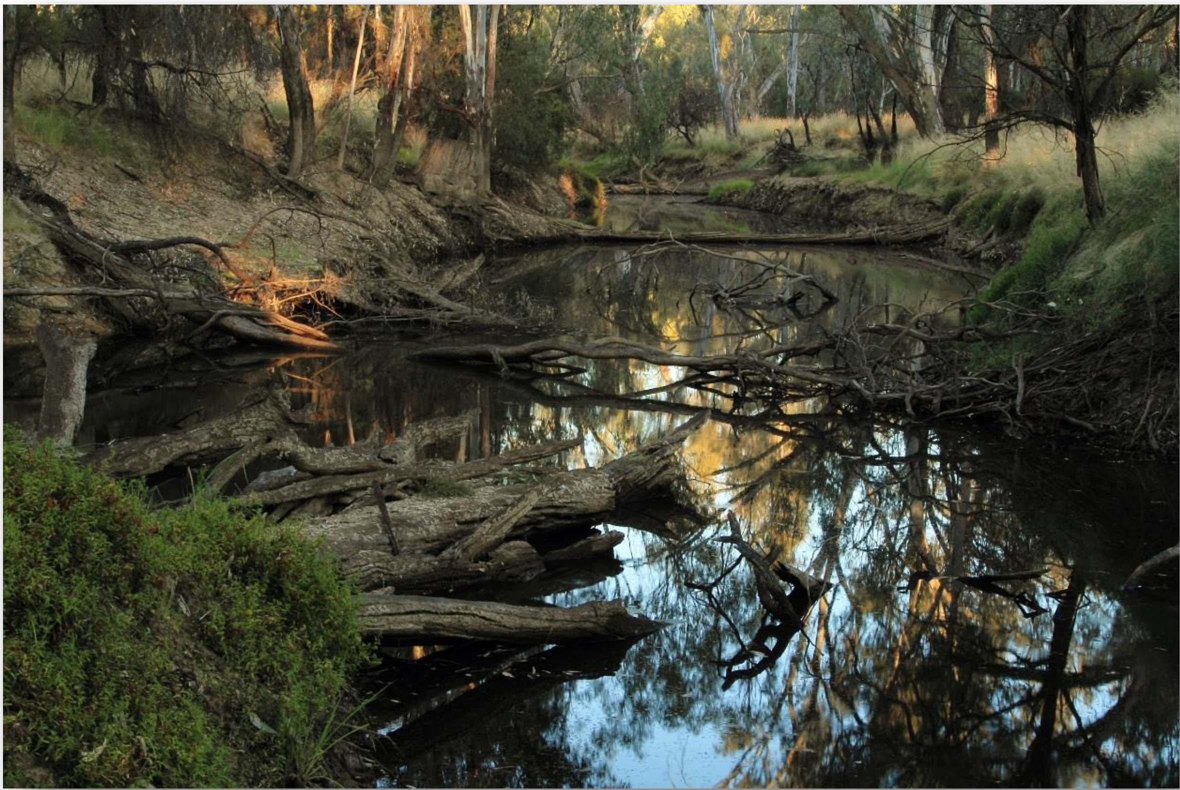
The Murray-Darling Freshwater Research Centre

The potential impacts of climate change on water quality in the southern Murray–Darling Basin

Prepared by: Darren Baldwin



Australian Government

MURRAY–DARLING BASIN AUTHORITY

Final Report

MDFRC Publication 139

The potential impacts of climate change on water quality in the southern Murray–Darling Basin

Final Report prepared for the Murray–Darling Basin Authority by The Murray–Darling Freshwater Research Centre.

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Table of Contents

The potential impacts of climate change on water quality in the southern Murray–Darling Basin

About this document — Executive summary

Manifestations of climate change	Key water quality processes						
	Blue-green algal blooms	Acidification	Mobilisation of sediments	Salinity	Eutrophication	Mobilisation of heavy metals and metalloids	Hypoxia
Increased atmospheric CO_2		+					+
Increase in water temperatures	+++	+			+		++
Increased ephemerality	++	++		+	+		++
Increased incidence and intensity of bushfires	+	+	+++	+	+	+++	+
Increased storm intensity			++	+	+	++	+
Increased incidence of dust storms			+		+	+	

Predicted consequences of climate change: blank = little or no impact; + will slightly increase the impact or occurrence; ++ will increase the impact or occurrence; +++ will substantially increase the impact or occurrence.

About this document

Return to matrix

This report explores potential impacts of climate change on known water quality issues in the southern Murray–Darling Basin. It is presented as a series of essays varying in length from 500 to 5000 words depending on available information and potential importance of the phenomenon into the future. The use of stand-alone essays was adopted because the different manifestations of climate change can interact with different water quality parameters that may be of interest to managers. Similarly, the same water quality parameter may be impacted by multiple manifestations of climate change. The complexity of interactions makes generation of a linear narrative (typically found in a report) difficult. Furthermore, each essay lent itself to different approaches. For example, during the preparation of this report, a mega blue-green algal bloom occurred along the Murray River, which meant that the essay on blue-green is more immediate in tone and data than the other essays.

Producing the report as a series of essays rather than a linear narrative can lead to a certain degree of repetition. To minimise repetition, hyperlinks can be used to navigate throughout the document. Therefore, the document is best viewed on a computer screen (ultimately it is hoped that the document will be published on the web).

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Executive summary

Return to matrix

There have been a number of predictions suggesting that the weather pattern in the southern Murray–Darling Basin (MDB) will be affected by climate change in the coming century, which in turn, will impact on known water quality issues in the MDB. Of all the issues associated with climate change, it is likely that increasing water temperature, increasing incidence of ephemerality (through drought), greater frequency of bush fires, and greater intensity of storms will have the most dramatic effects.

Increased temperature will most likely result in mega blooms of blue-green algae in river systems of the southern MDB into the future. Blooms of blue-green algae in river systems (as opposed to in standing water) are rare. There have been five blue-green algal blooms in the Murray River in the last 13 years, and the frequency of such blooms will probably increase into the future with changing climate (particularly increased temperature). Most blooms originate in Lake Hume and then progress downstream. Until this year, blooms only formed when the level in Lake Hume Fell below 10%. This year's bloom differed in that it was formed by an unusual species (*Chrysosporum ovalisporum*) and Lake Hume was at 35% when the bloom was first reported. I hypothesise, that above long-term average air temperature and solar irradiation coupled with river operations (particularly cold water inflows from the Murray River) contributed to the formation of the bloom. However, a lack of useful monitoring data within Lake Hume has hampered any attempt at definitively identifying the underlying causes of the bloom. I propose developing a monitoring program for Lake Hume (and potentially Lake Mulwala) to inform mitigating strategies for what is, most likely, going to be an ongoing problem.

Increased ephemerality (as a consequence of drought) will periodically cause water quality issues into the future. It is well documented that numerous wetlands and lakes in the Basin (Including Lake Albert and Lake Alexandrina contain acid sulfate soils. If these sediments are exposed to the atmosphere they produce acid. Acidification events associated with sediment drying (similar to those observed during the Millennium Drought) are more likely to occur into the future. Ephemerality can also cause periodic water quality issues if river continuums become fragmented into pools because of drought. Inundating fragmented pools or dry river channels can produce poor water quality at the wetted front.

The largest risk associated with increased fires, coupled with more intense storm events is the mobilisation of sediments from the upper catchment. Of particular concern is the mobilisation of the substantial store of mine tailings (of the order of a billion cubic metres) that have accumulated in the upper catchment as a result of historical gold mining. These tailings are rich in arsenic and may be contaminated with mercury.

Return to matrix

Increased atmospheric carbon dioxide

Return to matrix

Increased atmospheric concentrations of carbon dioxide (CO_2), is one of the principal drivers of climate change. Concentrations of CO_2 in the atmosphere have increased from about 280 ppm to 400 ppm currently, and are expected to reach 750 ppm before the end of the century (Riebesell et al. 2000; NASA undated). Increased carbon dioxide concentrations not only impact on water quality through changes in climate, but the increased concentrations themselves can impact directly on water quality — specifically through changes in pH, and, potentially, by enhancing the risk of hypoxic blackwater events.

pH

As the concentration of CO_2 in the atmosphere increases, there is a concomitant increase in the amount of CO_2 dissolved in the water column (following Henry's law). Dissolved CO_2 is related to pH by the equation:

$$[CO_{2\ aq}] \quad \Longleftrightarrow \quad [H^+] + [HCO_3^-] \quad K = 4.25 \times 10\text{-}7 \qquad (1)$$

where [CO_2 aq] is the concentration of dissolved CO_2 in solution, [H^+] is the concentration of protons; [HCO3-] is the concentration of bicarbonate ion and K is the equilibrium constant. pH is defined as the negative log of the concentration of protons. As the concentration of CO_2 in the water column increases, the pH in the water column falls.

The pH in the oceans has fallen by about 0.3 pH units due to anthropogenic increases in atmospheric CO_2. Because pH is on a log scale, 0.3 units represents about a 50% change (increase in protons). Unlike freshwaters (see below), in the absence of shifts in atmospheric CO_2 concentrations, the pH in the ocean should be quite stable over relative long (century) time scales. Therefore, because marine organisms have adapted to relatively constant oceanic pH levels, small shifts in pH can have quite dramatic effects on some marine organisms, particularly those that have a calcareous shell or external skeleton (Bonney et al. 2009).

The pH levels in inland waters in the southern MDB tend to be much more variable than in oceanic waters. Table 1 shows the maximum range and the range of the middle 90% of pH measurements at 4 sites on the Murray River: Jingellic, Yarrawonga, Swan Hill and Morgan.

Table 1. The pH levels at key sites in the Murray River (data supplied by the MDBA).

Site	Maximum pH range	Middle 90% of observations	Years of observation	Number of observations
Jingellic	5.6 – 9.3	6.3 – 7.7	1978 – 2012	1690
Yarrawonga	5.6 – 9.0	6.7 – 8.0	1975 – 2012	2165
Swan Hill	6.0 – 9.0	6.6 – 7.8	1976 – 2012	1818
Morgan	6.7 – 9.5	7.4 – 8.4	1968 – 2013	1914

In the 35+ years of observations, pH varied by at least 3 pH units (three orders of magnitude) between the lowest and highest recorded pH observation at each of the four sites. If the highest and lowest 5% of observations are excluded, the range is greatly narrowed, but is still at least an order of magnitude difference. Such variability would indicate an adaptation of Australian native species to a varied pH regime. A detailed assessment of water quality tolerances of aquatic organisms from the MDB (Watson et al. 2009) showed that there haven't been specific ecotoxicological tests to determine pH tolerance of MDB species. Rather, the only data available was the pH of the water where a particular organism (invariably fish) was observed. Some species like the Australian smelt (*Retropinna semoni*) seem to be quite tolerant — being found in different waterbodies with pH levels as low as 3.7 and as high as 9.8. However, these observations don't necessarily tell how healthy the populations were at these extreme values.

Climate change: In order to roughly assess the impact of increased CO_2 on pH in the Murray River, I used available pH and bicarbonate concentrations at Morgan (1968–2013) to calculate dissolved CO_2 levels using equation (1) for the middle 90% of observations. I then calculated what the pH would have been if the concentration of dissolved CO_2 doubled (consistent with a doubling of atmospheric CO_2 levels) and the bicarbonate concentration remained the same. A doubling of the CO_2 concentration would have led to a reduction in pH by about 0.3 pH units. This is a similar to modelling that suggested about a 0.5 unit decline in pH for streams in the Australian Alps under a 1000 ppm CO_2 atmosphere. The ecological significance of a 0.3 decline in pH is, as yet, unknown. However, given the other threats to aquatic pH in the southern MDB, especially from the exposure of sulfidic sediments, shifts in pH from increases in atmospheric CO_2 levels is probably a secondary problem.

Hypoxic blackwater events

Increased atmospheric CO_2 may contribute to a greater risk of generating hypoxic blackwater. Experiments have shown that increased atmospheric CO_2 can lead to greater photosynthesis by plants, greater above-ground biomass, and a decline in overall moisture requirements (Taub 2010) including for *Eucalyptus* sp. (da Silva and Ghini 2014). This increased growth has the potential to increase the biomass of leaf litter on the floodplain, which in turn equates to more carbon being exported from the floodplain. One of the key drivers of hypoxia blackwater is the amount of dissolved organic carbon (DOC) that is available for microbial respiration.

References

Return to matrix

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Increased water temperature

Return to matrix

Predicted air temperature in southern Australia

The Intergovernmental Panel on Climate Change (Christensen et al. 2007) has made a number of predictions regarding the climate of southern Australia in the period 2080–2099 compared to 1980–1999 based on the relatively conservative A1B scenario group (Cartwright and Simonds 2008) using 21 climate models. They estimate that average temperatures will be between about 2.4 and 2.8 °C higher by the end of this century. They also predict that it is very likely that southern Australia will have more hot summer days, more intense and more frequent hot spells, warmer overnight temperatures with fewer cold nights, and fewer frosts (Christensen et al. 2007).

Table 1. Predicted increase in temperature (°C) from a baseline of 1980–1999 to 2080–2099 (Christensen et al. 2007).

Season	Minimum	25%	50%	75%	Maximum
Summer	2.0 °C	2.4 °C	**2.7 °C**	3.2 °C	4.2 °C
Autumn	2.0 °C	2.2 °C	**2.5 °C**	2.8 °C	3.9 °C
Winter	1.7 °C	2.0 °C	**2.3 °C**	2.5 °C	3.5 °C
Spring	2.0 °C	2.6 °C	**2.8 °C**	3.0 °C	4.1 °C
Annual	1.9 °C	2.4 °C	**2.6 °C**	2.8 °C	3.9 °C

Although there are many mitigating factors such as extent of shading, wind speed and cold-water releases from dams, generally speaking water temperature is strongly influenced by ambient air temperature (Kelly and Koehler 1996), so it highly likely that water temperatures will increase in the southern MDB into the future. Increased water temperatures will impact on a variety of physical, biological and biogeochemical processes that affect water quality, occasionally in contradictory ways.

Dissolved oxygen

Less oxygen can physically dissolve in warmer water, so increased water temperature will lead to a reduction in the saturated dissolved oxygen concentration in water. However, this effect may be minor compared to other effects. Warmer water temperatures have the potential to increase the biomass of algae, particularly blue-green algae. Dissolved oxygen concentrations in water impacted by algal blooms can go from supersaturation to near hypoxia in a 24 hour period as the algae shift from photosynthesis during the day to respiration at night (e.g. McDonnel and Kountnz 1966).

Warmer water temperatures can lead to stronger thermal stratification in non-flowing water bodies, including dams, reservoirs and isolated pools. Thermal stratification is the differential heating of surface water, which absorbs heat and so becomes less dense and floats on the cooler bottom waters. The extent of stratification is primarily determined by the relative input of thermal energy (causing stratification) and turbulent kinetic energy (breaking down stratification; Bormans et al. 1997). The stronger the greater the temperature differential and depth to the hermocline, the more energy that is required to breakdown stratification. Because of oxygen drawdown by microbial processes occurring in sediments, the bottom water can become anoxic, which in turn can result in the mobilisation of nutrients, and the subsequent promotion of algal blooms (as was the case in Lake Hume in the 2006/7 algal bloom; Baldwin et al. 2008).

