Climate change and freshwater ecosystems in Oceania: an assessment of vulnerability and adaptation opportunities

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Human-forced climate change significantly threatens the world's freshwater ecosystems, through projected changes to rainfall, temperature and sea level. We examined the threats and adaptation opportunities to climate change in a diverse selection of rivers and wetlands from Oceania (Australia, New Zealand and Pacific Islands). We found common themes, but also important regional differences. In regulated floodplain rivers in dry regions (i.e. Australia), reduced flooding projected with climate change is a veneer on current losses, but impacts ramp up by 2070. Increasing drought threatens biota as the time between floods extends. Current measures addressing water allocations and dam management can be extended to adapt to climate change, with water buy-back and environmental flows critical. Freshwater wetlands along coastal Oceania are threatened by elevated salinity as sea level rises, potentially mitigated by levee banks. In mountainous regions of New Zealand, the biodiversity of largely pristine glacial and snow melt rivers is threatened by temperature increases, particularly endemic species. Australian snow melt rivers face similar problems, compounding impacts of hydro-electric schemes. Translocation of species and control of invasive species are the main adaptations. Changes to flow regime and rising water temperatures and sea levels are the main threats of climate change on freshwater ecosystems. Besides lowering emissions, reducing impacts of water consumption and protecting or restoring connectivity and refugia are key adaptations for conservation of freshwater ecosystems. Despite these clear imperatives, policy and management has been slow to respond, even in developed regions with significant resources to tackle such complex issues.

Key words: flow regime, floodplain, wetland, river regulation, estuary, snow melt rivers, glaciers, dryland rivers

INTRODUCTION

HUMAN-forced climate change has and will continue to affect the world's freshwater ecosystems, with social and ecological consequences (Smit and Pilifosova 2001; Milly *et al*. 2005; Palmer *et al.* 2008). Global projections indicate there will be fewer wetter basins while the proportion of drier basins will rise over time (Milly *et al.* 2005). Accelerated climate change is pervasive throughout Oceania (Australia, New Zealand and Pacific Islands), changing rainfall, temperature and increasing sea levels (Table 1), all of which affect freshwater ecosystems (IPCC 2007a). Freshwater ecosystems at low elevations on coast lines are vulnerable to sea level rise; glacial and snow-melt streams are particularly vulnerable to temperature changes and; wetlands in dry areas will experience increased drying with increasing temperatures and sometimes reduced run-off. Anthropogenic climate change impacts will compound significant anthropogenic threats in freshwater ecosystems (Vörösmarty *et al.* 2000; MEA 2005; Kingsford *et al.* 2009; Kingsford 2011), causing further biodiversity loss (Lake *et al.* 2000; Heino *et al.* 2009). For example, regulation and water abstraction has degraded Australian rivers and wetlands (Kingsford 2000) and Pacific islands (Parrish *et al.* 1978; Kingsford *et al.* 2009). Land-use in New Zealand and temperate Australia has destroyed many floodplain and wetland biotic communities (Jones *et al.* 1995; Keith 2004; Kingsford *et al.* 2009). The few freshwater protected areas are mostly in elevated, nutrient poor terrestrial landscapes that are not managed for freshwater values (Kingsford *et al.* 2004), with little protected land in the Pacific Islands (Kingsford *et al.* 2009).

Altered climate change will change the key drivers of freshwater ecosystems: flow regime (Poff *et al.* 1997; Bunn and Arthington 2002), water temperature (Caissie 2006) and water quality (Finlayson *et al.* 2009). Also, low lying coastal wetlands will be significantly affected by sea level rise. These factors will change regionally and for different types of ecosystems (Aldous *et al.* 2011; Poff *et al.* 2002).

In alpine regions, where snow melt drives high spring flows, increased temperatures will reduce snow and shift the timing of melt (Kingsford 2011). Similarly, increasing temperatures accelerate glacial retreat, altering the unique character of associated streams (Winterbourn *et al.* 2008). Rising temperatures affect all freshwater ecosystems, reducing the resilience of refugia during low flow periods (Magalhães *et al.* 2002) and threatening the survival of species (Pittock and Wratt 2001).

Reduced rainfall and runoff and consequent reductions in flow and flooding combined with increasing temperature will significantly impact

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Table 1. Climate change projections that will impact freshwater ecosystems in Australia, New Zealand and the Pacific islands. Data from Arnell (1999), Jones *et al.* (1999), Lal *et al.* (2002), Hennessey *et al.* (2003), Mullan *et al.* (2005), Wratt *et al.* (2006), Hennessy *et al.* (2007), IPCC (2007b), Ministry for the Environment (2008), BMT-WBM (2010) and Nicholls (2011).

Country/region	Temperature	Rainfall/ runoff	Sea level	Events
Inland Australia	2020: +0.2-1.5°C, 2050: +0.5 to 4.0°C 2080: +0.8 to 8.0°C.	2020: -5 to+5%, 2050: -13 to+13%, 2080: -27 to+27%. 2050- A decrease in runoff in the Basin of around 15-35%	n.a.	2030: up to 20% more droughts with projected increases in the Palmer drought severity index from 2000 to 2046 2070: 80% more droughts (defined as the 1-in-10 year moisture deficit from 1974 to 2003) in south-western Australia.
Northern Territory	2020: +0.1-1.0°C, 2050: +0.3 to 2.7°C 2080: +0.4 to 5.4°C.	2020: -10 to+5%, 2050: -27 to+13%, 2080: -54 to+27%.	2030: 14cm (IPCC emission scenario A1B, 95th percentile) 2070: 70cm (high emissions scenario)	Large flood events (currently 20 days per event) 2030: -12 or + 10 days per event. 2070: 0 or + 9 days per event
Australia alpine	2050: +0.6 to 2.9 °C	2050:-24% to +2.6	n.a.	2050: snow cover decline of 22-85%, earlier snowmelt
New Zealand	2040: 1°C 2090: 2°C relative to 1980-99 (mid-range IPCC A1B scenario	2040: +5% increase in the west 2090: +10% increase in the west : -5% decrease in the east and north	2090: 18 to 59 cm relative to average sea level over 1980-1999 and a potential of 2100- Plausible increase of 50–100 cm and possibly higher	The frequency of extreme rainfall events is expected to increase 7-20%, based on a temperature increase of 1-3°C Drought frequency is predicted to increase from 1 in 20 year frequency to a 5 to 10 year return frequency
Pacific Islands	2050: +0.4 to 1.3°C 2100: +0.6 to 3.5°C	2050: +4.1 to 5.7%	2100: 8–59 cm	Greater intensity of extreme events likely to cause more flooding and drought, as well as more Category 4 and 5 cyclones

