downstream, it appears that such releases could potentially help ameliorate increasing stream temperatures associated with climate change, assuming that the target streams were downstream of a suitable dam. The case studies also suggest that altering the timing and duration of releases could also reduce in-stream off-takes without the need to alter off-take infrastructure. However, retrofitting of multi-level off-takes is likely to provide greater control (i.e. 'shandying') of release water temperatures under a range of flow regimes. This will come at a high infrastructure cost. Several engineering options exist for mitigating cold-water pollution, but they are all costly (Sherman 2000). For example, a minimal cost option (i.e. draft tube-mixer) for Burrendong Reservoir was estimated at $2.2 million and $40,000 per annum to operate, compared with $25 million to retrofit multiple off-take valves (Inland Rivers Network 2000). Given this, we have outlined what we believe are some of the important considerations required before considering the use of cold-water releases as a future management option.

Step 1. Fine-scale temperature data to identify cold-water thermal refuges in Australian streams appears to be effectively non-existent. Therefore, mapping thermal refuges in reaches (e.g. kilometres) downstream of dam walls appears to be a logical first step. These reaches are where cold-water releases have the potential to have an impact, so identifying any current thermal refuges will provide managers with an understanding of the resources that may need to be protected in the future under climate change. Identifying reaches that contain groundwater recharge areas or inflow will be important, because they may be providing fish with important thermal refuge areas (Gaffield *et al.* 2005) and could be ameliorating climate-related increases in ambient stream temperatures.

A similar process of identifying cool-water refuges attributable to riparian vegetation cover has also been investigated to facilitate future management of brown trout in parts of the UK (Broadmeadow *et al.* 2011). Conserving these areas, by actions such as protecting riparian vegetation to provide maximum shade in reaches with in-stream thermal refuges, should be a high priority. This step would include a survey of incoming tributaries, incoming springs and profiles along the length of the main stem to identify intrusion of cold groundwater and potential mixing zones. Once identified, low-maintenance temperature loggers could be installed at these locations and within the main stem to determine the extent and stability of these refuges over space and time. Good quality temperature loggers that are now relatively easy to install can now be purchased without excessive costs (approx. $50-60 each) and have greatly facilitated the development of research associated with understanding temperature trends in lotic ecosystems (Webb *et al.* 2008).

The installation of loggers becomes more difficult within increasing stream size, depth and river discharge, but should be suitable for most streams and river in temperate Australia, and if placed securely, can be left in the stream to record temperature for long periods of time (i.e. at least 12 months). Since loggers are now more affordable for NRM agencies and water authorities, direct recording of in-stream temperature profiles should be a high priority, irrespective of whether they occur downstream of potential cold-water release sites given the dire warnings associated with climate change. A similar recommendation regarding the installation of temperature loggers was made by Ryan *et al.* (2001).

Step 2. Development of dynamic flow and temperature models for high priority dams such as those described by Krause *et al.* (2005) is needed to predict the effect of varying flow regimes on in-stream temperatures at varying distances downstream of the dam. Ideally, direct manipulations of cold-water flow releases would be used to fine-tune these models, particularly at smaller scales that might be relevant to the scales associated with thermal refuge patches. As part of their planning for the use of environmental water, the VEWH is now considering research proposals that seek to use small volumes of environmental water (Victorian Environmental Water Holder, 2012), so small-scale, direct manipulations of flow may be possible for some major water storages. The VEWH also recognises that emergency management actions may be necessary under certain circumstances to minimize the impacts of disturbance created by algal blooms and blackwater events. This may involve using a portion of water allocated for environmental flows to help ameliorate the impacts of extreme natural events (Victorian Environmental Water Holder, 2012). The use of short-term weather forecasts to predict extreme high in-stream temperature days may also be a potential method by which emergency management could be used to plan the timing of targeted cold-water releases to reduce mass mortalities of in-stream biota.

Step 3. If natural thermal refuges are present and are used by native or introduced fish in Australia, carefully-timed cold-water releases might be considered for facilitating migration patterns. This would work through the re-establishment of suitable thermal connectivity between naturally-occurring patches of thermal refuge or by allowing the migration of fish between tributaries that provide necessary habitat to complete crucial stages of their life cycle (e.g. spawning habitat). Cold-water releases could be used to facilitate fish migrations that are cued by temperatures.