Warmer water temperature has been identified as a key risk factor in predicting the onset of hypoxia during a blackwater event (Whitworth, Baldwin & Kerr 2014). As the water temperature rises, both the rate and amount of DOC leached from leaf litter increases. Furthermore, for every 10 °C rise in water temperature, the rate of DOC consumption approximately doubles (Figure 1). More available carbon and faster rates of decomposition means that oxygen in the water column will be consumed faster; hence, the risk of hypoxia is increased.

Microbial processes

The approximate doubling of the rate of microbial respiration with each 10 °C increase in temperature will impact other biogeochemical cycles mediated by microbes, which include sulfate reduction (responsible for the development of acid sulfate soils), iron reduction (responsible for the release of phosphorus from sediments under anaerobic conditions) and the myriad of processes that affect the cycling of nitrogen.

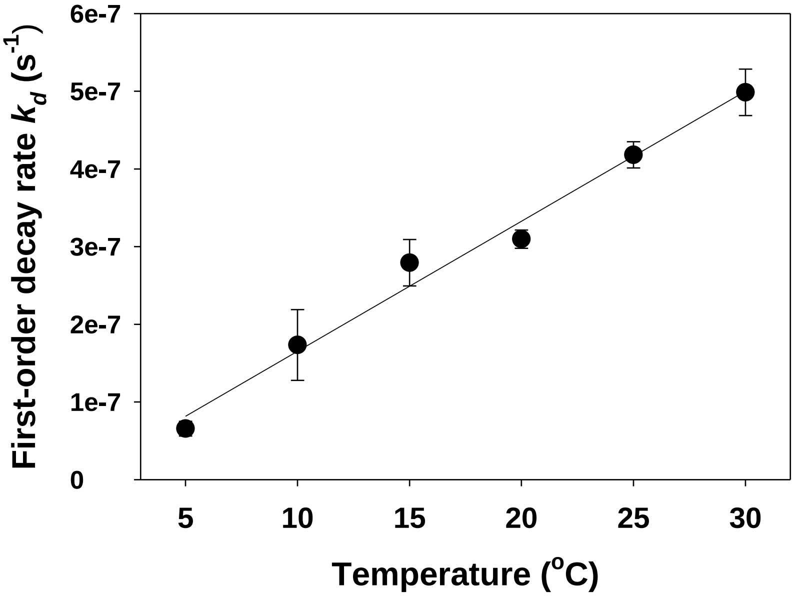

Figure 1. Change of the first order rate constant for DOC consumption with temperature (adapted from Whitworth, Baldwin & Kerr 2014).

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Return to matrix

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Increased ephemerality

Return to matrix

Ephemerality in the southern Murray–Darling Basin

Many aquatic ecosystems in the MDB are naturally ephemeral. This includes headwater streams in the upper catchments, lowland rivers and streams and floodplain wetlands and flood runners. Furthermore, river regulation has imposed or increased ephemerality on some systems. As an example, Figure 1(a) shows the number of years in the period 1890–2007 when flows would have entered Tuppal Creek, a distributary from the Murray to the Edwards River, under natural conditions (based on MSM-BigMod modelling data). In only a small number of years, there would have been no flow into the creek, with most years having flows lasting four months or longer. This contrasts with the results from modelling for the same period but assuming the current level of river regulation (Figure 1(b). Under that scenario, flow enters the creek in very few years.

The Intergovernmental Panel on Climate Change (Christensen et al. 2007) has made a number of predictions regarding the climate of southern Australia in the period 2080–2099 compared to 1980–1999 based on the relatively conservative A1B scenario group (Cartwright and Simonds 2008) using 21 climate models. One of the predictions is that there will be an overall decline in rainfall (Table 1). One of the consequences of a drying climate is that many creeks and rivers that had permanent flow will, at times, cease to flow and possibly completely dry out; while current ephemeral systems (including wetlands) may stay dry for longer. The increase in ephemerality will impact on water quality in the stream or river reach following cease-to-flow (or loss of connection in the case of wetlands), as well as have downstream impacts once flows resume.

Table 1. Predicted changes in precipitation in southern Australia (from Christensen et al. 2007).

Season	Minimum	25 %	**50 %**	75 %	Maximum
Summer	-23 %	-12%	**-2%**	12 %	30 %
Autumn	-31 %	-9%	**-5%**	13 %	32 %
Winter	-32 %	-20%	**-11%**	-4 %	9 %
Spring	-42 %	-27%	**-14 %**	-5 %	4 %
Annual	-27 %	-13%	**- 4 %**	3 %	12 %

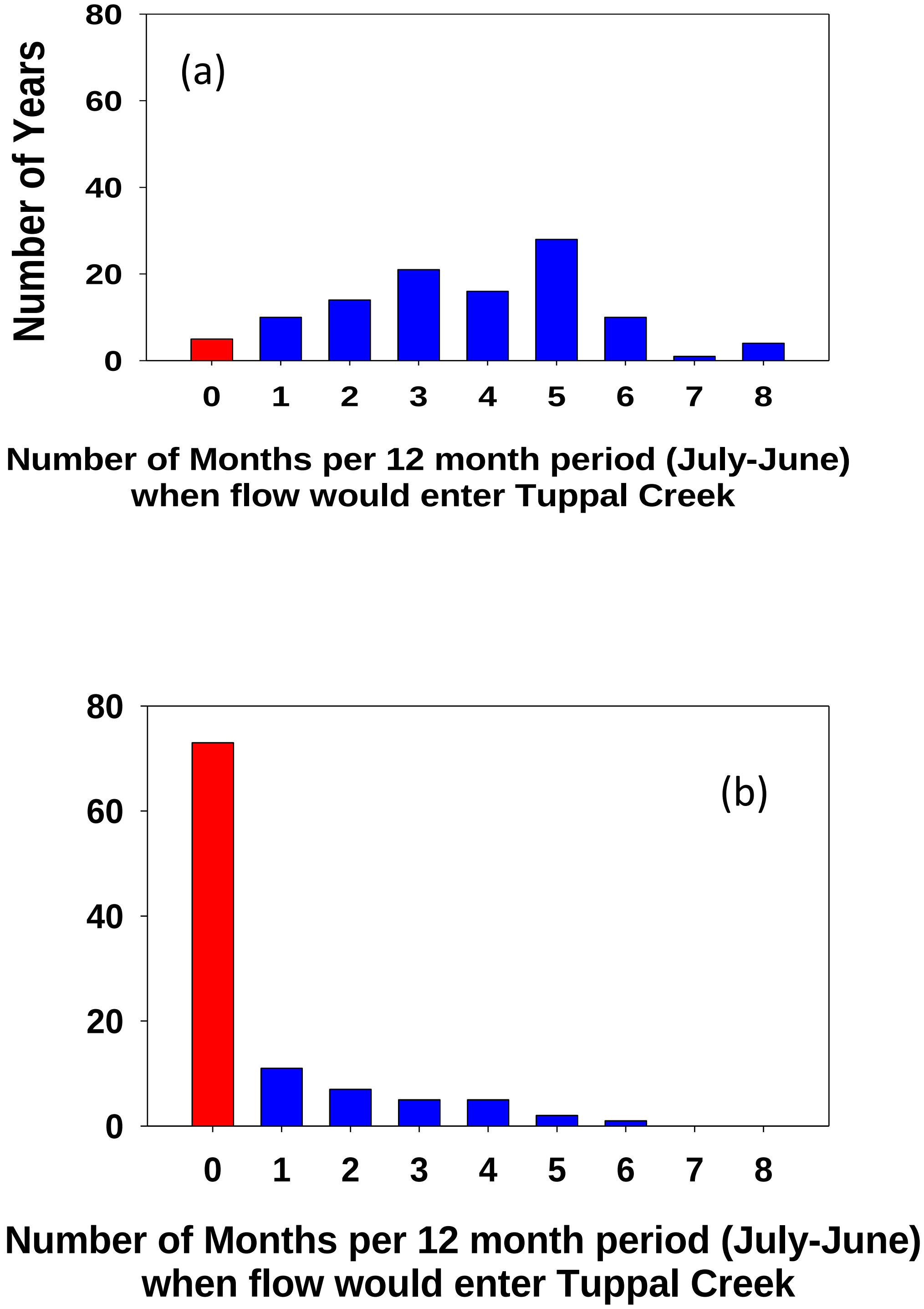

Figure 1. Changes in the incidences of flows into Tuppal Creek under (a) modelled natural conditions and (b) modelled actual conditions. Data supplied by the MDBA.

Changes in water physico-chemistry during drying and re-wetting [1]

Conceptually, the trajectory of key water physico-chemical parameters can be divided into two distinct phases following cease-to-flow. The first is the formation of remnant pools followed by slow contraction of the pool extent over time. Not all intermittent systems will form remnant pools and, therefore, this phase may not be relevant for all systems. The second phase is the exposure to air and subsequent drying of sediments. Obviously, if there is no water present, then it is impossible to measure water physio-chemistry. However, sediment exposure and subsequent drying will impact on water quality in the ephemeral systems once flow resumes — discussed below (Baldwin and Mitchell 2000; Mikha, Rice & Milliken 2005; McIntyre, Adams & Grierson 2009; Gómez et al. 2012).

Physico-chemistry in remnant pools

Temperature: The temperature regime within a residual pool of an ephemeral system is influenced by a number of factors including ambient air temperature, solar irradiation, wind speed and direction, pool depth and whether or not there is interaction with groundwater.

Water temperature will be strongly influenced by ambient air temperature and the amount of solar irradiance reaching the pool surface (Kelly and Koehler 1996). In temperate climates, cease-to-flow generally occurs in the warmer months, which coincides with the highest levels of solar irradiance; hence, water temperatures at these sites will generally increase during the drying phase. This is not necessarily the case in the wet/dry tropics, where cease-to-flow may actually occur in the cooler months (Townsend, Webster & Schult 2011). The influence of solar irradiation on the water temperature regime in a residual pool can be tempered by shading either by riparian vegetation (Rutherford et al. 2004) or landscape features such as adjacent mountains (Hrachowitz et al. 2010). It follows that residual pools in ephemeral streams in deep mountain valleys will generally have cooler water temperatures than those on open plains. Similarly, a pool with complete canopy cover will be cooler than an otherwise identical pool with sparse or no canopy cover.

Pool depth will strongly influence the temperature regime in a residual pool. Shallow pools will tend to heat up faster than deeper pools. Also, because shallow pools have lesser thermal mass than deeper pools, the diurnal variation in them would be expected to be greater than that in deeper pools — especially at depth (e.g. Turner and Erskine 2005). Indeed, deep (ca. 4 m) residual pools in ephemeral rivers have been shown to thermally stratify, with stratification lasting in excess of days (Baldwin and Wallace 2009). It has been reported that the cooler hypolimnetic water has been used as thermal refuge for heat-stressed organisms (e.g. Nielsen, Lisle & Ozaki 1994). However, this assumes that the dissolved oxygen concentration in the hypolimnion remains at levels that can support life (Turner and Erskine 2005). Following stratification, if the pool contains organic-rich sediments, sediment oxygen demand can deplete dissolved oxygen in the hypolimnion leading to hypoxic or even anoxic conditions (e.g. Baldwin and Wallace 2009).

Water temperature at the bottom of residual pools can also be moderated by the influx of (generally cooler) groundwater. Again, groundwater plumes have been known to be used as thermal refuges for temperature sensitive organisms (e.g. Nielsen, Lisle & Ozaki 1994).

Dissolved oxygen: The concentration of dissolved oxygen in a residual pool will depend to a large part on the overall temperature of the pool, whether or not stratification has occurred, whether or

[1] This section will appear in the yet to be published book chapter: Gomez R, Isabel M, Baldwin D.S. and Dahm C (2017) Water Physiochemistry in Intermittent Rivers and Ephemeral Streams in *Intermittent Rivers: Ecology and Management* T. Datry, N. Bonada and A. Boulton (eds).

not the pool can support and sustain substantial levels of primary productivity and the amount of allochthonous carbon input into the pool.
The amount of oxygen dissolved in water is directly dependent on the temperature of the water because of saturation — the warmer the temperature the lower the concentration of oxygen that can be dissolved in it (e.g. Hladyz et al. 2011). Temperature can also indirectly influence dissolved oxygen levels through thermal stratification, and impacts on both the rates of primary productivity and microbial activity (e.g. Baldwin and Wallace 2009). As noted above, deep pools can stratify, and if the pool has organic-rich sediments, microbial activity in the sediments can draw down oxygen levels in the hypolimnion leading to anoxia in the bottom water. Anoxia also leads to the mobilisation of nutrients, particularly N and P from the sediments to the overlying water column (e.g. Mortimer 1941; Baldwin and Williams 2007). If the thermocline breaks down, for example because of wind action (Turner and Erskine 2005) or a small unseasonal inflow into the pool (Baldwin and Wallace 2009), the anoxic nutrient-rich water will mix with the overlying water column. This will immediately cause a decline in the overall oxygen level within the pool. For example, a period of no flow in the Darling River in Australia resulted in the river establishment of disconnected pools followed by stratification. The arrival of an engineered flow release resulted in a mixing of the anoxic bottom water with the oxic surface waters. Mixing resulted in dissolved oxygen in the whole water column falling to below 2 mgL^{-1} (Baldwin and Wallace 2009).

Rates of primary production can increase in pools following cease-to-flow (e.g. Townsend, Webster & Schult 2011), which in turn, can lead to substantial modification of diurnal oxygen concentrations in the pool — increasing during the day into early evening, and then declining overnight. Primary production is favoured by the warmer temperatures associated with cease-to-flow (see above), but the increase in plant and/or algal biomass also requires a source of bioavailable nutrients. Fine organic-rich sediments are more likely to be a source of nutrients to the overlying water column than bedrock, cobbles or sand. Nutrients can also enter the remnant pool through interactions with groundwater, or anthropogenic sources such as allowing stock access to the pool.

Another modifier of dissolved oxygen in remnant pools following cease-to-flow is allochthonous carbon input from riparian litter (Hladyz et al. 2011). DOC leached from leaves falling into the pool can be metabolised by bacteria, in the process utilising oxygen. If the rate of microbial metabolism is greater than the rate of re-supply of oxygen to the pool from the atmosphere, levels of dissolved oxygen in the pool will begin to decline. Whether or not this is an important process in a given pool will depend on the timing of litter fall, the amount of carbon entering the pool (relative to the size of the pool) and to a lesser extent, the type of litter. Generally, the impact will be greatest when most of the litter enters the pool during the warmer months as the rate of microbial metabolism and, hence potential to draw oxygen concentrations down, is temperature-dependent (Whitworth and Baldwin 2014).

Salinity: The salinity within a residual pool will depend on the water quality in the pool at the time of disconnection, the volume of the residual pool (concentration x volume = load) and whether or not the pool interacts with groundwater. In the absence of groundwater inputs, the salinity in the pool will increase during the drying phase principally through evapoconcentration. For example, Hamilton et al. (2005) showed through the use of both conservative ion tracer concentration and oxygen and hydrogen isotope fractionation that evaporative water loss controlled the water levels in 15 residual pools in Cooper Creek, in Central Australia. In the absence of surface flows, the residual pools could fall to 10% of their bank-full volumes in 6–23 months, depending on the morphology of the pool (Hamilton et al 2005). If the pool does interact with groundwater, the salinity in the pool will, at some point, reflect the salinity in the groundwater (Costelloe, Western & Irvine 2007).