inland rivers and wetlands (e.g., Australia, MDBA 2010). In particular, the timing between flood events is critical to floodplain wetlands and extending this period beyond the tolerances of species and ecosystems reduces resilience of aquatic biota and processes (Jenkins and Boulton 2007). Climate change impacts on freshwater macroinvertebrates in floodplain wetlands may be considerable, given limited dispersal routes, changes in water quality (temperature and availability), and the considerable anthropogenic impacts already affecting freshwater systems (Woodward *et al.* 2010). As the driest inhabited continent, Australia is particularly affected by changing climate and impacts on water resources. Since 1950, Australian average annual and seasonal temperatures have increased, leading to more severe droughts (Nicholls and Collins 2006), increasing fire risk (Driscoll *et al*. 2010) and reducing flooding (Steffen *et al*. 2009). Water resource competition is predicted to rise with reductions in rainfall in south-west and inland Australia (Pittock and Wratt 2001, Hennessy *et al*. 2007). Conversely an increased frequency of high-intensity rainfall in northwestern Australia may increase flooding (CSIRO 2007), benefitting wetlands but damaging human communities. Most models predict an increase in extreme events, such as flooding and drought (Bates *et al.* 2008; Prowse and Brook 2011).

Projected sea level rise is a significant threat to coastal wetlands throughout Oceania (Kingsford *et al.* 2009). On Pacific Islands, sea level is projected to rise by 28–43 cm by 2100 (Legra *et al.* 2008) and in the Northern Territory freshwater swamps are already impacted by saltwater intrusion (Mulrennan and Woodroffe 1998). Estuarine wetlands in coastal New Zealand will also suffer severe impacts of climate change (Schallenberg *et al.* 2010). The structure and composition of communities will change as species disperse or disappear (Kingsford *et al.* 2009).

As a result of these changes, novel and unpredictable ecosystems will be created, with different species assemblages and dominant processes (Poff *et al.* 2002). For example, the timing of breeding and migration of birds is changing (Beaumont *et al.* 2006) and waterbird distributions (Chambers *et al.* 2005), potentially reflecting alterations in their foraging and breeding habitat (Larson 1994; Kingsford and Norman 2002). Interactions between CO_2, water supply, grazing and fire regimes will also be influential (Hughes 2003). By 2070, climate change and increased water demand in rivers (237 examined globally) is predicted to result in a loss of 75% of freshwater fish fauna from rivers with reduced discharge (Xenopoulos *et al.* 2005). Increased drying may cause severe loss of carbon and aquatic propagules (Jenkins *et al.* 2009).

The increasing threat of accelerated climate change (Prowse and Brook 2011) means that freshwater dependent socio-ecological systems must either adapt or fail (Palmer *et al.* 2008). Adaptations represent changes in processes, practices, or structures in response to expected or actual climate change (Smit *et al.* 2000). They can be either autonomous (ecological and human) or planned, and may or may not be beneficial (Smit *et al.* 2000, Barnett and O'Neill 2010). Planned adaptations may buffer ecosystems, although such efforts may be challenging. Due to the complexity of social-ecological systems, adaptation measures must be incorporated into management and policy. Due to the difficulty in predicting impacts and consequences (Poff *et al.* 2002), an adaptive management framework can address the impacts of climate change on freshwater ecosystems (Kingsford *et al.* 2011a).

We review the vulnerability of freshwater ecosystems and options for adaptation to these changes in climate in six different case studies from Oceania with the aim of identifying common threats, adaptations and management and policy options. We also examine the effects of current pervasive threats and how these interact with climate change (Kingsford *et al.* 2009). The six case studies were chosen to be representative of freshwater ecosystems in Oceania. First, we assess the challenges for the Macquarie Marshes with its long history of environmental flow management to reduce impacts of regulation (Kingsford *et al.* 2011a). Second, we examine snow melt rivers in Australia, where climate change is projected to significantly alter flow by reducing snowfall and melt (Hennessy *et al.* 2007), amplifying impacts of hydro-electric schemes. We contrast these Australian alpine systems with snow melt and glacial fed streams from New Zealand, where adaptation opportunities are limited to the gradual loss of glaciers and warming of streams (Winterbourn *et al.* 2008). Then, we review how sea level rise will affect estuarine wetlands in coastal New Zealand and freshwater wetlands in the Northern Territory of Australia; altering the extent of tidal influence and salinity and flow regimes (Mulrennan and Woodroffe 1998; Schallenberg *et al.* 2010). Our final case study is from the Pacific Islands (Melanesia, Micronesia and small, developing island states of Polynesia) where sea level rise will significantly constrain adaptation opportunities (Legra *et al.* 2008).

Macquarie Marshes, Australia

The Macquarie Marshes is a forested arid-zone (*sensu* Boulton 2000) wetland on the lower

Macquarie River, New South Wales, Australia in the highly regulated Murray-Darling Basin (Kingsford 2000). It is representative of floodplain wetlands located on other rivers in the Murray-Darling Basin including the Gwydir, Lachlan, Murrumbidgee and Murray Rivers (Kingsford 2000; Kingsford *et al.* 2004). The Murray-Darling Basin is one of Australia's largest and economically vital catchments (MDBA 2010). Many parts of the Basin are arid or semi-arid, typified by isolated ephemeral wetland ecosystems that connect during times of flooding. The healthy functioning of floodplain wetlands, such as the Macquarie Marshes, underpins their ecology and ecosystem services they provide. In particular, many are managed for free-range livestock production (sheep and cattle); landholders depend on the regular flooding for their livelihoods (Kingsford 1999). The Macquarie Marshes and its flooding regime is integral for the survival of colonial waterbirds and other waterbirds, resulting in listing under the Ramsar convention as a wetland of international significance (Kingsford 2000).

The Macquarie Marshes supports up to 74 species of waterbirds (Kingsford and Thomas 1995; Thomas *et al.* 2011), including 44 breeding species (Kingsford and Auld 2005). About 10% of the Macquarie Marshes are legislated as a protected area Nature Reserve with parts of this Nature Reserve and a private property "Wilgara" recognized as a significant wetland under the Ramsar convention since 1986 (Thomas *et al*. 2011). Most other parts of the Marshes are privately owned and used for livestock production. The Macquarie River system contributes about five percent of flows to the Barwon-Darling River system that eventually flows through to the River Murray mouth in South Australia. Renowned as one of Australia's more significant waterbird breeding sites (Kingsford and Johnson 1998; Kingsford and Auld 2005) and supporting between 10 000 and 300 000 waterbirds after floods, the area of the Marshes was estimated at 200 000 ha in the 1990s (Kingsford and Thomas 1995). At this time, there was 72 000 ha of semi-permanent wetland vegetation including lignum *Muehlenbeckii florulenta*, common reed *Phragmites australis*, cumbungi *Typha orientalis*, water couch *Paspalum paspaloides*, and extensive floodplain eucalypts (river red gum, *Eucalyptus camaldulensis,* coolibah, *E. coolabah* and blackbox, *E. largiflorens*) (Paijmans 1981). The Marshes is a biodiversity hotspot with records of 130 species of birds other than waterbirds, 15 species of fish, four species of turtle, 30 species of lizards, 14 species of snakes and 15 species of amphibians (NPWS 1993; DECCW 2010).