Information regarding important migration cues is another significant knowledge gap for many Australian freshwater fishes (but see recent work which has suggested a role for temperature in grayling migrations; Koster and Dawson 2010) and some species in the

Northern Hemisphere can exhibit large variation between individuals with respect to the influence of temperature and commencement of migration patterns (Howell *et al.* 2010). However, some of the techniques required to track the use of thermal refuges by fish can be expensive and difficult to apply (e.g. fish tagging and tracking).

While this scenario focuses on the facilitation of fish migration and habitat provision, it is possible that other components of the ecosystem (e.g. sensitive macroinvertebrates) may also benefit from the releases provided that a source of colonists is available. Basic life-history information and the relationship between physical and chemical parameters remain as major knowledge limitations for predicting changes to faunal assemblages in freshwater ecosystems.

Step 4. Ongoing research is needed in to the temperature tolerances of Australian native biota to assess species and region-specific thresholds to ensure that we have a basic understanding of the importance of dynamic temperature regimes to aquatic biota and freshwater ecosystem processes. Temperature tolerance data should be developed in the field, to enable us to understand the environmental limits of a species' distribution, in addition to laboratory-based methods such as LD_{50}. These data will provide us with the ability to anticipate potential impacts higher stream temperatures associated with climate change on freshwater ecosystems.

In conclusion, we agree with the recommendations made by Ryan *et al.* (2001) that rigorous monitoring of in-stream temperature and biological communities is urgently needed to provide a greater understanding of the impacts of cold-water releases in southern Australia. Ecological monitoring frameworks are continually being developed (based on Victoria Environmental Flows Monitoring and Assessment [VEFMAP]) for many systems to assess ecological changes associated with environmental flow releases (Victorian Environmental Water Holder, 2012). However, in the absence of funding to sample macroinvertebrates and fish, temperature loggers should be installed at multiple sites above and below maximum and medium research priority dams (Ryan *et al.* 2001) together with adjacent, non-regulated streams as a starting point. This will increase our understanding of the potential impacts of cold-water releases and changes to in-stream temperatures associated with human induced climate change.

The use of cold-water releases to ameliorate increases in stream temperatures associated with human-induced climate change does show potential in theory, but we have found little direct evidence that cold-water releases are currently a viable management option for ameliorating projected increases in in-stream temperatures. This technique will only be viable when sufficient water is available as part of any environment flow allocation, and so it may not be an option at all during periods of drought. In addition, water storage levels will also play important role even if sufficient environmental water is available. For example, water volume will influence whether stratification will occur over summer and water drawdown below the thermocline in storages with low water levels can result in a sudden change from cold-water to warm-water releases, which has been observed for some storages during the 'Millennium' drought in Victoria (SKM 2005). Therefore, at this point in time, there are many uncertainties and knowledge gaps which represent limitations in our ability to effectively apply the strategy. Also, any planned cold-water releases will need to meet legislative requirements, which may differ among states and could present a further constraint on this technique being implemented in the future. Until we understand the possible impacts on a wide range of biota (i.e. not just cold-water fish) and the direct and indirect effects of future climate on in-stream temperatures, we do not recommend to use cold-water releases to ameliorate high stream temperatures and extreme caution must be used if implementing this strategy. However, it is possible that this technique may become more viable in the future as climate change progresses, if knowledge gaps can be addressed and if new, cheaper techniques for delivering water are developed.

APPENDIX 1

Table A1. Known thermal tolerances of freshwater fish in southern Australia. * denotes tolerances from overseas studies

Scientific name	Common name	Status	Lower temp (°C) threshold	Upper temp (°C) threshold	Optimal range (°C)	Spawning time	Spawning temperature (°C)	Early life stage thresholds (°C)	Distribution in southern Australia
Native									
Ambassis agassizii	Agassiz's glassfish								SA, NSW
Anguilla australis	Short-finned eel					Nov - May$_{10}$			VIC, SA, NSW, TAS
Anguilla reinhardtii	Long-finned eel					Spawn annually in ocean			TAS, VIC, NSW
Craterocephalus fluviatilis	Murray hardyhead	Threatened	10$_{8}$	28$_{8}$		Sept - Apr$_{1}$			NSW, VIC, SA
Craterocephalus stercusmuscarum fulvus	Unspecked hardyhead	Threatened	9.3$_{2}$	36$_{2}$,29$_{10}$, 38$_{14}$		Oct - Feb$_{1}$	>24$_{1}$		SA, NSW
Atherinosoma microstoma	Small-mouth hardyhead	Restricted				Aug - Dec in the MDB, up to 9 months in VIC$_{1}$			SA, VIC, TAS