Turbidity: Turbidity in the residual pool will depend on the nature of the sediments in the pool at the time of cease-to-flow, the morphology of the pool relative to prevailing winds (e.g. Kreiling et al. 2007), the presence of organisms in the pool that are capable of bioturbation (Flecher et al. 1985; King et al. 1997; Braig et al. 2003), and whether or not an algal bloom occurs in the pool (discussed later). Based on the principles of sedimentology, residual pools with course substrate (bedrock, cobbles or sand), turbidity in the pool should remain low through the drying phase (Mulder and Alexander 2001). Conversely, in pools with finer substrate, the substrate can be suspended through wind action or bioturbation. Wind-driven turbidity will be more prevalent in shallower water than deeper water, and therefore, turbidity should increase as the water depth in the pool declines with evaporation. Pools with fine sediment are also more likely to have algal blooms (discussed below), which will also enhance turbidity. Turbidity is an important factor affecting dissolved oxygen in the water column through its effect on photosynthetic rates (Davies-Colley and Smith 2001).

pH: The pH in pools following cease-to-flow will be influenced by the inherent buffering capacity in the water column and sediments, whether or not there is an increase in primary productivity in the pool following cease-to-flow, and in certain circumstances if the pool interacts with groundwater. The buffering capacity of the both the water column and the sediments will be strongly influenced by the geology in the catchment, and hence, will most likely be spatially variable. An increase in primary productivity following cease-to-flow (discussed above) will also affect the pH of the pool by shifting the $CO_2/HCO_3^-/CO_3^{2-}$ equilibrium during photosynthesis and respiration. Removal of CO_2 during photosynthesis will raise pH, with a maximum sometime in the late afternoon, while an increase in CO_2 through respiration will lower pH overnight.

Extremely low pH levels (pH < 4 and as low as 1.8) have been observed in a large number ephemeral creeks, rivers and wetlands during the drying phase in some parts of the world (Hall et al. 2006; Lamers et al. 2001; Lamontagne et al. 2006). Investigations have shown that these waterbodies have been exposed to saline groundwater containing elevated levels of sulfate. Sulfate reduction in the sediments led to the accumulation of large stores of reactive metal sulfides. While the sediments remained submerged, the sulfidic material remained inert, but as the water levels in these systems began to fall, the sulfidic material began to oxidise, in the process producing acid (Baldwin and Fraser 2009).

Complete drying and the re-wetting front

Changes that occur to substrates following complete drying of ephemeral systems can have impacts on water quality following the flow resumption. Probably the most important processes are the (i) oxidation and subsequent crystallisation of sediments on exposure to air followed by desiccation, (ii) the effect of desiccation on soil/sediment micro-organisms and (iii) the build-up of organic matter and/or salt in the bed of the water course.

During the drying process, sediments are exposed to the air. As many fine, organic-rich sediments are anoxic below the top few millimeters when covered by water, any reduced mineral phases will be oxidised when exposed to air. As noted above, if reduced sulfur species (e.g. metal sulfides, elemental sulfur) are present in the sediments, it can lead to the mobilisation of acid, leading to a fall in sediment pH. On re-wetting, this acid can be mobilised, leading to acidic water in the wetting front (Baldwin and Fraser 2009).

During desiccation, mineral phases begin to convert to more crystalline forms. In the case of iron, during desiccation amorphous iron oxyhydroxides are converted to more crystalline phases such as goethite and hematite. Iron oxides, in particular, play an important role in the cycling of phosphorus.

As the iron oxides become more crystalline, their affinity for phosphorus declines (Attygalla et al. 2016), and on re-wetting they will release a pulse of bioavailable phosphorus (Attygalla 2015).

Extreme desiccation of sediments can lead to a process called the *Birch effect*, where, on re-wetting there is a pulse of carbon and nitrogen released from the sediments, coupled with a sharp increase in sediment respiration. There is still some conjecture on the underlying drivers of the phenomenon, but current thinking suggests that the pulse of carbon and nitrogen comes from organic osmolytes produced by bacteria (Lado-Monserrat et al. 2014). During extreme drying events, there is a significant contraction of the soil water solution (the film of water surrounding soil particles in which soil bacteria live). Because of this contraction, individual bacterial cells are confronted with a substantial change in the osmotic potential in their immediate environment. In response, they produce osmolytes (organic molecules — some of which contain nitrogen) to help regulate their cell volumes (Warren 2014). Once the soil water is replenished (for example on reflooding), these osmolytes are released from the microbial cells; hence, the pulse of carbon and nitrogen. The organic osmolytes can be used by microbes for respiration, hence the increase in cell respiration.

During the dry phase organic matter from different sources can also accumulate in the channel of the water course during the dry phase. Some will derive from aquatic organisms killed during the drying process; although some of this organic matter is used as a subsidy (*sensu* Polis, Anderson & Holt 1997) for terrestrial food webs (Steward et al. 2012). Amphibious and terrestrial plants can colonise water courses following exposure of the sediments. This can be especially important in more arid regions where exposed sediments may contain significantly more soil moisture than surrounding soils. Finally, litter from fringing vegetation, transported by wind, or sporadic rainfalls can accumulate in the dry bed of the water course during the drying phase. On reflooding, the carbon (and nutrients) stored in the bed of the water course can be mobilised on reflooding (e.g. Skoulikidis and Amaxidis 2009). Consumption of DOC released from accumulated litter by aquatic bacteria can result in a rapid decline in dissolved oxygen, resulting in hypoxia or anoxia in the wetted front (Hladyz et al. 2011). Inundation will lead to the death of terrestrial vegetation that had colonised the bed of the water course, with the potential to further suppress dissolved oxygen, but over a longer time frame.

Finally, if reaches of the water course had interacted with groundwater during the drying phase, any salt subsequently stored in the sediments will be mobilised on re-wetting. Taken together, depending on the circumstances, the wetted front following inundation of a dry water course has the potential to be low in dissolved oxygen, saline, nutrient-rich and/or acidic. However, the potential pulse of poor water quality on commence-to-flow is a direct (and natural) consequence of the drying of ephemeral systems (Skoulikidis and Amaxidis 2009) and should be taken into consideration in framing water quality objectives for these systems (Skoulikidis 2008).

Conclusion

As more systems become ephemeral or the period of drying increases, water quality in those systems will likely decline. As river systems dry to pools, the likelihood of blue-green algae in those pools will increase. Exposure of sulfidic sediments (common in the lowland regions of the southern MDB, especially in the Edward-Wakool creek system and the lower Murray wetlands) will continue lead to acidification. Re-wetting will lead to the mobilisation of carbon and salt, with the likelihood of anoxia occurring at the wetted front.

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Increases in bushfire occurrence and intensity **Return to matrix**

The impact of bushfires on water quality has recently been reviewed (Smith et al. 2011; Cawson et al. 2012; Bixby et al. 2015; Leigh et al. 2015). Of particular relevance is the review by Smith et al. (2011) who examined the impact of fire on water quality with particular emphasis on studies in south-eastern Australia.

There are a number of processes that occur during the fire event that specifically impact on water quality. The most obvious are the loss of substantial amounts of vegetation biomass, including understory vegetation, the baking of soil surfaces and the generation of smoke, ash, and debris.

The loss of vegetation and the baking of soils directly influence runoff in the catchment post-fire. The loss of vegetation decreases soil contact cover, increasing the soil area directly impacted by rainfall, as well as reduces water loss through plant transpiration (Smith et al. 2011). In addition to the direct effects of the fire on vegetation, substantial areas of land are cleared as fire control lines during the fire event. For example, Dunkerley et al. (2009) estimate that during the 2003 alpine fires in north-eastern Victoria, 9000 km of fire lines, each up to 60 meters across were cut through vegetation exposing the underlying soil. Soil baking leads to soil hydrophobicity, which inhibits rainfall infiltration into the soil profile (Lane, Sheriden & Noske 2006). Taken together, these processes lead to substantial increases in water runoff from burnt catchments. For example, Lane, Sheriden and Noske (2006) have shown that following a fire in the Kiewa Valley, Victoria, runoff in two sub-catchments increased by between 40 and 95%, and these increases in runoff persisted for at least two years post-fire.

The increase in runoff from catchments leads to a concomitant increase in erosion, not just from the burnt zone, but also through gully erosion downstream (Smith et al. 2011). Fires in south-eastern Australia increase sediment yields from catchments by between 1.3 to about 1450 times compared to either pre-burn averages, or sediment yields from adjacent unburnt catchments (Smith et al. 2011). The increase in sediment yield can persist for several years following the fire. Factors influencing the amount of sediment yield from a specific catchment include post-fire rainfall patterns (increasing if there are severe events immediately following the fire), the area burnt and the severity of the fire (Smith et al. 2011). Post-fire salvage harvesting of trees can also increase sediment loads from catchments, with 180-fold greater sediment load reported from a burnt pine plantation with post-fire harvesting compared to an adjacent eucalypt forest without harvesting (Smith et al. 2012). As well as sediment, post-fire rain and wind events will mobilise ash and other debris generated from the fire. Ash contains elevated levels of readily extractable nutrients, carbon, salts and metals, including iron.

Sediment, ash and debris exported from burnt areas have substantial effects on downstream water quality. Firstly, sediment mobilisation can lead to substantial peaks in turbidity — with values of 2000–3000 NTU being reported (Wasson et al. 2004; Dahm et al. 2015), which is at least 40 times higher than would be expected for such systems. Post-fire runoff also contains elevated levels of both dissolved and particulate nutrients — especially nitrogen and phosphorus. For example, Sheridan et al. (2007) showed up to an approximately 450 times increase in phosphorus and 100-fold increase in nitrogen export from a number of tributaries of the Murray River in the year following the 2003 fire. While it is possible for mobilised nitrogen to be lost from aquatic ecosystems through denitrification, mobilised phosphorus will preferentially be stored in downstream depositional zones (e.g. dams and weir pools). As many blue-green algae can fix nitrogen, elevated phosphorus in such environments enhances the likelihood of blue-green algal blooms rather than the proliferation of other taxa (see essay on blue-green algae). It is of note that Emelko et al. (2016)

suggest that the impacts of wildfire on phosphorus dynamics in streams could persist for up to seven years post-fire.

As well as mobilising nutrients, post-fire runoff can also move metals and metalloids in the environment. A North-East Water study (North-East Water 2003 — as cited in Smith et al. 2011) reported at least a greater than 30-fold increase in iron, copper, zinc, chromium, lead, and arsenic levels in the Ovens River following a storm event after the 2003 high country fires. Arsenic and mercury are of particular concern for the Murray River because of the legacy of extensive historical gold-mining activities in the river's upper-catchment during the second half of the nineteenth centenary and the beginning of the twentieth century. It has been estimated that 1.8 billion cubic metres of material has been worked and reworked in the colony of Victoria between 1850 and 1900, with most of that activity in the MDB (P. Davies, Department of Archeology and History, La Trobe University pers. comm.). The gold winning process used substantial amounts of mercury, while gold smelting produced considerable amounts of arsenic (much of the gold deposits were found in arsenopyrite minerals). It has been suggested that elevated levels of arsenic in the sediments of Lake Hume (Baldwin et al. 2008) and mercury in Lake Mulwala (Baldwin and Howitt 2007) and the Ovens River (Churchill, Metheral & Suter 2004) are linked to historical gold mining activities.

Sediment and ash mobilisation has been linked to transient declines in water quality as the material moves through the catchment network. These changes in water quality are analogous to those observed at the wetted front during the first-flush of ephemeral channels. Anoxia, caused by consumption of DOC (Lyon and O'Connor 2008; Dahm et al. 2015), spikes in salinity because of mobilisation of anions and cations (Reale et al. 2015,) and declines in pH (Dahm et al. 2015) have all been reported. The effects seem to be more noticeable in lower-order streams than in higher order streams, prompting Bixby to note 'that flow pathways, geomorphology and biogeochemical processes moderate fire effects on water quality along the river continuum'.

Fires and climate change

There have been a number of review articles and syntheses that have examined the effect of climate change on wildfires (Flannigan, Stocks & Wooton 2000; 2005; 2009). The key factors describing fire activity, particularly in terms of future-casting (Flannigan 2009) are:

- fire weather — i.e. the weather variables that affect the likelihood of a fire starting, propagating and being able to be suppressed. These include temperature, rainfall, humidity and wind.
- area burned — which is influenced by topography, fragmentation, land use and location
- fire occurrence — the likelihood of a fire starting in a specific location based on prevailing climatic and other factors
- fire season — the time of year when fires are most likely to occur.

The effects of climate change on fire activity are estimated by using either global (e.g. Flannigan et al. 2013) or regional (e.g. Pitman, Narisma & McAneney 2007) climate models. Flannigan et al. (2013) used three global climate models to determine the Cumulative Severity Rating (CSR — a weather-based fire danger index) globally for the periods 2014–2050 and 2091–2100 to compare against a reference period (1971–2000). Their modelling suggests that, at a global scale, there will be expected increases in fire occurrence and intensity as well as area burned. For the Australian continent, there is up to a three-fold increase in CSR by 2041–2050 relative to the 1971–2000 baseline. Furthermore, the fire season will be year-round. Also, using global modelling, Huang, Wu and Kaplan (2015) have predicted up to a four-fold increase in fire frequency in Australia in 2050 (relative to a 2000 baseline), with increases in eastern Australia linked to declining rainfall.

Most applications of regional scale modelling to determining changes in fire activity have been focused on North America (Flannigan et al. 2005); however, there have been a number of studies that have looked at future fire activity in Australia. Williams, Karoly and Tapper (2001) modelled the effect of doubling the concentration of atmospheric CO_2 on the MacArthur Forest Fire Index (FDI). They showed that under this scenario, there is an increase in fire danger throughout Australia because of the number of days with high and extreme fire danger. The principal driver of this change was increased maximum temperatures. For the southern-connected MDB, modelling suggested between a 20–30% increase in FDI under the doubling CO_2 scenario. Using a slightly different approach, Pitman et al. (2007) suggested that by 2050 there would be a less than 10% increase (compared to current day) in FDI for the southern MDB, but a 10–25% increase in FDI for the northern MDB; by 2100 this had increased to 10–25% for the southern MDB and 25–50% for the northern MDB. Like Williams et al. 2001, Pittman et al. (2007) suggested increases in FDI were related to temperature, but also impacted by an overall decrease in humidity.

King, de Ligt and Cary (2011) coupled a carbon accounting model and a fire regime simulator to three climate regimes (1975–2005; a moderate changed climate for 2070; and a more extreme 2070 climate model) to predict fire outcomes for the south-eastern Australian High Country. Unsurprisingly, their modelling predicts an increase in fire incidence, larger areas burned and higher fire intensities, apart from other effects.

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Return to matrix

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Increased storm intensity

Return to matrix

While most climate models predict that under climate change, overall there will be less runoff over most of the southern MDB, paradoxically the intensity of storm events is expected to increase (Christensen et al. 2007; CSIRO 2010); with the intensity of large storm events (72 hours duration) increasing until 2030 before declining, and then the intensity of small events (2 hours duration) to continue to rise (CSIRO 2010).