The Macquarie River is heavily regulated (70% of the flows in the river) by Burrendong (operational in 1967) and Windamere Dams (operational in 1984) with weirs (Dubbo, Narromine, Gin Gin, Warren and Marebone) providing further flow control in downstream reaches (Kingsford and Auld 2005). Flows are also affected by structures that redirect and capture flooding on the floodplain (Steinfeld and Kingsford 2011). Flows to the Macquarie Marshes now depend on an environmental flow allocation, local flooding, flows from unregulated tributaries and dam spills (Kingsford *et al.* 2011a). Regulation and water abstraction for irrigated agriculture has significantly impacted the flow regime; the median annual modelled flow is currently only 43% of the natural flow, and only 59% of the natural floodplain is inundated (Ren *et al.* 2010). Floodplain inundated with high frequency (i.e., annually) now covers only 5184 ha, less than a quarter of pre-regulation levels (Thomas *et al.* 2011), reducing soil carbon levels, invertebrate biodiversity (Jenkins *et al.* 2009), waterbird communities (Kingsford and Thomas 1995) and vegetation communities, particularly river red gum (Thomas *et al* 2011; Steinfeld and Kingsford 2011). Although biota are adapted to a boom and bust regime, regulation has already exceeded tolerance thresholds in many parts of the system, substantially reducing the extent of the Marshes and shifting the mosaic of flood histories and associated biodiversity and processes (Thomas *et al.* 2011).

Perhaps the most dramatic example of the impact of regulation on biotic thresholds and resilience occurred in the recent decade-long drought, the "Big Dry" (Prowse and Brook 2011), the first since the current levels of abstraction. During the period from 2001-2009, the Marshes experienced unusually long dry periods when the Water Sharing Plan failed to balance water use in the catchment and was ultimately suspended (NWC 2009). Following decades of reduced flooding, waterbird populations plummeted (Kingsford *et al.* 2011b), more than half the wetland vegetation was lost, and what remains is in decline (DECCW 2010; Thomas *et al.* 2011). Responding to the widespread degradation of wetlands last decade the State and Commonwealth Governments invested in water buy-back for wetlands in the Basin, increasing the environmental water allocation from 125 000 ML for the Marshes (Kingsford and Auld 2005) to about 300 000 ML.

In the Murray-Darling Basin, runoff has reduced as rising air temperatures increase evapotranspiration, compounding impacts of many interception structures (e.g., Steinfeld and Kingsford 2011) and land use (i.e., clearing) (CSIRO 2007). A 15% reduction of inflows has occurred with a 1°C rise in average temperature, and these are predicted to fall by 55% if temperatures rises by 2°C by 2060 (Cai and Cowan 2008). By 2030, climate change is predicted to increase the time between floods by 10–24% with floods declining in size by 5–38% (Table 1) (CSIRO 2008). Within the Murray-Darling Basin annual streamflow is likely to fall 10–25% by 2050 and 16–48% by 2100 (CSIRO 2008). These predictions are in the range of those for inflows to the Macquarie Marshes which are projected to decline by 0–15% in 2030 and 0–35% in 2070 (Jones and Page 2001). Plantations will also affect flows (Herron *et al.* 2002), but not to the same extent (CSIRO 2008).

Reduced rainfall, runoff and subsequently decreased flooding with climate change has affected inland wetlands but still only represent a veneer on top of the impacts already experienced due to river regulation (Jenkins *et al.* 2011). Although the biota of arid-zone wetlands are adapted to a boom and bust ecology (Kingsford *et al.* 1999), regulation has already exceeded the limits for many taxa and climate change will exacerbate these impacts. Projections for 2070 are more severe, and with river regulation, may irreversibly alter the character of the Macquarie Marshes. The projected increase in drought intensity is a significant threat (Prowse and Brook 2011), given the sensitivity of wetland biota to increased time since flooding and current impacts of regulation (Kingsford and Thomas 1995; Jenkins and Boulton 2007; CSIRO 2008, Thomas *et al.* 2011). These changes to flow will restrict natural overbank flows leaving many wetlands reliant on targeted environmental flow allocations (Aldous *et al.* 2011). The combination of the arid climate and high water demands (Poff *et al.* 2002) will make these freshwater ecosystems particularly vulnerable to the effects of climate change. Despite this, existing management infrastructure means there is also a broad scope of adaptation strategies (i.e., dam re-operation, re-licensing) that can be adapted to mitigate climate impacts (Pittock and Hartman 2011; Watts *et al.* 2011).

In the Macquarie Marshes, environmental flows are the key mechanism to adapt to climate change and strong institutional arrangements exist via the Environmental Flows Reference Group established under the Water Sharing Plan and water buy-back (Kingsford *et al.* 2011a). The challenge is to develop flow rules that consider future climate change to enable carry-over and release of larger volumes of environmental water. There is an opportunity to develop clear objectives that assist in transparent decision-making for environmental flows aimed at achieving conservation of ecosystems. Given the complexity and scale of the Macquarie Marshes and potentially competing objectives, there is a need for an explicit adaptive management process. This is currently under development allowing incorporation of climate adaptation options (Kingsford *et al.* 2011a).

Snow melt rivers in Australia

Snow melt creates seasonally variable thermal and hydrological regimes that promote a unique diversity of specialist aquatic fauna adapted to cues provided by thawing in spring (sensu etc.). These include thermal and hydrological cues for spawning, migration, and metamorphosis, coinciding with increased availability of resources from high flows (Yarnell *et al.* 2010). Globally, climate change is expected to alter the timing and volume of snow melt flows through earlier snow melt, reduced snow depth and cover, and increased evapotranspiration (Adam *et al.* 2009).