Scientific name	Common name	Status	Lower temp (°C) threshold	Upper temp (°C) threshold	Optimal range (°C)	Spawning time	Spawning temperature (°C)	Early life stage thresholds (°C)	Distribution in southern Australia
Leptatherina wallacei	Western hardyhead					Sept - Feb$_{10}$			WA
Pseudaphritis urvillii	Congolli; tupong			20$_{8}$		Mar - Aug?$_{1}$			TAS, VIC, SA, NSW
Nematalosa erebi	Bony herring		9$_{8}$	38$_{14}$		Oct - Feb$_{2}$	> 28$_{14}$		Widespread & abundant
Nematalosa vlaminghi	Perth herring								WA
Potamalosa richmondia	Freshwater herring					Jul - Aug			NSW, VIC
Mogurnda adspersa	Southern purple-spotted gudgeon	Threatened	19$_{8}$	34$_{8}$		Nov - Feb$_{1}$	>20$_{1}$		NSW
Gobiomorphus australis	Striped gudgeon					Feb - May$_{10}$	21$_{10}$		NSW, VIC
Gobiomorphus coxii	Cox's gudgeon								VIC, NSW
Hypseleotris klunzingeri	Western carp gudgeon					Feb - May$_{10}$	>22.5$_{10}$		SA, VIC, NSW
Hypseleotris spp.	Carp gudgeon					Oct – Apr$_{2}$	>22.5$_{1, 17}$		SA, VIC, NSW, ACT

Scientific name	Common name	Status	Lower temp (°C) threshold	Upper temp (°C) threshold	Optimal range (°C)	Spawning time	Spawning temperature (°C)	Early life stage thresholds (°C)	Distribution in southern Australia
Philypnodon grandiceps	Flat-headed gudgeon					Oct – Apr$_{1}$	18-27$_{1}$		SA, NSW, VIC, TAS
Philypnodon macrostomus	Dwarf flat-headed gudgeon	Uncommon				Oct - Apr$_{1}$	19-22 in aquaria$_{1}$; inhabits areas 15 - 30$_{2}$		VIC, NSW, SA
Gadopsis marmoratus	River blackfish	Threatened	5$_{10}$	25$_{10}$		Oct - Jan$_{1}$	>16$_{1}$	hatch in 14 days at 15$_{1}$	VIC, NSW, SA
Gadopsis bispinosus	Two-spined blackfish	Threatened	1.7$_{28}$	28$_{28}$		Oct - Dec$_{10}$	>18$_{28}$	13.5-23.5$_{28}$	VIC, ACT, NSW, TAS, SA
Galaxias brevipinnis	Climbing galaxias	Translocated		27 – 28*$_{33}$		Mar - Aug $_{1}$	12$_{29}$		SA, VIC, NSW, TAS
Galaxias fuscus	Barred galaxias	Threatened				Aug - Sept$_{1}$	2-3$_{1}$		VIC
Galaxias maculatus	Common galaxias		<10$_{32}$	35.4*$_{33}$	~20$_{14}$	Mar - May$_{1,10}$		Eggs 4.4 – 17*$_{34}$	WA, SA, VIC, NSW, TAS
Galaxias occidentalis	Western minnow					Following winter rains$_{10}$			WA

Scientific name	Common name	Status	Lower temp (°C) threshold	Upper temp (°C) threshold	Optimal range (°C)	Spawning time	Spawning temperature (°C)	Early life stage thresholds (°C)	Distribution in southern Australia
Galaxias olidus	Mountain galaxias	Declining	1.7[15]; 5.5[29]	32[30]		Sept – Nov, can occur Mar - May[1]	13 - 15[31]		NSW, VIC, SA
Galaxias rostratus	Flat-headed galaxias	Declining				Aug - Sept[1]	9 - 14[10]		NSW, VIC, SA
Galaxias truttaceus	Spotted galaxias	Translocated or remnant?		<20[15]		Mar - Aug[1]	11[29]	Eggs <12[29]	VIC, TAS, WA
Galaxiella munda	Mud minnow					Jul - Oct[10]			WA
Galaxiella nigrostriata	Black-stripe minnow					Jul - Oct[10]			WA
Galaxiella pusilla	Eastern little galaxias					Aug - Oct[10]			SA, VIC, TAS
Lovettia sealii	Tasmanian whitebait					Aug - Sept[10]			TAS
Neochanna cleaveri	Tasmanian mudfish					Jul - Aug			VIC, TAS
Geotria australis	Pouched lamprey	Rare		30[14]		Jul[1]			VIC, SA, TAS, WA