High intensity storm events can lead to increased soil erosion, particularly if there have been fires in the catchment. The effects of these mobilisation events may be of local importance, but their effects can be both quite damaging and long lasting. As an example, the Granite Creeks system near Euroa in north-east Victoria is now impacted by 'sand slugs' — that is, deep pools in the creek system are now filled with sand that was mobilised from the upper catchment. Land clearing, grazing, and gold mining all contributed to soil instability in the upper catchment (Davis and Finlayson 2000). A significant proportion of the sand impacting on the creek system was potentially mobilised in one extreme event — the flood of 1916 (Davis and Finlayson 2000).

Mobilised sediment can have a number of ecological impacts including the smothering of habitat, increased turbidity and the mobilisation of nutrients and heavy metals.

Internationally, storm surges coupled with rising sea levels are seen as a threat to water quality in low lying areas through salinisation of freshwater environments. In the southern-connected Murray–Darling Basin, the only area at risk is the Coorong and Lower Lakes. It is uncertain at this point as to what extent the current barrages can protect the Lower Lakes from salinisation, but it is of note that during the Millennium Drought, in the absence of storm surges, the median salinity at Goolwa increased ten-fold from about 1200 mgL^{-1} (≈ 1700 EC) to about 12000 mgL^{-1} (≈ 17000 EC) through evaporation and the influence of groundwater (Mosely et al. 2012). Interestingly, storm surges should improve water quality in the southern Coorong, because it would permit mixing.

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Return to matrix

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Dust storms

Return to matrix

Dust storms have the potential to impact on water quality. These storms can transport significant amounts of material. For example, it has been estimated that a large dust storm that affected south-eastern Australia on the 23 October 2002 mobilised between about 3 to 5 million tonnes of sediment (McTainsh et al. 2005), while a dust storm on the 23 of September 2009 mobilised somewhere between 2 and 3 million tonnes of material (McGowan and Soderholm 2012). Because much of the material is from central Australia, particularly the Lake Eyre Basin, this dust can be a significant source of salt as well as iron (Box, Radhi & Box 2010). Once deposited in the landscape, the salt can obviously be re-mobilised in the catchment during rain events, while the iron could stimulate algal blooms (Cropp et al. 2013).

Dust storms and climate change

Currently, large dust storms that have the potential to impact on water quality in the southern MDB are infrequent, and generally occur in *El-Niño* dominated years (Speer 2013) . Climate change is expected to increase the periods of low precipitation, which would lead to the loss of vegetation and the subsequent drying out of soils, both of which contribute to the formation of large dust storms. However, climate modelling suggests that the weather patterns responsible for large dust storms will contract poleward as the climate changes. Therefore, it has been suggested that overall the frequency of large dust storms that can impact on the eastern Australian states will remain about the same (Speer 2013).

Dust storms and water quality in the southern Murray–Darling Basin

Although large dust storms can mobilise substantial amounts of material, they are quite rare in the eastern states of Australia, occurring on a roughly decadal frequency (Speer 2013). Furthermore, much of the mobilised material is more likely to deposit in marine systems than on land (Cropp et al. 2013). Therefore, although dust storms have the potential to increase salinity, turbidity and algal blooms, other erosional processes (such as extreme rain events, or erosion following bush fires) are likely to have a greater impact on water quality into the future.

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Blue-green algae

Return to matrix

Blue-green algae (cyanobacteria) are a group of photosynthetic prokaryotes and, as such, aren't true algae, but rather bacteria. Blue-green algae pose a significant water quality risk when present in large numbers, for a number of reasons. Firstly, a large number of blue-green algal species produce compounds that are toxic to higher organisms. Some of these toxins (microcystins, nodularian and cylindrospormopsin) affect the liver (hepatatoxins) while other toxins (Anatoxin-a, Anatoxin-a(S) and saxitoxins) affect nerve synapses or axons (neurototoxins). The physiological purpose of these toxins remains unknown, although it has been suggested that this family of microcystins may be associated with iron transport within the cells (Klein, Baldwin & Silvester 2013). There have been numerous documented cases of the death of domestic animals from drinking water containing blue-green algae), but no human deaths that can be unequivocally linked to exposure to toxins (Falconer 2001). However, in 1980, 150 children and adults from Palm Island, near Townsville in North Queensland, were hospitalised with hepatoenteritis (Byth 1980), with the cases coinciding with the dosing of the local reservoir to remove a bloom, of what was subsequently identified as the blue-green algae *Cylindrospermopsis* sp. (Hawkins et al. 1985). *Cylindrospermospis* sp. has been shown to produce a toxin (cylindrospermopsis) that induces hepatoenteritis in animals (Hawkins et al. 1985). In 1988, there was an outbreak of gastroenteritis in Brazil's Bahia state that resulted in 88 of the 2000 people affected dying (Teixeira et al. 1993). The outbreak coincided with a bloom of the blue-green algae *Anabaena* sp. and *Microcystin* sp. that formed on the nearby Itaparicia Dam following its commissioning, and the number of reported cases fell dramatically after the bloom was killed by dosing the dam with copper sulfate (Teixeira et al. 1993). In 1996, 50 dialysis patients died in Caruaru, Brazil from liver failure after being exposed to the blue-green algal toxin microcystin; microsystin was detected in the liver tissue of the patients (Jochimsen 1998).

All blue-green algae produce a compound called beta-methylamino-L-alanine (BMAA (Cox, Banack & Murch 2003; Cox et al. 2005)). BMAA is a simple amino acid that has been shown to induce changes in the brain structure of monkeys that are similar to human patients with Alzheimer's disease (Cox et al. 2016). Therefore, it is possible, but as yet not proven, that prolonged exposure to blue-green algae, and hence BMAA, can lead to an increased risk of neuron diseases.

Contact exposure of blue-green algae has been linked to a variety of other, non-lethal symptoms in humans, such as allergic reactions, headaches and dizziness (see Stewart et al. 2006), putatively caused by a group of compounds known as lipopolysaccharides; although a systematic review of the literature couldn't definitively link exposure to cyanobacterial lipopolysaccharides with these symptoms (Stewart et al. 2006). Pilotto et al. (2004) have shown that direct exposure to both intact and lysed blue-green algal cells caused skin rashes in about 20% of healthy volunteers.

In addition to the effects on humans and domestic animals, blue-green algal blooms can also have adverse effects for the environment. In Africa, pansteatitis, also known as yellow fat disease, has caused the death of substantial numbers of crocodiles. It has been suggested the crocodiles are believed to have contracted the disease from eating fish that also had died from the disease (Huchzermeyer et al. 2013). There are two possible mechanisms for fish to contract pansteatitis. Blue-green algae lack essential fatty acids necessary for the growth of higher organisms. Zooplankton feeding on blue-green algae are depleted in poly-unsaturated fatty acids and sterols. If zooplankton are the major component (directly or indirectly) of a fish's diet, then they will be depleted in those fatty acids, which can lead to pansteatitis. An alternate explanation is that pansteatitis in the fish is caused by exposure to pro-oxidants that cause lipid peroxidation. One likely pro-oxidant is microcystin (see e.g. Amado et al. 2011), the toxin produced by a number of different blue-green algae. Irrespective of the pathway, it is highly likely that if blue-green algae become the predominant primary producer in lowland rivers in Australia, it will compromise the health of native Australian fish.

Blue-green algae and climate change

A number of authors (Paerl and Huisman 2008; Paerl and Huisman 2009; Elliott 2012; O'Neil et al. 2012) have suggested that climate change will favour both the incidence and global expansion of blue-green algal blooms.

A number of factors associated with climate change positively impact on blue-green algae. Firstly, blue-green algae grow better at higher temperatures compared with other algal species (Paerl and Huisman 2009), with maximum growth rates found at water temperatures greater than 25 ^{o}C (Elliott 2012). A laboratory study also suggests that warmer temperature favours toxin-producing over non-producing strains of *Microcystis* sp. (Davis et al. 2009). Temperature also indirectly favours blue-green algae. Many blue-green algal species contain gas vesicles that allow them to regulate their position in the water column. Warmer temperatures, particularly in summer, can lead to the onset of [stratification] in non-flowing water bodies. Because of competition for limiting nutrients in the surface mixed layer (SML), this layer can become depleted in nutrients. Conversely, the non-mixed bottom water can be nutrient-rich (Baldwin et al. 2008). The nutrients are derived both from the sediments, as well as dead biota raining down from the SML. Because the non-mixed bottom water is generally not in the photic zone, there is less competition for limiting nutrients. As blue-green algae with gas vesicles can regulate their position in the water column, they have the potential to access the nutrients in the non-mixed bottom waters (Elliott 2012). Many blue-green algae can also fix nitrogen, so aren't susceptible to low levels of this nutrient in their environment. Gas vesicles also confer an advantage in high light climates. If algal cells receive too much light, it can inhibit their ability to photosynthesise by overloading their photosystems (Oliver et al. 2003). By regulating their position in the water column, blue-green algae have the potential to maximise their ability to photosynthesise under a variety of light climates.

Blue-green algal blooms in the Murray River

Generally speaking, until the Millennium Drought, blue-green algal blooms in the Murray River were not common (Gutteridge, Hasking & Davey 1974; Croome et al. 1975; Sullivan, Saunders & Welsh 1988; Walker and Hillman 1982; Water ECOscience 2002). As an example, Figure 1 shows the blue-green algal cell counts at Heywoods Bridge on the Murray River directly downstream of Lake Hume. Substantial concentrations of blue-green algae cells were found at Heywoods Bridge in 1981–82 and from about 2003–11. Prior to 2016, the occurrence of a blue-green algal blooms at Heywoods Bridge, and other sites downstream of Lake Hume (Baldwin et al. 2010; Al-Tebrineh et al. 2012; Bowling et al. 2013; Bowling et al. 2016) only occurred when the water in Lake Hume fell below 10% capacity (Figure 1).

In 2006/2007, the Murray–Darling Basin Authority (MDBA) commissioned a detailed assessment of Lake Hume in order to determine the drivers of blue-green algal blooms in Lake Hume, and by inference, the Murray River. The study showed that conceptually, during the period of extreme drawdown, the reservoir can be thought of as consisting of three separate but inter-related parcels of water (Figure 2) (Baldwin et al. 2008). The warm surface mixed layer (SML) during summer is about 6 metres deep and covers the whole lake; the depth of the SML is consistent with earlier thermodynamic studies of the lake (Sherman 2005). Water inflows from the Mitta Mitta River originally from Lake Dartmouth in summer can be up to 15 ^{o}C colder than the SML. Therefore, flows from the Mitta Mitta River undershoot the SML in the lake and flow along the bottom of the reservoir to the dam wall; without substantial interaction with the SML. Inflows from the Murray River are more sporadic and coincide with flows diverted through the Snowy Mountain Scheme. The temperature of flows into the lake from the Murray River are similar to the to that of the SML at

Bethanga and the flows appear to move within the SML towards the dam wall having little influence on the deeper waters at Bethanga, which remain cool. These Murray River inflows are usually insufficient to promote total mixing of the surface and bottom waters, resulting in essentially stagnant water under the SML at Bethanga (and probably throughout the deeper sections of the Murray River arm of the reservoir. The stratification of the lake in the Murray River arm results in the build-up of nutrients in the bottom waters. Periodically, this stratification can be broken down by strong winds or storms, moving nutrient rich water to the surface, where it can be used to fuel algal growth. When the levels of water are sufficiently low (less than 10%), any algae growing in the surface layer can be entrained in outflows from the lake.

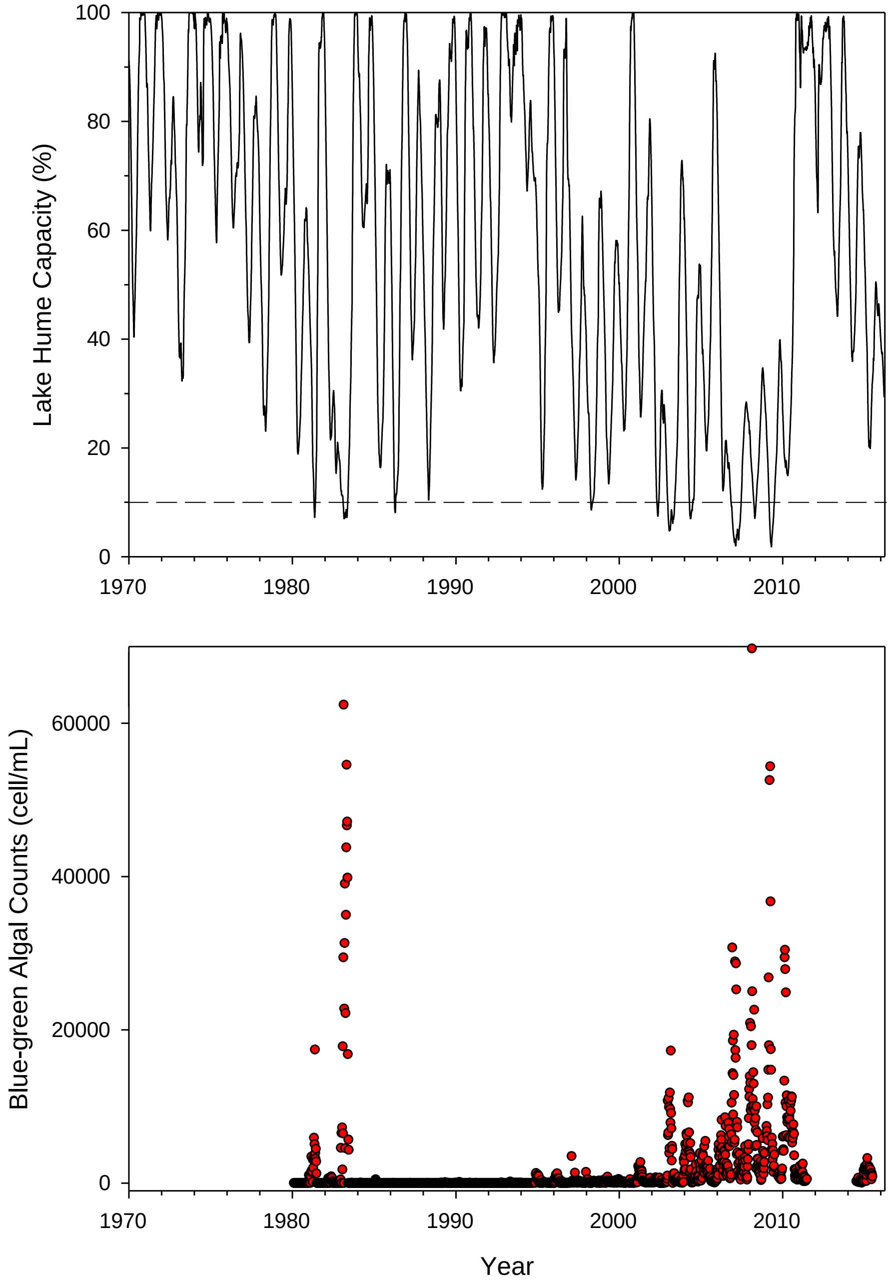

Figure 1. Capacity of Lake Hume (%) from 1970 to 2015 (top panel) and blue-green algal cell density (cells/mL) directly downstream of the lake at Heywoods Bridge (data supplied by the MDBA).