There is a relatively small area of land affected by snow cover in Australia (Pickering and Armstrong 2003). Associated Australian snow melt rivers are mostly in the steep and rugged Snowy Mountains region of New South Wales, and North Eastern Victoria. The upper parts of the mountains have a relatively high annual rainfall (1400–1800mm; Morton *et al.* 2010), mostly stored as snow during winter. During spring and early summer, accumulated snow above ~1500m asl melts into streams and drains either south into the Snowy River or north and west to the Murray and Murrumbidgee Rivers. Perhaps due to their remote and rugged location, the ecology of Australian snow melt systems is poorly known (Bevitt *et al.* 1999), particularly the dependence of endemic biota on flow and temperature. Mountain galaxias *Galaxias olidus* are widely distributed throughout Australia and are the only fish found above the snowline during winter (Lintermans 2009). Although this species was once common across south-eastern Victoria, they are now restricted to the upper reaches, not yet invaded by brown *Salmo trutta* and rainbow trout *Oncorhynchus mykiss* (Raadik and Kuiter 2002). The two-spined blackfish *Gadopsis bispinosus* lives only in cool upland or montane streams that drain to the upper Murray River (Lintermans 2009). Several frog species are endemic to alpine and subalpine regions including the alpine tree frog *Litoria verreauxialpina*, the southern corroboree frog *Pseudophryne corroboree*, northern corroboree frog *Pseudophryne pengelleyi* and baw baw frog *Philoria frosti;* (Osborne *et al.* 1999). Many macroinvertebrate species are also highly endemic in the Australian alps. Two mayfly nymphs *Coloburiscoides* spp. have only been recorded in upland streams about Mt. Kosciuszko and the Bogong High Plains (Suter and McGuffie 2007). Several stoneflies *Leptoperla curvata* and *L. cacuminis* and mayflies

Tasmanophlebia lacuscoerlei and *Ameletoides lacusalbinae* are also only found in the Snowy Mountains region (Campbell *et al.* 1986).

Flows from many of these alpine systems were significantly altered by river regulation. The Snowy Mountains Hydroelectric Scheme (SMHS) completed in the 1950s and 1960s diverted 99% of flows (~1 000 GL) from the southern flowing Snowy River to the western flowing rivers, generating hydroelectricity and providing water to the inland irrigation industry (Ghassemi and White 2007). The SMHS is an extensive network of large and small impoundments and diversion structures that enables water to be shunted among reservoirs, passing through various hydropower stations before eventually flowing west via the Murray and Murrumbidgee valleys (NoW 2010). The resulting altered hydrology and lost longitudinal connectivity has severely impacted on aquatic communities in rivers affected by the SMHS (Pendlebury *et al.* 1996; Erskine *et al.* 1999). Some environmental flows are returning to the Snowy and some of the montane rivers within the Scheme (NoW 2010).

The river system and its dependent biota, upstream of impoundments, remains largely unaffected but will be affected by climate change, particularly biota dependent on timing and quantity of snow melt. Endemic biota in snow melt systems are particularly vulnerable to climate change (Brown *et al.* 2007). Climate modelling predicts precipitation rates will change by between +2.6 and -24% by 2050 while air temperature is predicted to increase by between 0.6 and 2.9°C (values relative to 1990; Table 1; Hennessey *et al.* 2007). These changes will increase water temperature and reduce flow but also change the timing of flows through reduced area and duration of snow cover. Snow depth has already decreased by 40% in the Snowy Mountains since 1962 (Nicholls 2005) with further declines of 22-85%, predicted for 2050 (Table 1; Hennessey *et al.* 2003). Snow melt across four rivers in the Snowy Mountains now occurs a month earlier than in the 1950s (Reinfelds, I, unpubl. data). This pattern is consistent with observation in the mid altitude snow melts rivers of Northern Western USA (Stewart 2009). The duration of snow cover is further predicted to reduce by 15–100 days and snow depth by 10–99% by 2050 (Hennessey *et al.* 2007). Climate change affects the distribution and phenology of alpine aquatic fauna elsewhere (e.g., Briers *et al.* 2004; Harper and Peckarsky 2006) but little information exists for Australian species, particularly in alpine regions (Williams and Russell 2009).

Australian macroinvertebrate species that favour cold, fast flowing water are most likely to decline in abundance and contract in distribution with climate change (Chessman 2009). These effects mostly occur through changes in temperature which governs species' distributions by lethal effects, such as extreme heat or cold, or sub-lethal effects such as impaired growth or reproduction (e.g., Briers *et al.* 2004). The driving importance of the thermal regime (Caissie 2006) may shift the viable range of distribution of stenotherms upward, increasing the vulnerability of species already facing extinction. Increases in ambient temperature and reduced rainfall are predicted to increase frequency and intensity of bushfires (Hennessey *et al.* 2007), affecting sedimentation and erosion (Suter and McGuffie 2007). For example, in 2002-2003, wildfires throughout the Snowy Mountains increased inputs of sediment and ash to the Snowy River (Russell *et al.* 2008), and the Alpine rivers of the Snowy Mountains, markedly changing aquatic invertebrate communities and threatening the extinction of endemic species (Suter and McGuffie 2007).

The potential for Australian snow melt streams and their biota to adapt to long-term climate change is impeded by inaccessibility and few viable options for managing water temperature and/or flow. Below reservoirs, regulation structures like the SMHS offer opportunities to manage climate impacts (Pittock 2009). Flows from reservoirs could be managed to mimic the natural flow regime, protecting rivers against drought and regulation (Poff *et al.* 1997), and unnatural thermal regimes (Olden and Naiman 2010). However, existing impacts on rivers due to regulation (e.g., Bevitt *et al.* 1999) and aggressive invasive species (e.g., trout, Raadik and Kuiter 2002) probably outweigh the comparatively minor short term impacts from climate change. Moreover, flows from reservoirs are usually already of low water quality (Sherman 2000). Already, ski resort operators use snow-making technology as an adaptation to reductions in snow cover but these are relatively small scale and temporarily effective (Hennessey *et al.* 2008), with a minor effect on the hydrology of streams. Any adaptation options for snowmelt ecosystems in Australia are limited by current lack of knowledge of the relationship of temperature and flow with aquatic organisms.

Glacial and alpine river systems in New Zealand

A substantial part of the South Island of New Zealand is covered by the mountain chain of the Southern Alps, extending about 700 km north-south (40–47°S) and 100 km east-west. Much of this area is steep, with gradients of 0.4 or more, and located above the tree line (~1400 m at the northern end of the chain and ~1000 m at its southern end). Above the tree line, subalpine shrubs, tussock grasses and bare scree slopes extend to permanent snow and ice. Annual precipitation in this sub-alpine and alpine zone is high, ranging from 1500–2000 mm in the relatively dry eastern foothills to 5000–12000 mm in the western part of the mountain chain, where the weather patterns are dominated by wet westerly winds from the nearby Tasman Sea. The summertime snowline descends to1600 m in the west and 2200 m in the east (Mark and Dickinson 1997). As a consequence of the high precipitation and the steep terrain, the alpine and sub-alpine zone contains numerous cold fast-flowing streams and small rivers, which possess a high water quality based on their physicochemistry and ecology (Winterbourn 1997). Because the Southern Alps also contain over 3 000 glaciers ranging in area from about 1 to 10 000 ha (2001 data; Chinn, 2001; 2004), many sub-alpine and alpine streams are glacier-fed, and snow melt during spring and summer contributes a considerable part of the annual discharge in the non-glacier-fed streams.