Scientific name	Common name	Status	Lower temp (°C) threshold	Upper temp (°C) threshold	Optimal range (°C)	Spawning time	Spawning temperature (°C)	Early life stage thresholds (°C)	Distribution in southern Australia
Afurcagobius tamarensis	Tamar River goby	Restricted		23[8]		Sept - Nov[1]			VIC, NSW, SA,TAS
Pseudogobius olorum	Swan River goby			28[8]		Sept - Nov[10]			WA, SA
Afurcagobius suppositus	South-western goby								WA
Pseudogobius spp.	Blue-spot goby					Sept - Nov[8]			VIC, TAS, NSW
Redigobius macrostomus	Largemouth goby								VIC, NSW, TAS
Tasmanogobius lordi	Tasmanian goby								
Nannoperca obscura	Yarra pygmy perch	Restricted	10[8]	30[8]		Sept - Oct[10]	16-24°C 1		SA, VIC
Nannoperca australis	Southern pygmy perch	Threatened	3[8]	38[8]		Sept - Jan[1]	>16°C 1		SA, NSW, VIC, TAS
Edelia vittata	Western pygmy perch					Jul - Nov[10]		Hatching occurs in 60-72 hours at 16-22°C [11]	WA

Scientific name	Common name	Status	Lower temp (°C) threshold	Upper temp (°C) threshold	Optimal range (°C)	Spawning time	Spawning temperature (°C)	Early life stage thresholds (°C)	Distribution in southern Australia
Nannatherina balstoni	Balston's pygmy perch					Winter$_{10}$			WA
Lepidogalaxias salamangroides	Salamanderfish					May - Aug$_{10}$			WA
Melanotaenia fluviatilis	Murray -Darling rainbowfish	Uncommon	10_{10} 18_{2}	28_{2}		Nov – Feb$_{2}$	>20_{1}		NSW, SA, VIC
Mordacia mordax	Short-headed lamprey					Sept - Feb$_{1}$; Aug - Nov$_{10}$			VIC, NSW, SA, TAS
Aldrichetta forsteri	Yellow-eyed mullet		14_{8}	24?$_{8}$		Dec - Mar$_{8}$			NSW, VIC, SA, WA
Mugil cephalus	Mangrove mullet								WA
Maccullochella peelii peelii	Murray cod		10_{2}	37_{2}	26.5_{24}	Sept – Dec$_{2}$	>15_{1}		SA, NSW, VIC
Macquaria ambigua ambigua	Golden perch		4_{2}	37_{2}	29_{24}	Sept - Feb$_{1}$; Oct - Dec$_{2}$	>20_{1}	Eggs 27-31_{26}	SA, ACT, NSW, VIC
Bostockia porosa	Nightfish					Aug - Sept$_{10}$			WA

Scientific name	Common name	Status	Lower temp (°C) threshold	Upper temp (°C) threshold	Optimal range (°C)	Spawning time	Spawning temperature (°C)	Early life stage thresholds (°C)	Distribution in southern Australia
Maccullochella macquariensis	Trout cod	Threatened			~26.5[28]	Sept – Nov[2]	14-22[18]	Eggs >15.5[24]	VIC, NSW
Macquaria australasica	Macquarie perch	Threatened				Oct - Dec[1,2]	>16.5[19]	hatching at 15-17°C[1]	VIC, NSW, ACT
Macquaria colonorum	Estuary perch					Jul - Aug[10]			SA, VIC, NSW, TAS
Macquaria novemaculeata	Australian bass					May – Aug[10]			VIC, NSW
Tandanus tandanus	Freshwater catfish	Declining	4[2]	38[2]		Oct – Dec[2]	20-24[1]		VIC, NSW, SA
Tanandus bostocki	Freshwater cobbler							hatching 7 days at 19-20°C[11]	WA
Prototroctes maraena	Australian grayling								VIC, NSW, TAS
Pseudomugil signifer	Pacific blue-eye		15[10]	28[10]					NSW
Retropinna semoni	Smelt	8.7[28]	37.9[28]	28[2]; 37.9[28]	< 30[28]	Sept - Feb[2]; Jul – Mar[10]	15[10]; 15 -18[35]	15 - 18[35]	SA, VIC, NSW