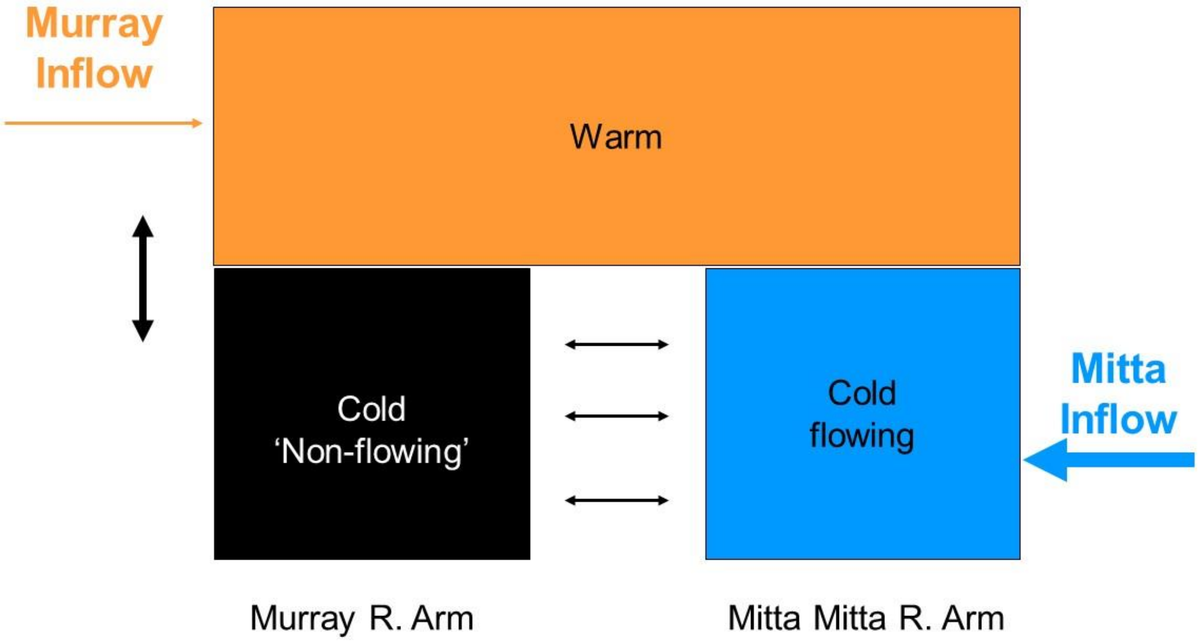

Figure 2. Conceptual model of the hydrology of Lake Hume during summer inflows (after Baldwin et al. 2008).

During summer, outflows from the lake are regulated through two low level outlets: the hydro-electric outlet (168.5 –162.5 m above Australian Height Datum; AHD) and the Irrigation outlet (161.5–157.5 m above AHD). At 10% capacity, the water level in the lake is approximately at 170.7 m above AHD. Once entrained, the algal community structure in the river downstream of Lake Hume remains similar to that which occurred in the lake — at least as far downstream as Corowa (Baldwin et al. 2010). Once the bloom reaches Lake Mulwala, it can potentially seed a further bloom in that water body, which then propagates further down the river (Al-Tebrineh et al. 2012).

While this model helped predict the blue-green algal blooms in the Murray River in 2009 and 2010, it proved a poor predictor of the algal bloom that occurred in the Murray River in the late summer and autumn of 2016 (Figure 3). The dominant algae in the bloom was *Chrysosporum* (previously *Aphanizomenon*) *ovalisporum* (C. Merrick pers. comm.; L. Bowling pers. comm.). During the 2016 bloom it was first detected at Union Bridge, Albury in low numbers on the 4/2/2016, suggesting the bloom formed in Lake Hume sometime in the second half of January 2016. Sampling frequency prior to the full onset of the bloom was irregular to non-existent, particularly in Lake Hume, so it is not possible to unequivocally say whether the bloom started only in Lake Hume or in both Lake Hume and Lake Mulwala.

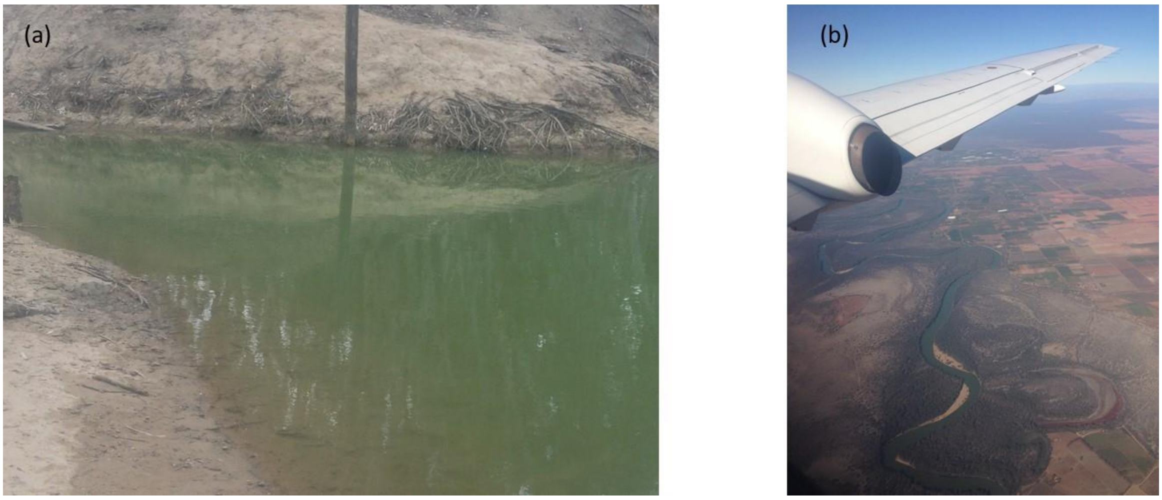

Figure 3. Blue-green algae in the Murray River (a) at Barmah township on 21 April 2016 and (b) upstream of Mildura on 5 May 2016.

C. ovalisporum is a small nitrogen fixing blue-green algae (Gophen et al. 1999). Some strains of this blue-green algae can produce the hepatatoxin, cylindrospermopsin (Banker et al. 1997). *C. ovalisporum* has previously been reported in the Murray River system, but generally in very low numbers; there hadn't been a reported bloom of this species in the Murray River since monitoring began in 1978 (Figure 4; MDBA Algal Data Base). Previous blue-green algal blooms in the Murray River have been dominated by *Anabaena* sp. and *Microcystis* sp. (Baldwin et al. 2010; Al-Tebrineh et al. 2012). Blooms of *Chrysosporum* sp. have been reported in Israel Lake Kinneret (Pollingher et al. 1998; Gophen et al. 1999), Lebanon (Slim et al. 2014; Fadel et al. 2015), Spain (Quesada et al. 2006), Italy (Messineo et al. 2010) and Greece (Gkelis et al. 2005). Blooms generally occur in these locations when water temperatures exceed 26 °C (Gophen et al. 1999). It is not uncommon for surface water temperatures in Lake Hume to exceed 26 °C in summer and early autumn. Why the current bloom was dominated by *C. ovalisporum* and not the other blue-green algae remains unknown, and probably will not be able to be determined with the current available data. It is of note that the sudden occurrence of *C. ovalisporum* in Lake Kinneret was associated with unusually warm water temperatures, low wind inputs and a low N:P ratio in the water (Gophen et al. 1999). The average minimum air temperatures in January, February, March and April 2016 were 1.9, 0.8, 3.4 and 1.1 °C, respectively above the long-term averages; while the maximum daily air temperatures for the same period were 0.3, 1.5 and 3.2 and 3.5 °C, respectively above the long-term averages (Figure 5). However, it is not simply temperature. The maximum and minimum average daily temperature for January, February March and April for the last five years, with a few exceptions, exceeded the long-term average temperature for that month. Solar irradiation was also above the long-term average for most of February and March 2016 (Figure 6). The high irradiation levels coincided with a period of lack of storm activity (Figure 7).

Apart from the unusual species, what sets the current bloom apart from other blue-green algal blooms that have occurred in the Murray River is that, until this bloom all previous blue-green algal blooms below Lake Hume occurred when water levels in the Lake fell below 10% capacity (see above). In the current bloom, Lake Hume was at 37% capacity on 1 February 2016. Under normal conditions at that level, the SML (which will contain the algae) is sufficiently above the outlets from the lake, that the surface water is not released from the lake; at 37% capacity the lake surface is at 179.5 m above AHD.

In the absence of data, I can only speculate on the drivers of the current bloom. It is possible that dam operation may have played a role in bloom formation and propagation in the dam. Blue-green algal bloom dynamics in Lake Hume is strongly influenced by periodic breakdown of stratification (Baldwin et al, 2008). When the dam level is low (ca. 10%) the breakdown of stratification can be driven by storm events and multiple breakdowns can occur in one summer. Storm activity was not particularly strong in 2016, but there were multiple instance of cold-water inflows into the dam from the Murray River system (Figure 8). It is possible that these pulses of cold water into the dam set up a zone of upwelling near Bowna when the cold water inflows hit the easterly extent of the dam. If this is indeed what happened, then the upwelling zone would have brought nutrients into the SML multiple times between January and the beginning of April. The blue-green algal bloom developed in February 2016, which also coincided with a substantial decline in flows in the Mitta Mitta River (Figure 9). The role of this cessation of flow on hydrodynamics within the dam are unknown.

Our ability to assign causes to the current bloom has been hampered by the loss of ongoing monitoring of depth profiles of temperature, oxygen, nutrients and algal dynamics in Lake Hume. Implementation of a properly designed monitoring program (e.g. Baldwin and Boulding 2008) would allow for a better understanding of the factors contributing to blue-green algal blooms in the lake, which opens up the potential to design strategies and/or infrastructure to mitigate the effects of what, is most likely, an ongoing problem.

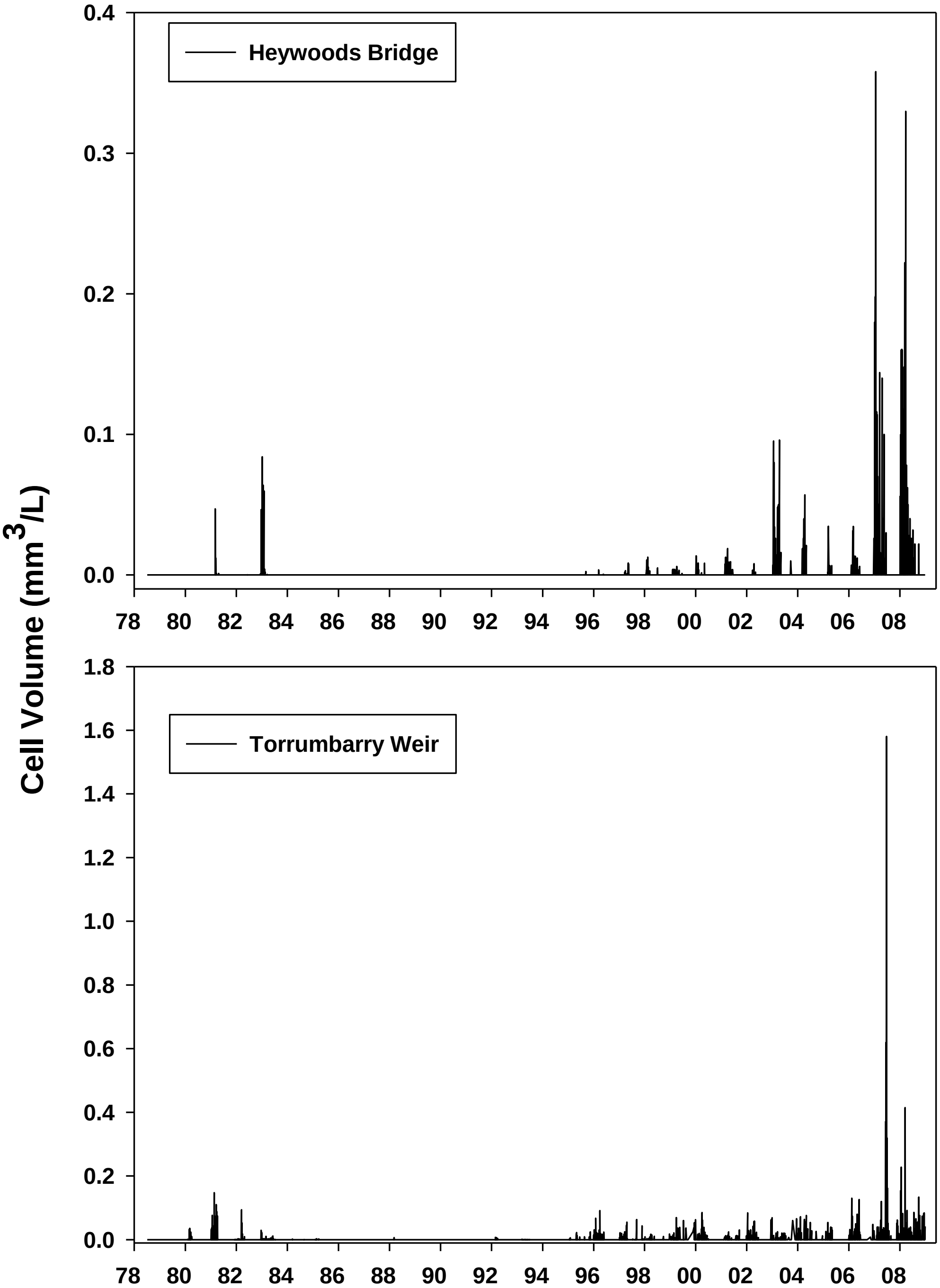

Figure 4. Cell volumes of *Chrysosporum* (previously *Aphanizomenon*) *ovalisporum* at Heywoods Bridge and Torrumbarry Weir for the period 1978 to 2009 (data supplied by the MDBA).