Due to its long isolation from other land masses, New Zealand possesses a high percentage of endemic animals and plant species, compared to most other parts of the world (see e.g., Daugherty *et al.* 1993; Wallis and Trewick 2009). This high degree of endemism has been well-documented for large animals such as birds (Lee *et al.* 2010) or fish (Hickey *et al.* 2009), and also for plants including many alpine species (Dawson and Lucas 1996; Peat and Patrick 1996). By comparison, the aquatic fauna of high-altitude streams, especially its taxonomy and degree of endemism, is far less well known. A few studies exist that allow projections regarding the potential effects of climate change (Table 1) on sub-alpine and alpine stream communities. For example, four species of the mayfly genus *Deleatidium* are confined to the upper limits of lotic freshwaters on the altitudinal zone from the tree line to the snow line in the Southern Alps and their neighbouring ranges (Hitchings 2009). All four species are adapted to cold, fast-flowing streams and rivers and the frequently severe conditions imposed by an unstable terrain and high winds. There is also a fifth cold-water specialist mayfly species, in the same genus, found at five stream sites in Mt Aspiring National Park, including three glacier-fed sites (Winterbourn *et al.* 2008). A rise in water temperatures associated with global warming would likely result in a southward retreat of these alpine specialists with climate change (Winterbourn *et al.* 2008; Hitchings 2009). A projected 3°C warming (Ryan and Ryan 2006) would displace the regression of annual degree days on latitude southward by 670 km, correspondingly reducing glaciation and warming of alpine streams and resulting in extinction of the five alpine mayfly species (Winterbourn *et al.* 2008; Hitchings 2009).

A similar concern is held for specialist stonefly populations. Their larval stages inhabit high-altitude streams where there is a high degree of endemism in the Fiordland region of the Southern Alps (Peat and Patrick 1996), with eleven endemic species, four occurring predominantly above the tree line. Because of the generally high degree of endemism in New Zealand, further aquatic invertebrate groups common in subalpine and alpine streams such as *Trichoptera* (see Peat and Patrick 1996) and net-winged midges (Blephariceridae) probably also contain several more endemic species adapted to cold water temperatures. The net-winged midges are particularly well known for their preference of cold, fast-flowing and highly turbulent streams (see e.g., Frutiger 2002; Frutiger and Buergisser 2002), and their larvae can be highly abundant in alpine streams of the Southern Alps, such as in Arthur's Pass National Park (C.D. Matthaei, unpubl. data). Such endemism in the Southern Alps of New Zealand (Hitchings 2009) may have resulted from repeated periods of glacial advance and retreats which has similarly produced high endemism rates in the cold-adapted aquatic fauna from Western Europe (Ward 1994). Most of these endemic aquatic species are vulnerable to extinction due to climate-change induced loss of extreme cold-water habitats (Winterbourn *et al.* 2008; Hitchings 2009) even though it is likely that many are yet to be formally identified by science.

It is thought that taxa may adapt to flow and temperature changes in snow melt streams by migrating to high altitudes, but this option is short-term and the steep gradients will give rise to waterfalls, rather than a shift in similar habitats upwards, as temperatures rise. This option will not be possible in glacial systems, which will continue to retreat and vanish as the climate warms as they have in the last 100 years (Chinn 2004). Adaptation options seem limited, apart from establishing captive colonies which are sustained under controlled conditions. Translocation is also suggested for taxa unable to migrate in systems where connectivity is lost (Turak *et al.* 2011). However, in the case of glacial, and to a lesser extent snow melt streams, these habitats are vanishing, leaving limited places to colonize. Clearly, even if these options were possible, they could only be implemented for a few species.

Coastal wetland systems in New Zealand

Estuaries are the interface between freshwater and marine environments, and are therefore highly sensitive to climate-related changes in either environment (Winder *et al.* 2011). The dynamics of opening regimes, patterns of saltwater influx and mixing, and other aspects of water chemistry are influenced by a complex interaction between the quantity and quality of freshwater entering the estuary from upstream, and tidal sea water intrusion through the estuary berm (Schallenberg *et al.* 2003a; Schallenberg *et al.* 2010). These clearly interact to produce habitats and ecosystem processes that support aquatic biodiversity. The Otago coastline forms the southeast corner of the South Island of New Zealand with a rich diversity of estuarine habitats, including permanently open estuaries and many small, intermittently open systems (Lill *et al.* 2011). Tidally influenced freshwater wetlands and shallow lakes are also a feature of the continuously open estuaries, such as the Waihola/Waipori wetlands in the lower Taieri River catchment (Schallenberg *et al.* 2003a; Hunt 2007). Collectively, these ecosystems support a rich biodiversity, with significant populations of native fish including inanga *Galaxias maculatus*, giant kokopu *G. argenteus*, and native birds including fernbird *Bowdleria punctata*, Australasian bittern *Botaurus poiciloptilus*) and New Zealand scaup *Aythyanovae seelandiae* (David *et al.* 2002; Hunt 2007; Kattel and Closs 2007).

The impacts of climate change on these estuarine systems and the species they support will vary depending on their different characteristics and dependencies (i.e., open estuaries and associated tidal freshwater lakes and wetlands or intermittently open estuarine systems) (Schallenberg *et al.* 2010; Lill *et al.* 2011) and the increasing effects of sea level rise in particular. Continuous high freshwater discharge from large rivers, such as the Taieri and Clutha, maintains a permanently open channel to the sea (Schallenberg *et al.* 2003a). Such estuaries usually have predictable patterns of tidal variation and seawater intrusion, interspersed by brief periods of extreme rainfall and drought (Schallenberg *et al.* 2003a). In these systems, biological communities exhibit longitudinal zonation, reflecting prevailing patterns of seawater dilution and mixing at various points along the estuary (Sutherland and Closs 2001). The influence of the tides extends upstream into freshwater wetlands that are only occasionally subjected to saline intrusion during periods of exceptionally low freshwater discharge (Schallenberg *et al.* 2003a). These freshwater wetlands support diverse invertebrate communities and are critical spawning habitat for inanga, the juveniles of which support significant recreational fisheries (Sutherland and Closs 2001; Kattel and Closs 2007). The wetlands are only marginally above sea level (Schallenberg *et al.* 2003a), and particularly vulnerable to sea level rise which will alter their hydrology, generating unpredictable changes in freshwater wetland distribution and structure (Schallenberg and Burns 2003; Schallenberg *et al.* 2010). Increasing the frequency of even minor saline intrusions into these systems reduces their biodiversity (Schallenberg *et al.* 2003b), and threatens the viability of inanga fisheries, given the low salinity required for egg fertilisation during spawning (Hicks *et al.* 2010).