Scientific name	Common name	Status	Lower temp (°C) threshold	Upper temp (°C) threshold	Optimal range (°C)	Spawning time	Spawning temperature (°C)	Early life stage thresholds (°C)	Distribution in southern Australia
Retropinna tasmanica	Tasmanian smelt					Sept - Feb			TAS
Bidyanus bidyanus	Silver perch; black bream	Threatened	2_{10}	37_{2}		Sept - Feb$_{1}$	23-30$_{10}$		SA, NSW, VIC, ACT
Leiopotherapon unicolor	Spangled perch		$5_{2,}$	44_{2}	>7.2$_{1}$	Nov - Feb$_{1,2}$	20 - 26$_{10}$	hatch in 45-55 hours at 23-26$_{1}$	NSW
Amniataba caudavittata	Yellow trumpeter								WA
Alien									
Cichlasoma nigrofasciatum	Convict cichlid			33*$_{20}$				29 during maturation*$_{20}$	VIC
Tilapia mariae	Black mangrove cichlid		11*$_{21}$	37*$_{21}$	25 - 31*$_{21}$				VIC
Misgurnus anguillicaudatus	Oriental weatherloach		2_{38}	> 42$_{1}$	2-30$_{1}$	Dec – Feb$_{1}$	18 – 30*$_{25}$		ACT, NSW, VIC, SA
Carassius auratus	Goldfish		0.3*$_{22}$	43.6*$_{22}$		Dec – Feb$_{1,2}$	17-23$_{1}$		Widespread & abundant

Scientific name	Common name	Status	Lower temp (°C) threshold	Upper temp (°C) threshold	Optimal range (°C)	Spawning time	Spawning temperature (°C)	Early life stage thresholds (°C)	Distribution in southern Australia
Cyprinus carpio	Carp		0*[23]	30*[23]	~25*[23]	Oct – Dec[2]	17-25[1;] 15-25[3]		Widespread & abundant
Rutilus rutilus	Roach								VIC
Tinca tinca	Tench		1 - 12[37]	26 - 39[37]		Sept - Feb[1]	18 - 27[37]		VIC
Perca fluviatilis	Redfin perch			31[1,4]	23.7 – 26.8*[27]	Aug - Sept[2]	11-15[5]		Widely distributed through MDB, WA
Gambusia holbrooki	Eastern gambusia		0.5[6,7]	38[6,7] 44[8]	23.2 – 36.2*[27]	Dec - Feb[2]	25 - 38[31]		Widespread & abundant
Gambusia affinis	Mosquitofish								WA
Phalloceros caudimaculatus	Speckled mosquitofish								WA
Salmo trutta	Brown trout			22-26[9]	<15[1, 10]; 10 - 20[11]; 12-19[9]	Jun – Aug[28]	< 22[36]	hatching <15*[12]	SA, VIC, ACT, NSW, WA
Salvelinus fontinalis	Brook char;			30*[12]		Feb - May[1]		hatching	TAS, NSW

Scientific name	Common name	Status	Lower temp (°C) threshold	Upper temp (°C) threshold	Optimal range (°C)	Spawning time	Spawning temperature (°C)	Early life stage thresholds (°C)	Distribution in southern Australia
	brook trout							$<15^{*}{}_{12}$	
Oncorhynchus mykiss	Rainbow trout		$0 - 4_{37}$	31_{13}	$10 - 22_{10}$	Apr - Aug$_{1}$	$< 22_{36}$	Eggs $> 18_{24}$	VIC, ACT, NSW, SA, TAS
Salmo salar	Atlantic salmon			$32^{*}{}_{12}$		May - Nov $_{1}$			VIC, NSW, TAS
Salmo gairdneri	Rainbow trout							hatching $<18.{}^{*}5_{12}$	WA

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