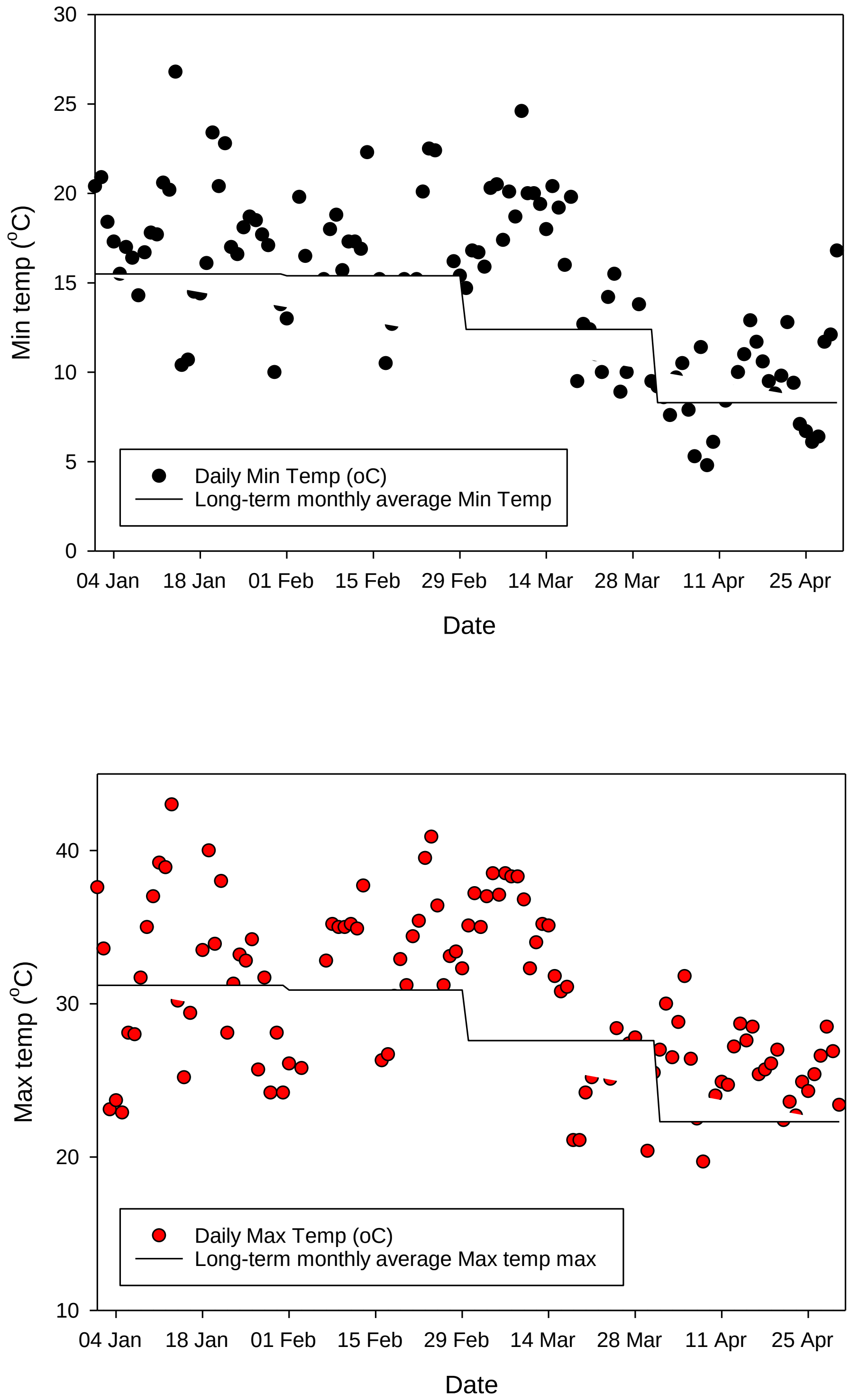

Figure 5. Minimum (upper panel) and maximum (lower panel) air temperature at Albury Airport from January to May 2016 against the long-term monthly average temperature (data from the Bureau of Meteorology).

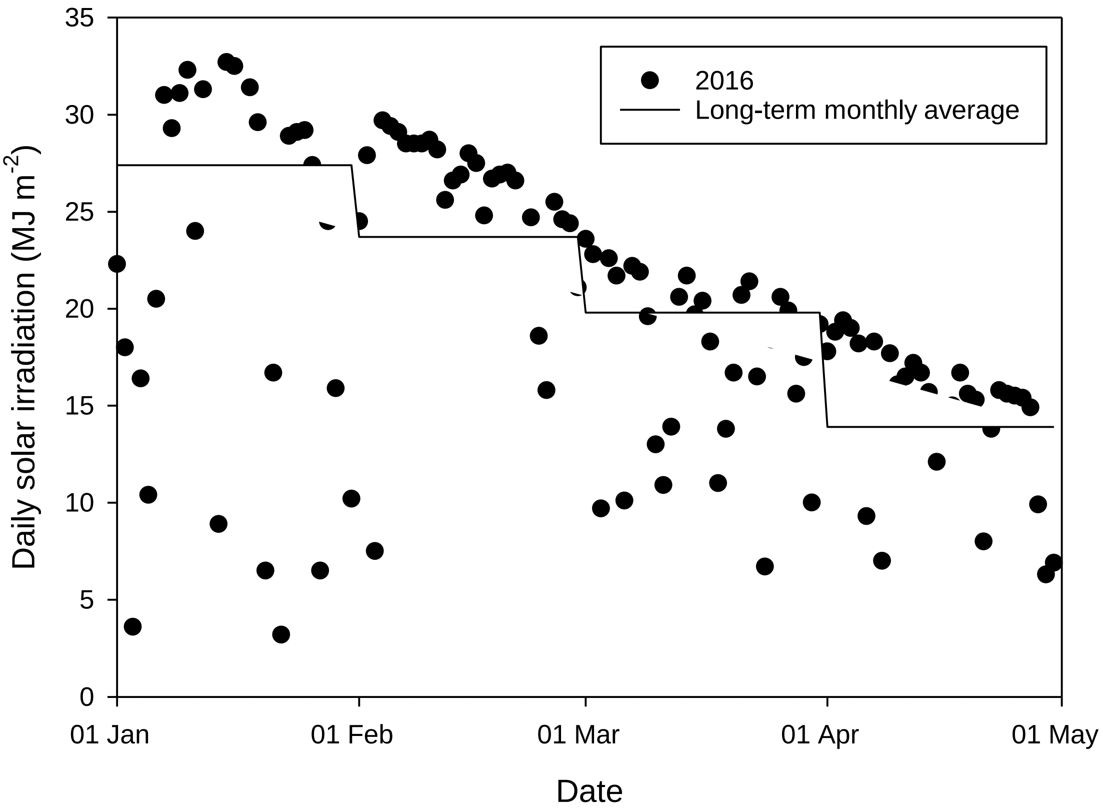

Figure 6. Daily solar irradiance levels at Albury Airport. Data from January to May, 2016 compared to the long-term monthly solar irradiance (data from the Bureau of Meteorology).

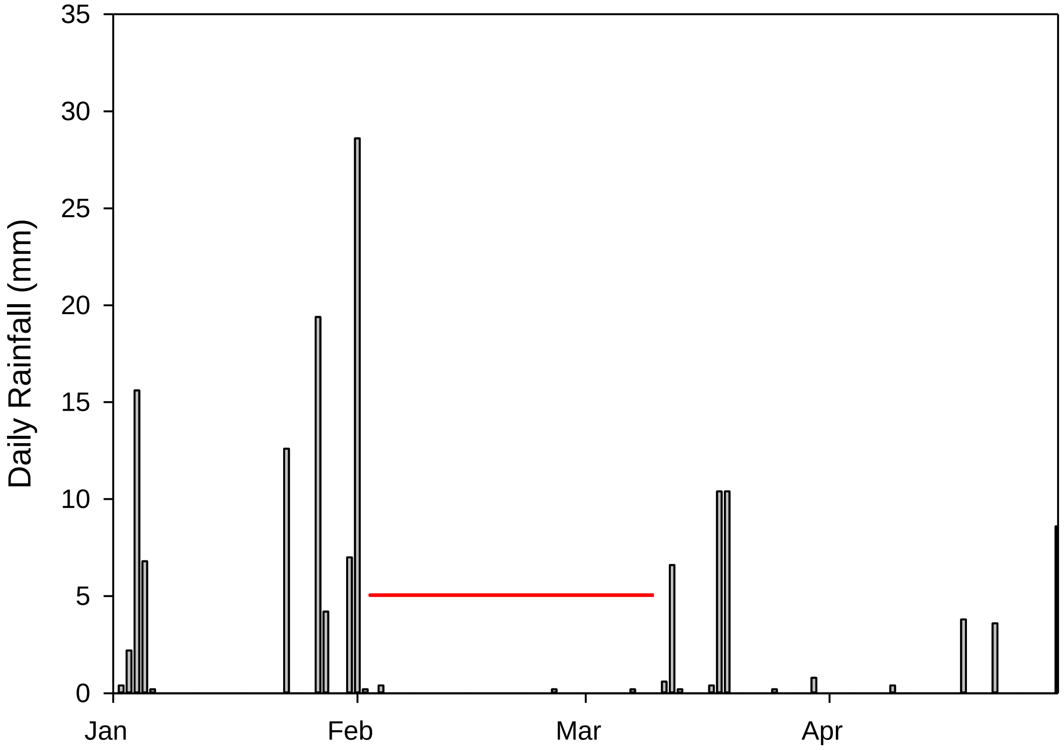

Figure 7. Daily rainfall measured at Albury Airport for the period January –April 2016 (data from the Bureau of Meteorology). The red bar shows the period during February and March 2016 with above-average solar irradiance.

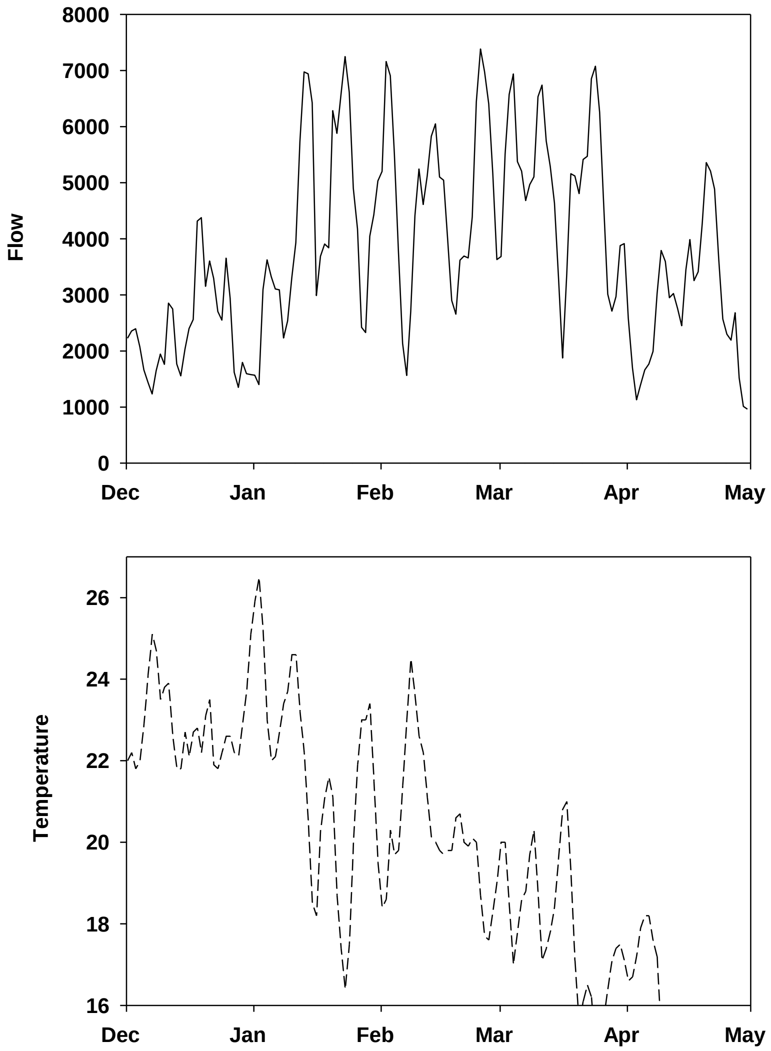

Figure 8: River flows (upper panel) and water temperature (bottom panel) at Jingellic, upstream of Lake Hume on the Murray River. Cold water temperatures are associated with increases in flow (data supplied by the MDBA).

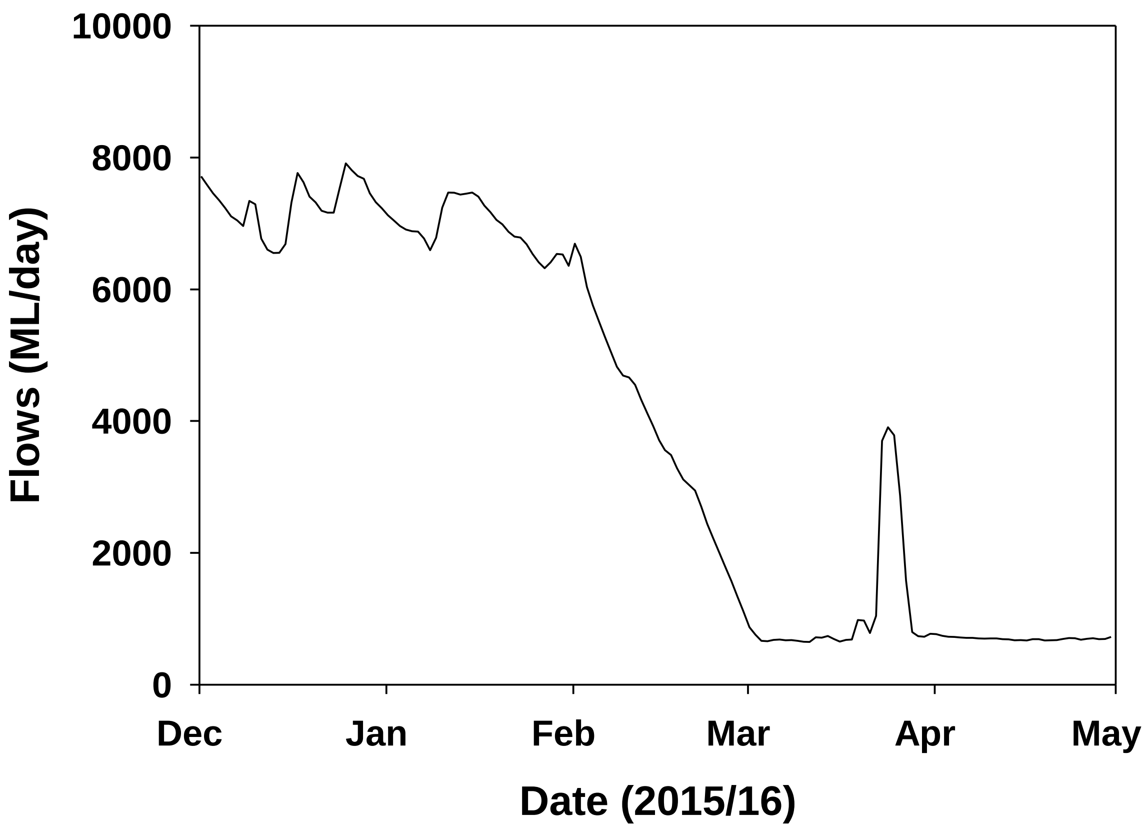

Figure 9 River flows at Tallandoon upstream of Lake Hume on the Mitta Mitta River. Inflows began to fall in early Feburary

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Return to matrix

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Acidification

Return to matrix

pH is a measure of the concentration of H+ ions in water, and is assessed on a log scale from 0 (highly acidic) to 14 (highly basic). Because it is on a log scale, every change of one pH unit represents a ten-fold increase or decrease in H^+ concentrations. Typically the pH of freshwater systems varies from about 6.5 (slightly acidic) to 8 (slightly acidic; ANZEC and ARMCANZ 2000). If the pH remains outside of these levels, it can lead to ecological damage, with the most severe ecological effects associated with acidification (Tibby et al. 2003). Lower pH levels alter the mobility and toxicity of elements, particularly metals (Tibby et al. 2003).