The impacts of climate change (Table 1) on small, intermittently open estuaries are likely to be more complex, given these systems are more numerous and diverse (Lill *et al.* 2011). They exist along a continuum, from ones rarely closed to rarely open (Lill *et al.* 2011). The biodiversity of communities that live within them reflect this gradient of exposure to the marine environment, with invertebrates and fish more characteristic of freshwater habitats dominating estuaries that rarely open to the sea (Lill *et al.* 2011). Rising sea levels and changing patterns of rainfall and evaporation (Table 1) will interact to alter patterns of saline intrusion and freshwater discharge into estuaries in unpredictable ways (Laurance *et al.* 2011). Increased temperatures and evaporation will increase rates of upstream water abstraction for irrigation, reducing freshwater inputs to estuaries that dilute discharge of nutrients and other pollutants (Schallenberg *et al.* 2010; Laurance *et al.* 2011; Winder *et al.* 2011). Eutrophication of intermittently open estuaries due to accumulation of diffuse inputs of nutrients and other pollutants is already significant in many New Zealand estuaries (Schallenberg *et al.* 2010), including those along the Otago coastline (Lill *et al.* 2011).

Estuarine systems, subject to the influence of river and sea, are dynamic and resilient (Lill *et al.* 2011). Indeed, within the past 10,000 years sea levels have been lower and higher than today, and estuarine systems have adapted (Hunt 2007). The extensive cockle beds buried in the sediments of Lake Waihola (now effectively a freshwater system) indicate that this shallow lake functioned as a marine system within the past 4000 years (Hunt 2007). However, present-day patterns of diversity have developed in response to the relatively predictable environmental conditions prevailing in the recent past. Increasing the frequency of extreme events (IPCC 2007a) within these present-day systems will reduce their within-system diversity and threatens their ecological stability and function (Schallenberg *et al.* 2010; Winder *et al.* 2011), at least over temporal scales relevant to human societies (i.e. 2-3 generations, 100 years). The critical challenge for managing the impacts of climate change on large open estuaries is determining whether tidally structured communities can migrate up rivers. Whether this can occur will be dependent on system-specific features such as local hydrology and geomorphology (e.g., the proximity of high gradient landforms or large areas of human settlement upstream of existing estuarine river reaches), the speed of climate-related changes, and the scope and options for human-assisted habitat restoration and creation (Hickford *et al.* 2010; Hickford and Schiel 2011). The management of intermittently open estuarine systems must focus on maintaining a diversity of estuary opening regimes across a regional scale (Lill *et al.* 2011). To some degree, opening regimes can be managed artificially by mechanical breeching of the estuary berm (Lill *et al.* 2011), however protection of freshwater discharge from excessive abstraction is also a critical element in ensuring the sustainability of these systems (Schallenberg *et al.* 2010).

The recently promulgated National Coastal Policy Statement 2010 and National Policy Statement for Freshwater Management 2011 provide the overarching legislative framework for management of estuarine systems in New Zealand (DOC 2010, New Zealand Government 2011). While both policies clearly state sustainable management of ecosystems is a core government priority, the failure to set clear minimum environmental standards and limits within either policy means both statements are open to a broad interpretation of what minimum environmental standards may be acceptable. Climate change is mentioned in five of the 29 coastal policies in the coastal statement, in relation to impacts of rising sea level on coastlines, property and residents. The single policy that addresses biodiversity does not refer to rising sea level nor provide any adaptations to protect coastal wetlands (DOC 2010). Similarly the freshwater management statement makes brief mention of climate change and none of adaptation, requiring regional council's to set objectives for freshwater including quality, levels and environmental flows for "all bodies of fresh water in its region (except ponds and naturally ephemeral water bodies)", having regard to the "reasonable foreseeable impacts of climate change" (New Zealand Government 2011).

Wetlands in the Kakadu region, northern Australia

The wetlands of the Kakadu Region of the Northern Territory are among the more biodiverse and well known in Australia (Morton *et al.* 1993a; Finlayson *et al.* 1997) and the world (Junk *et al.* 2006). A series of rivers flow north to the sea, the Wildman, East Alligator, South Alligator and West Alligator Rivers, particularly during the monsoon season, and they have created a range of different wetlands: inter-tidal mudflats, mangroves, hyper saline flats and freshwater floodplains and streams (Finlayson *et al.* 2006). Tidal influence can extend up to 105 km up the river with a range of 5-6m at the mouth (Woodroffe *et al.* 1989).

Kakadu National Park (19,800 km^2) is part of the region with particularly high biodiversity, some of which is highly dependent on the aquatic ecosystem; it has a quarter of Australia's land mammals (77 species), more than a third of Australia's birds (271 species), 132 reptile species, 27 frog species, about 1600 plant species and more than 10,000 species of invertebrates (SEWPaC 2011). This outstanding biodiversity resulted in listing of the area on the IUCN World Heritage Register (1981) and listing as a wetland of international importance under the Ramsar Convention. The system of wetlands is particularly significant for the high concentrations of waterbirds (Morton *et al.* 1990; Morton *et al.* 1993a,b,c) which undoubtedly reflect their considerable productivity. Most of these birds collect on the large freshwater wetlands on the floodplains of the rivers, where more than one million magpie geese *Anseranas semipalmata* may be found during the dry season (Morton *et al.* 1993a). Given the region's proximity to the sea, some of the wetlands are highly vulnerable to long-term sea level change, exacerbated by invasive species (Bayliss *et al.* 1997; Finlayson *et al.* 1997; Brook and Whitehead 2006). About 20cm sea level rise occurred during the 20th century and projected levels may rise up to one metre by 2100 (Table 1, Brook and Whitehead 2006).