Over the last 40 years, there has been an observed decline in pH in streams and rivers draining the upper catchment of the southern connected MDB; with long-term declines of up to 1 pH unit being reported (Tibby et al. 2003). The reasons for this decline are still unknown, but could reflect a return to pre-European settlement levels. A study of the diatom record in the sediments of a billabong on the Goulburn River upstream of the confluence with the King River showed that, prior to European settlement, the pH in the billabong was between 6.5–6.7. Following European settlement, the pH increased by more than 0.5 pH units (Tibby et al. 2003). The increase in pH was attributed to the transport of base cations (potassium, calcium and magnesium) into the wetland from the upper catchment through soil erosion. Declining levels of base cation export from the upper catchment could lead to acidification (Tibby et al. 2003).

Acid sulfate soils

Acidification in middle and lower reaches of the southern connected MDB have been linked to the exposure and oxidation of acid sulfate soils (sulfidic sediments; The Environment Protection and Heritage Council and the Natural Resources Management Ministerial Council 2011; Murray Darling Basin Ministerial Council 2011). 'Acid sulfate soils' is the generic name given to soils or sediments that either contain sulfide minerals or contained sulfides minerals that have subsequently oxidised. Until very recently, it was believed that acid sulfate soils didn't occur in inland environments. However, acid sulfate soils have now been identified throughout inland Australia. Acid sulfate soils form under waterlogged conditions. Under anaerobic conditions, a group of bacteria can convert sulfate, which is associated with salinisation, to sulfide. The sulfide reacts with metals, particularly iron, to form sulfide minerals. When the mineral sulfides are exposed to oxygen, they can oxidise, producing acid (The Environment Protection and Heritage Council and the Natural Resources Management Ministerial Council 2011; Murray Darling Basin Ministerial Council 2011). pH levels of less than 2 have been recorded in wetlands associated with the lower Murray River (McCarthy et al. 2006). During the Millennium Drought, acidification of Lake Albert and Lake Alexandrina was only averted by significant management intervention.

Acidification and climate change

Predicted changes to the climate will influence pH. pH in the MDB is, in many instances, regulated by bicarbonate-carbonate equilibria, which are, in turn moderated by atmospheric CO_2 concentrations. Atmospheric CO_2 is predicted to increase into the future, with increased CO_2 levels up to 1000 ppm by the end of this century. I have estimated that a doubling of CO_2 concentrations will lower riverine pH in the southern connected MDB by about 0.3 pH units. Fire incidence is also predicted to increase in the MDB (Williams, Karoly &Tapper 2001; Pitman, Narisma & McAneney 2007). Initial runoff from fire can result in a drop in receiving water pH by up to 0.75 pH units (Dahm et al. 2015), but the effect is transitory. Finally, while the risk of salinisation is not predicted to markedly increase under a changing climate, large stores of acid sulfate soils will remain in numerous river channels, wetlands and lakes in the mid- and lower-reaches of the Murray River. More frequent drying events will result in more frequent oxidation of this material, producing acidification events.

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Mobilisation of sediments

Return to matrix

Sediment mobilisation is an important issue in the MDB. It has been estimated that up to 28.7 million tonnes of sediment enters the river network of the MDB annually, with most sediment coming from hillslope delivery (8.4 million tonnes/year) and gully erosion (4.6 million tonnes/year; de Rose, Prosser & Weisse 2004). Delivery of such substantial amounts of material can pose some significant water quality issues, including increased turbidity and mobilisation of contaminants.

Turbidity: Turbidity is a measure of how much suspended material there is in a waterbody and is directly related to the erosional processes in the catchment and resuspension processes in the river network. High levels of turbidity can impact on the function of aquatic ecosystems. Turbidity affects light penetration in water bodies, so can directly affect photosynthetic organisms, particularly submerged macrophytes. Because many blue-green algae can adjust their position in the water column, turbid waterbodies can be dominated by blue-green algae. Highly turbid systems are often characterised by a low diversity of aquatic habitat types through sediment smothering.

Contaminants: Sediment mobilisation can create a number of significant water quality issues. Nutrients associated with mobilised sediments can increase the risk of eutrophication in receiving waters. It has been suggested that about 50% of the sediment-bound phosphorus in the MDB comes from either gully or hillslope erosion from the upper catchment (Land and Water Australia 2002). A legacy of historical gold-mining in the upper catchments in the southern MDB has been the accumulation of mining spoils and tailings, which contain elevated levels of arsenic and which may be contaminated with mercury. Elevated levels of mercury and/or arsenic in Lake Hume, Lake Mulwala and other receiving water may be a direct result of these mining activities (Bycroft et al. 1982; Churchill, Metheral & Suter 2004; Baldwin and Howitt 2007; Baldwin et al. 2008.)

As noted elsewhere, increased fire intensity and/or increased storm intensity will most likely lead to increased erosion in the upper catchments of the southern MDB, and so it is expected that sediment mobilisation will increase into the future.

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Salinisation

Return to matrix

Salinisation of Australia's inland waterways is arguably the largest water quality threat to these systems. The drivers of salinisation are well documented and include the clearing of native vegetation, a shift from trees to shallow-rooted annual crops and irrigation agriculture. All of these processes have mobilised salt and raised the depth of shallow (naturally saline) aquifers. The impact of salt mobilisation is well documented in Australia (Nielsen et al. 2003), and includes lethal and sub-lethal impacts on native flora and fauna as well as changes to biogeochemical processes, including the development of acid sulfate soils in inland waterways.

Climate change and salinity

The impact of climate change on salinisation is hard to predict. Increased bushfires will mobilise cations from the upper catchment, which will increase the load of salt to lowland reaches (Reale et al. 2015). Ephemerality will also impact on salinity levels. The salinity within a residual pool will depend on the water quality in the pool at the time of disconnection, the volume of the residual pool (concentration x volume = load) and whether or not the pool interacts with groundwater. In the absence of groundwater inputs, the salinity in the pool will increase during the drying phase principally through evapoconcentration. For example, Hamilton et al. (2005) showed through the use of both conservative ion tracer concentration and oxygen and hydrogen isotope fractionation that evaporative water loss controlled the water levels in 15 residual pools in Cooper Creek, in Central Australia. In the absence of surface flows, the residual pools could fall to 10% of their bank-full volumes in 6–23 months, depending on the morphology of the pool (Hamilton et al. 2005). If the pool does interact with groundwater, the salinity in the pool will, at some point, reflect the salinity in the groundwater (Costelloe, Western & Irvine 2007). This of course assumes that groundwater levels haven't declined because of decreased precipitation — either directly because of lower discharge, or indirectly because of greater pressure on groundwater resources for consumptive purposes. Cartwright and Sullivan (2008) suggest that there isn't sufficient data available to predict how groundwater levels will change in the face of climate change, but they do suggest that increased demand for groundwater for consumptive purposes may have a much greater effect than direct climate change impacts such as lower precipitation.

Internationally, storm surges coupled with rising sea levels are seen as a threat to water quality in low lying areas through salinisation of freshwater environments. In the southern connected MDB the only area at risk is the Coorong and Lower Lakes. It is uncertain at this point to what extent the current barrages can protect the Lower Lakes from salinisation, but it is of note that during the Millennium Drought, in the absence of storm surges, the median salinity at Goolwa increased ten-fold from about 1200 mg/L (≈ 1700 EC) to about 12000 mg/L (≈ 17000 EC) through evaporation and influence of groundwater (Mosely et al. 2012).

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Eutrophication

Return to matrix

Eutrophication refers to an excess of nutrients, particularly nitrogen and phosphorus, in aquatic ecosystems. Excess nutrients can lead to excessive growth of plant and algal material in waterways, including blue-green algae[2]. Dissolved oxygen concentrations in eutrophied systems can go from supersaturation to near hypoxia to in a 24 hour period as the algae shift from photosynthesis during the day (which produces oxygen) to respiration at night (which consumes oxygen; e.g. see McDonnel and Kountnz 1966). The effects of hypoxia are discussed in another essay. Supersaturation of oxygen is also problematical. It can cause gas-bubble disease in fish (Bouck 1980), which can result in death.

There is a risk of hypoxia when the algae or macrophytes in the eutrophied system die. The sudden influx of bioavailable carbon is consumed by bacteria; they also consume oxygen. If they consume oxygen faster than the rate that it can be supplied from the atmosphere, then hypoxia can result, a similar process to that observed in hypoxic blackwater events. Hypoxia following the death of algae or aquatic plants, is more common in still rather than flowing waters.

Eutrophication and climate change

The changing climate will more than likely enhance the rate of eutrophication in aquatic ecosystems in the southern MDB.

Transport: One of the key sources of nutrients in waterbodies is from terrestrial sources in the catchment. Nutrients associated with particulate matter are moved during erosional episodes. Fires, loss of vegetation during drought and increased storm intensity all have the potential to accelerate erosion and hence particulate nutrient transport. Fires have also been shown to enhance the rate of delivery of dissolved nutrients from the catchment to receiving bodies (e.g. Sheridan et al. 2007).

Internal loading of phosphorus: Internal loading refers to the cycling of nutrients within a waterbody. Numerous studies have shown that even if nutrient supplies are stopped from outside sources, there can be enough nutrients stored in the system, usually in the sediments, to maintain algal blooms into the future. This is particularly true for phosphorus. Large stores of phosphorus can be found in the fine sediments of most standing waterbodies, usually bound to iron or aluminum minerals (e.g. Baldwin 1996). Under aerobic conditions, these minerals are quite stable and little, if any of the phosphorus in the sediments is released to the overlying water column (e.g. Mitchell and Baldwin 1998). However, under anaerobic conditions, certain bacteria can reduce the iron minerals, liberating the phosphorus to the overlying water column. In still water, anoxic conditions usually form following thermal stratification. Predicted increases in temperature will more than likely create more instances of thermal stratification and stronger thermoclines, meaning that stratification will persist for longer.

Sediment drying: Increased temperatures coupled with reduced precipitation can lead to the periodic drying out of sediments. On re-wetting, these sediments can produce a pulse of nutrients, usually nitrogen (Wilson and Baldwin 2008; Lado-Monserrat et al. 2014), but occasionally phosphorus (Attygall 2015). This is discussed in more detail in the essay on ephemerality.

Nitrogen fixation: As discussed on the essay on blue-green algae, changes to the climate, particularly increases in temperature will favour blue-green algae, rather than other types of algae, into the future. Many blue-green algae, including the species responsible for the mega-bloom in 2016, can fix nitrogen from the atmosphere, therefore reducing their reliance on external source of nitrogen.

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[2] Because of their toxic properties, and the likelihood that blue-green algal blooms will continue into the future, they are dealt with in their own essay.

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Heavy metals and metalloids [Return to matrix]

Metals and metalloids in the southern connected Murray-Darling Basin

Gold Mining: A survey of metals and metalloids in the sediments of Lake Hume (Baldwin et al. 2008) showed that sediments from some sites exceeded the action level of the Australian Interim Sediment Quality Guidelines for Arsenic (20 mg/kg) (Figure 1). High levels of arsenic have also been recorded in groundwater from north-eastern Victoria, especially near Wangaratta (Jacobs 2014). Elevated mercury levels have been reported in sediments and surface waters in Reddy Creek, a tributary of the Ovens River (Churchill, Metheral & Suter 2004) as well as sediments in Lake Mulwala, on the Murray River immediately downstream of its confluence with the Ovens River (Baldwin and Howitt 2007). The elevated levels of arsenic and mercury, in particular, are undoubtedly related to historical gold mining in the upper catchment. Most of the gold deposits found in north-eastern Victoria are associated with arseno-pyrite minerals (Williams et al. 2009). Both mining and smelting processes re-distributed the arsenic in the landscape. Mine spoils and tailings, in particular, are known to contain elevated arsenic levels (McArthur et al. 2000; EPA Victoria 2009). Mercury was used in the extraction of gold from its ore (Bycroft et al. 1982) and was widely used in the Victorian Goldfields (Smyth 1869). Levels of mercury in mine tailings in Victoria have been reported to be as high as 130 mg/Kg — over 1000 times higher than the action level of the Australian Interim Sediment Quality Guidelines. It has been estimated that 1.8 billion m^3 of material has been worked and reworked through gold mining in the colony of Victoria between 1850 and 1900, with most of that activity in the MDB (P. Davies, Department of Archeology and History, La Trobe University pers. comm.). While not all of this material will necessarily be contaminated, it does suggest a potential problem. A recently funded Australian Research Council Project is investigating (ARC 2015) how historical gold mining has affected river systems in the MDB, including the impact of arsenic and mercury contamination on riverine ecology.

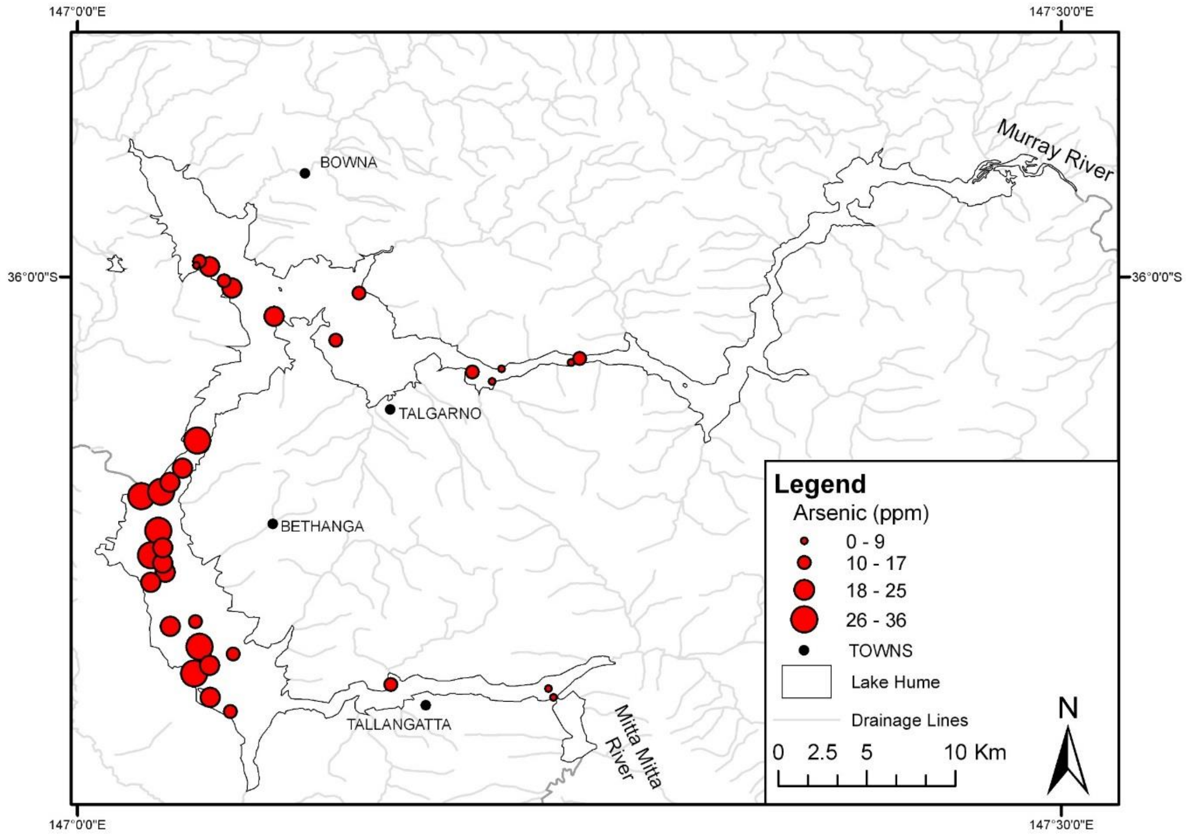

Fig 1. Arsenic levels in the sediments of Lake Hume

Metal associated with inland acid sulfate soils: In the last decade, a large number of wetlands associated with the Murray River have been found to contain acid sulfate soils (sulfidic sediments, Murray–Darling Basin Ministerial Council 2011). When exposed to the air and re-wetted, they can cause acidification, which in turn can solubilise metals from the sediment. For example, using 24 hour incubation experiments, Simpson et al. (2010) showed that the concentrations of aluminum copper and zinc in the water overlying previously dried sediments were 100 times higher than the Australian water Quality Guidelines. Similarly, Fraser et al. (2010) reported that the total metal concentration in the water of Bottle Bend Lagoon, an acidified wetland near Buronga on the Murray River, was approximately 8 g/L.