The coastal floodplains of Kakadu are only 3-4 m above sea level (Finlayson *et al.* 1997). Globally, average sea level has risen by 21 cm since 1880 (Church and White 2011) with projections of potentially more than one metre over the next century (Prowse and Brook 2011). The river valleys were probably inundated by the sea about 8 000 years ago with swamps establishing about 6 000 years ago and some plant associations only 2 000 years ago (Chappell and Woodroffe 1985; Woodroffe *et al.* 1993). Freshwater wetlands are separated from dendritic saltwater channels by low levee banks (Finlayson *et al.* 1997) which may be overtopped with saltwater surges or linked by intruding tidal channels (Winn *et al.* 2006), allowing salt water to flow into freshwater wetlands, changing their physico-chemical composition. This may also be exacerbated by reductions in run-off and cyclonic activity which have altered geo-morphology (Winn *et al.* 2006). Salinisation has already changed the state of some freshwater wetlands: freshwater plants have died and salt tolerant ones have invaded the wetlands (Finlayson *et al.* 2009). For example, the tidal part of East Alligator River has invaded inland by four kilometres since the 1950s, increasing the amount of saline mudflats by nine times and killing more than half the paperbark *Melaleucaleuca dendra* forest (Winn *et al.* 2006). In addition, water buffalo *Bubalus bubalis*, introduced in the 1980s, broke through some of the levee banks, allowing salt water intrusion and changing ecosystem states from freshwater to estuarine (Steffen *et al.* 2009). Magpie geese rely predominantly on the wild rice *Oryza* spp. and *Eleocharis* spp., sustained by fresh water flow. Saltwater intrusion is exacerbated in dry monsoonal periods, low-frequency and intensity cyclonic storm cells which are not counteracted by wet season flows (Winn *et al.* 2006). Subsidence of freshwater wetlands, as well as changes to wind direction and strength and rainfall variability may contribute further to the problem.

There are predicted to be increased saltwater intrusions on Kakadu wetlands in the dry season in the South Alligator River catchment in 2030 and 2070, associated with respective sea level rise increases of about 14cm and 70cm corresponding to IPCC emission scenario A1B, 95th percentile and a high emissions scenario respectively (BMT-WBM 2010). This would be through increased tide height and velocity extending tidal channels into the floodplain and increasing overtopping of levees (BMT-WBM 2010). There may be a threshold effect where there is widespread inundation of the coastal plains by tidal flows, resulting from exponential expansion of creek networks (Woodroffe 2007). This is expected to be detrimental to freshwater biodiversity (e.g., magpie geese, pig-nosed turtle *Carettochelys insculpta*, potadramous fish and freshwater crocodiles *Crocodylus johnstoni*). Such impacts are likely to also affect cultural values and tourism, dependent on freshwater biodiversity. Such major changes would certainly substantially change the ecological character of the Kakadu wetlands and possibly its status as a hotspot of biodiversity. This vulnerability may be countered by projected increased rainfall and run-off with climate change (Winn et al. 2006; Steffen et al. 2009).

Opportunities for adaptation for this low lying system are particularly problematic, given how extensive it is and the force of wet seasons flows. Conceivably levees could be erected around the more vulnerable and important freshwater wetlands. There are extensive earthen barrages already built at the mouth of the Mary River, further to the east, which also has significant freshwater wetlands (Steffen *et al.* 2009). Given the relative absence of built environment, there may also be a natural migration of freshwater wetlands upstream although the catchment of steep ranges that do not allow for any wetland development are reasonably close to the sea in this region. Decisions to build protection of these systems will involve value judgments of the preferred condition, given that many of the wetlands were once saline more than 6,000 years ago.

Wetlands of the Pacific Islands

Freshwater resources of the Pacific Islands (see Kingsford *et al.* 2009; Kingsford and Watson 2011) are characterized by coastal wetlands, predominantly estuarine mangrove wetlands and salt marshes. Only Papua New Guinea, Fiji, Vanuatu and Samoa have significant inland freshwater reserves, particularly Papua New Guinea with several million hectares of wetlands (Osborne 1993). While most Pacific Island nations have some coastal wetlands, the small size of islands and often low-lying coral and limestone-based nature of the land masses of many Pacific Islands limits freshwater resources and aquatic ecosystems. Only six of the nearly 2 000 areas designated under the Ramsar convention as wetlands of international importance are on Pacific Islands.

While island ecosystems are, by nature, vulnerable to disturbances, many coastal wetlands, including estuaries, have been heavily modified and intensively developed over the past few decades (Kingsford *et al.* 2009), significantly increasing their vulnerability to climate change. With the exception of Papua New Guinea, most Pacific Islanders live in the coastal zone, and population growth is generally high (UNEP 1999; Kingsford *et al.* 2009). Extensive coastal development is supported by tourism, which accounts for up to half of gross domestic product for some Pacific Island nations (Jiang *et al.* 2010). Development pressures in many Pacific Island countries have led to exploitation of mangrove wetlands in particular. As in other regions, Pacific Island watersheds have substantially changed with development, including deforestation, agricultural practices and urbanization leading to altered hydrology and sedimentation effects (UNEP 2006).

Given most Pacific Island wetlands are coastal, climate change is already adversely impacting these ecosystems. Sea level rise is the most serious problem, contributing to coastal erosion, saline incursions of ground and surface waters from high sea level. Sea level is projected to rise 18–59 cm by 2100 (Table 1; IPCC 2007a). Sea level rise effects are likely to be exacerbated by other projected climate changes (Prowse and Brook 2011), with El Niño climate patterns and greater intensity of extreme events likely to cause more flooding and drought, as well as more Category 4 and 5 cyclones (Table 1; Nicholls 2011). Small island regions in the Pacific, Indian Oceans and Caribbean Sea are most at risk globally from sea level rise (IPCC 2007a).

Wetland loss presents significant risks to biodiversity, given the high degree of endemism and large proportion of globally threatened species found on Pacific Islands. Many Pacific Island species are threatened with extinction (Kingsford *et al*. 2009). While dependent on various factors such as tidal range and sediment supply, some mangrove forests and their associated ecosystems, will be lost on small islands with sea level rise (IPCC 2007a). In some areas, increased vertical accretion and inland migration of coastal wetlands could occur as an autonomous adaptive response to sea level rise (Nicholls 2011), but the ability of wetlands to adapt naturally to sea level rise is limited in areas with extensive coastal infrastructure. Furthermore, adaptive capacity may also be limited by the lack of sediment inputs on low-lying islands, and there is some debate as to whether this response is likely to be significant (Alongi 2008). There is also uncertainty about complex ecosystem responses (McLeod 2010). This pattern reflects global projections of significant decline of coastal wetlands due to sea level rise over the next century, particularly when combined with direct human impacts (Gillman *et al*. 2007; Legra *et al*. 2008; Nicholls *et al*. 1999). For example, native mangrove species are projected to reduce in area by 13% by 2100, across the 16 Pacific Island territories (UNEP 2006). There is potential to ameliorate effects of climate change on coastal wetlands particularly, through the management of other threatening activities, (i.e., harvesting of mangrove forests for timber; Kingsford *et al*. 2009).