Heavy metals and climate change

Contaminated mining spoils in the upper catchment are susceptible to erosional processes, and therefore climate change impacts that increase erosion will increase the likelihood that the contaminated spoils will move into waterways where they have the potential to be accumulated into the biota (Williams et al. 2009). Increased risk of fire, increased intensity of storm events and dust storms can all contribute to sediment mobilisation. Therefore, in a changing climate we would expect to see increased concentrations of arsenic and mercury, particularly in aquatic environments where sediments accumulate (e.g. in weir pools and in dams).

Sulfidic sediments are common in the Edward-Wakool region as well as in riverine wetlands of the Murray River (including the lower lakes) downstream of Euston. The risk of mobilisation of metals from these sites only occurs when the sediments are exposed to oxygen — i.e. following drying

(Environment Protection and Heritage Council, 2011). With the likelihood of lower precipitation into the future, including increased ephemerality, it would be expected that the incidence of mobilisation of metals from impacted sites will increase in the future.

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Hypoxia

Return to matrix

Hypoxia refers to the condition where the water column contains low levels of dissolved oxygen. Once the dissolved oxygen levels falls below about 4 mg O_2/L, organisms that rely solely on dissolved oxygen in the water column for respiration, particularly fish, become stressed; levels below about 3 mg O_2/L can lead to death (Kerr, Baldwin & Whitworth 2013 and references therein).

In the southern connected MDB, there have been a number of fish deaths associated with hypoxia over the last 20 years. There were two different drivers of these hypoxic events: disturbance of stratified pools and hypoxic blackwater events

Stratification

Thermal stratification is the differential heating of surface water, which absorbs heat and so becomes less dense and floats on the cooler bottom waters. The extent of stratification is primarily determined by the relative input of thermal energy (causing stratification) and turbulent kinetic energy (breaking down stratification; Bormans et al. 1997). As the surface layer heats up, the increased buoyancy of the surface water resists downwards mixing, and increased kinetic energy is needed to overcome the buoyancy gradient. Stratification can be disrupted from the air-water interface by wind energy and from the sediment-water interface by turbulence generated by water flow over the stream bed. Stratification can generate a number of water quality problems including de-oxygenation of the lower section (hypolimnion) of the water column (Figure 1). Oxygen consumption by microbes in the sediment removes oxygen in the bottom waters (hypolimnion). Because the bottom waters aren’t mixing with surface waters, re-oxygenation from the air doesn’t occur, leading to hypoxia in the bottom waters.

Although persistent thermal stratification in rivers is not a typical occurrence because discharge normally provides sufficient kinetic energy to prevent its establishment, it can occur within deep pools during low and no-flow periods associated with ephemerality (e.g. Turner and Erskine 2005 and references therein).

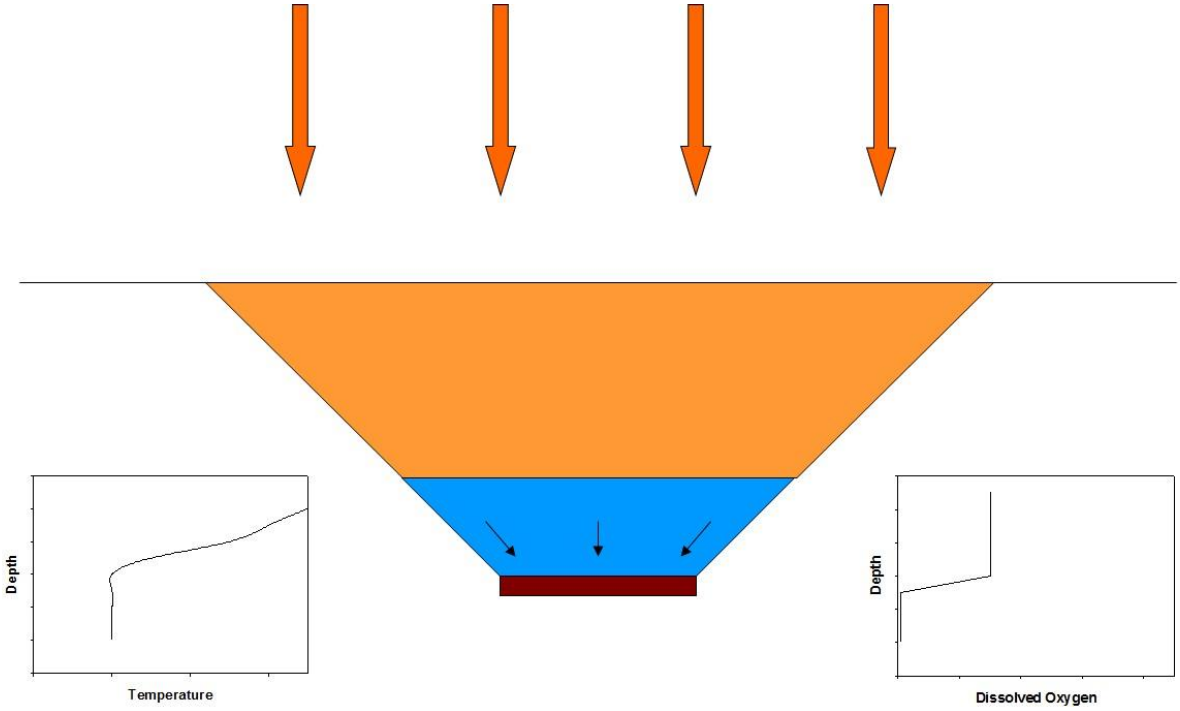

Figure 1. Onset of stratification followed by de-oxygenation in a standing waterbody. De-oxygenation occurs because of the demand for oxygen by micro-organisms in the sediment.

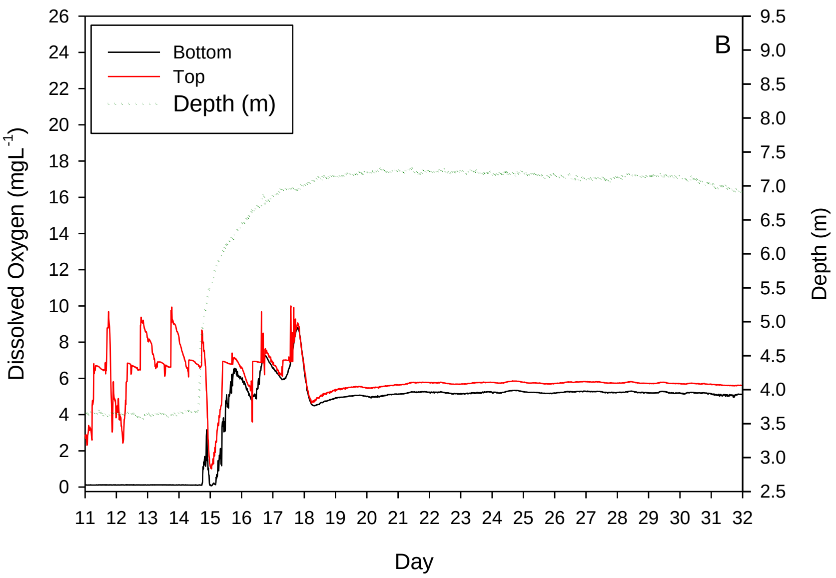

Figure 2. Dissolved oxygen levels in an isolated pool in the Darling River prior to and immediately after a small inflow. The inflow created complete mixing, resulting in a period of hypoxia in the pool (from Wallace et al. 2009).

Problems arise when a mixing event occurs. Mixing events can be caused by strong winds or incoming flows. Figure 2 shows the dissolved oxygen levels in a pool on the lower Darling River (from Wallace et al. 2008). Prior to an inflow on day 15, the water column in the pool was stratified. Surface water contained between about 4 and 8 mg O_2/L (varying diurnally because of the effect of algae photosynthesis and respiration and changes in water temperature). The bottom water was anoxic. On day 14, a small flow reached the pool which mixed the surface water with the bottom water. For a period of 5 hours on day 15, the dissolved oxygen concentration was below 2 mg/L, the level below which fish die. While no fish deaths were recorded during that event, earlier flows following periods of cease-to-flow have been recorded. In February 2004, between 1000 and 5000 large Murray cod (*Maccullochella peelii peelii*) died in the lower Darling River between Menindee and Pooncarie during a single event (Ellis and Meredith 2004). Immediately prior to the deaths, the lower Darling had ceased flowing due to an active decision to cease downstream releases to secure storage for later supply. During the no-flow period, the river was reduced to a series of disconnected remnant pools. In February 2004, a release of water from Menindee Lakes was made in order to deliver water to downstream users. Reports from landholders indicated that large Murray cod died several days after the front of water released from the Menindee Lakes passed down the lower Darling River.

Hypoxic blackwater

Over the course of the Millennium Drought in south-eastern Australia, there were a number of instances when regulated flows over a floodplain or down a dry river channel resulted in substantial fish kills (Baldwin and Wallace 2009). The most dramatic of which was the hypoxic blackwater event of 2010–11, which affected about 2000 km of the Murray River and its tributaries and persisted for 6 months (Whitworth, Baldwin & Kerr 2012). These fish deaths have mostly been attributed to hypoxic blackwater events. Blackwater events, characterised by high levels of DOC in the water column, are a natural part of the ecology of lowland river systems and have been reported since before periods of significant river regulation. During a flood, carbon compounds are leached from litter (leaves, bark, twigs etc.) and other organic materials (in-channel saplings, macrophytes etc.) on the floodplain or in dry flood runners/creek channels in much the same way that tea is leached from tea leaves. The carbon and nutrients released from the litter are believed to play an important role in the functioning of riverine-floodplain systems (Robertson *et al.* 1999; Francis and Sheldon 2002). For example, Junk et al. (1989) has shown that in the Amazon River basin, much of the production that occurs in the river channel is based on carbon derived from the floodplain. Therefore, it is important that periodic flooding occurs in river systems to allow the transfer of carbon from the floodplain to the river. The timing of high flow events have been altered as a consequence of river regulation. However, as we show below, appropriate timing of these inundation events is crucial if degradation of water quality is to be minimised.

Blackwater events can markedly change water quality. Microbes can immediately use about one-third of the carbon leached from leaf litter (Baldwin 1999). As these micro-organisms consume the dissolved carbon, they also consume oxygen from the water column — often at a faster rate than the oxygen can be replenished. Therefore, blackwater plumes often have very low levels of dissolved oxygen. The lack of dissolved oxygen can cause the death of fish and other aquatic animals in the plume. It is generally accepted that fish will begin to suffer at dissolved oxygen concentrations less than 5 mgL^{-1}, and that few species can tolerate conditions of less than 3 mgL^{-1} for prolonged periods (Gilligan, Vey & Asmus 2009). In addition, the carbon leached from native plants, especially red gum leaves, can be directly toxic to native fish at elevated concentrations (McMaster and Bond 2008).

A number of factors are critical in determining whether or not a blackwater event will result in native fish and crustacean deaths. The two most important factors are carbon standing stocks and water temperature.

Carbon Stocks: The amount of carbon leached during a wetting event will depend primarily on the amount of litter and other organic material accumulated since the last flood and the rate of litterfall during the period of inundation.

A number of factors determine the amount of litter present at the time of floodplain inundation or dry channel flooding. Factors contributing to litter standing stock include the average annual deposition of various plant species, the density of plant species in the area, seasonal effects on growth and standing stocks, the time since last flood and transportation (by wind, for example). Available carbon in the standing stock may be depleted by grazing, microbial degradation, leaching of soluble organics, and chemical and photochemical degradation.

During an inundation event, processes that increase DOC in the water column include primary production, leaching from floodplain and in-channel organic material (including macrophytes, bark, leaves, grass and wood) and leaching from soil. The amount of DOC released varies between litter components, with higher amounts leaching from leaves than from bark and twigs (O'Connell et al. 2000). Biological, chemical and photochemical degradation can remove DOC from the water column, consuming oxygen in the process.

Water temperature: Water temperature is critical in determining whether or not a blackwater event will lead to native fish and crustacean deaths. Dissolved oxygen and temperature are linked, so that as water temperature increases, the amount of oxygen that can be dissolved in the water decreases. At 20 °C, the solubility of oxygen is approximately 9 mg L^{-1}, while at 35 °C, water can contain only 7 mg O_2 L^{-1}. Most importantly, the rate at which carbon is respired (and dissolved oxygen is consumed) by bacteria is temperature-dependent. For every 10 degree increase in water temperature, the rate of oxygen depletion approximately doubles (Whitworth, Baldwin & Kerr 2014). Primary controls for the concentration of dissolved oxygen in rivers are (i) the physico-chemical processes of gas-exchange (oxygen diffusion into the water column across the air-water interface and water-gas saturation relationships); (ii) oxygen production from photosynthesis; (iii) sediment and pelagic oxygen demand (e.g. heterotrophic and autotrophic respiration, nitrification); and (iv) chemical oxygen demand (Kelly et al. 2007). When oxygen demand becomes the dominant process, dissolved oxygen concentrations are depleted and hypoxic and or anoxic conditions will become established. Inundation of floodplains and dry channels during periods of high temperature therefore increases the risk of hypoxia.

Hypoxia and climate change

I predict that the incidence of hypoxic events will increase into the future; at least in the medium term.

Warmer temperatures coupled with increases in ephemerality will result in the fragmentation of river continuum into discrete pools. Warmer temperatures will lead to stratification in these pools, which can subsequently lead to hypoxia if pools turn over.

Hypoxic blackwater events are also more likely to occur in the future. Increases in atmospheric CO_2 will promote the production of greater amounts of leaf litter, while extended periods of drought will allow for a greater period of time for litter to accumulate on the floodplain. Periodic storms will generate flooding which will then mobilise the carbon, potentially creating hypoxic conditions (depending on the timing of the storm events). However, in the longer term, a lack of periodic flooding may see an overall decline in floodplain vegetation. Lower carbon stocks on the floodplain could ultimately result in less carbon mobilisation from the floodplain during flooding events.

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Return to matrix

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