Accurate projections of the extent of coastal wetland affected by sea level rise are required but improved planning of coastal zone development could protect some vulnerable wetlands. Measures to reduce erosion, particularly during extreme events, include preservation of natural barriers, such as coral reefs and dunes, or construction of physical barriers such as sea walls (Mimura 1999). Protection of coastal wetlands themselves affords protection from erosion and inundation to adjacent ecosystems. Although the direct reliance of Pacific Island societies on biodiversity is widely recognized, adaptation measures are more likely to be implemented if they offer immediate socio-economic or climate benefits as well as long-term climate change adaptation (Nicholls 2011; Duffy 2011; Grantham *et al.* 2011). For example, adaptive measures such as rainwater collection can preserve groundwater resources to lessen effects of saltwater intrusion, benefiting natural wetlands and crops.

Pacific Island nations face substantial barriers to the implementation of policy and management options with respect to climate change (Kingsford *et al.* 2009; Duffy 2011). The lack of financial resources in most Pacific Island nations means that environmental issues are not as high a priority as trade and economic development, health and public sector reform, and adaptation to climate has historically been reactive (Hay and Mimura 2006). Accordingly, environmental priorities with respect to climate change focus on provision of clean drinking water and protection of homes and food-growing capacity from sea-level rise. There is an absence of wetland policies (Ellison 2009, SPREP 2011) and a recent SPREP report referred to swamps as "perfect breeding grounds for mosquitoes that cause malaria and dengue fever" (Wickham *et al.* 2009). Further constraints come from a lack of technical skills (Kingsford *et al.* 2009) and often ad-hoc funding from foreign aid (UNEP 1999, Duffy 2011). Systems of governance, and therefore forms of environmental policy and uptake of initiatives are highly variable at the national level.

At a regional level, several multilateral environmental agreements have been widely adopted across Pacific Island countries. The United Nations Framework Convention on Climate Change has been ratified in many of the Pacific Islands. With the Australian Bureau of Meteorology, the South Pacific Sea Level and Climate Monitoring Project collects oceanic, sea-level and weather data in 11 nations and has strengthened technical and scientific capacity (UNEP 1999). Climate change and biodiversity management are key roles of the intergovernmental Secretariat of the Pacific Regional Environment Programme (SPREP). Co-operation and representation of Pacific nations in these ways will continue to be important in building the capacity of individual states to develop climate change adaptation policies to protect coastal wetlands at a more local level. There is also a critical need to manage other threats (Kingsford *et al.* 2009) to build resilience in freshwater ecosystems.

Synthesis

Climate change significantly threatens freshwater biodiversity, demonstrated by the six case study freshwater ecosystems in three major ways: increases in temperatures, changes to rainfall, run-off and consequent effects on flow and sea level rise (Table 2). Increased temperatures will particularly affect alpine areas where flow regimes are dictated by temperatures and residence time of snow. Snow melt rivers in Australia are heavily regulated but impacts of temperature affect dependent biota in their unregulated upper catchments (Hennessey *et al.* 2007). Similarly, alpine and glacial rivers in New Zealand will be affected, impacting on cold-adapted species as temperatures rise. In arid-zone rivers and wetlands, already degraded by anthropogenic impacts (i.e., the Macquarie Marshes), projected water loss by 2030 is minor compared to losses already experienced by river regulation (Jenkins *et al.* 2011). Projected increases in severe droughts and consequent reductions in run-off will mean less flow for downstream wetlands, compounding dramatic losses in biodiversity already experienced from river regulation and diversion of flows upstream. Coastal wetlands throughout Oceania, already impacted by human developments and water use, will also be irrevocably changed by sea level rise and salt water intrusion (Finlayson et al 2009; Schallenberg *et al.* 2010).

Clearly, reductions in overall greenhouse gas emissions will limit the extent of impacts on freshwater ecosystems. There are also many opportunities to improve resilience of ecosystems to climate change, through broad policy and specific management. And yet, despite the projected threats and current evidence of biodiversity loss due to climate change in freshwater ecosystems, the management and policy response is slow. In Australia, the impacts of sea level rise on coastal systems in the Northern Territory was anticipated in the late 1990s (Finlayson *et al.* 1997) and yet policy to reduce carbon emissions is yet to be implemented. Even major policy documents for managing freshwater ecosystems pay scant attention to the long-term impacts of climate change. For example, the Guide to the upcoming Murray-Darling Basin plan (MDBA 2010) adopted only a 3% reduction in surface water availability by 2012, half the wettest (and least likely given rainfall projections) scenario modeled by CSIRO (2008) (Schofield 2011). The Guide contained only brief reference to climate change as did the Adaptive Environmental Management Plan for the Macquarie Marshes (DECCW 2010). In New Zealand, the abundance of water and remoteness of melting glaciers may reduce the political imperatives for adaptation policy dealing with climate change, although at least the government has introduced a carbon trading scheme. In the Pacific Islands, lack of resources is the greatest constraint to the development and implementation of policy options to protect coastal wetlands from climate change (Morton *et al.* 2009; Kingsford *et al.* 2009; Duffy 2011).

Specific adaptations to threats of increased temperatures, altered flow regimes and sea level rise for freshwater ecosystems are possible albeit challenging and limited in spatial applicability (Table 2). A key adaptation will be to increase the resilience of freshwater aquatic ecosystems to other threats, particularly river regulation given its pervasive effect on biodiversity. For terrestrial regions, increasing the protected area system and its size is a critical adaptation option but this may be less successful for aquatic ecosystems unless the freshwater inflows are protected. Certainly, the establishment of protected areas on rivers provides a strong policy imperative for governments to actively manage river systems. For example the protected area in the Macquarie Marshes has strongly positioned the conservation agency to support and implement the buy-back of environmental flows for the ecosystem to meet their legal responsibilities. In arid-zone rivers, where regulation and climate change both increase drying, water buy-back and environmental flows provide the main adaptation to both regulation and human-forced climate change. Any success in clawing back the impacts of regulation with the purchase of environmental flows may be eroded by the projected impacts of climate change in drying these systems. For river systems that are not yet regulated, there should be an imperative to protect their flows (Kingsford *et al.* 2009; Pittock and Finlayson 2011), as this will increase their resilience to climate change. There will also be adaptation opportunities focused on improvements in the management of environmental flows. Many rivers are primarily managed for extraction and so dam management and release of environmental flows offers considerable opportunities for improvements (Watts *et al.* 2011; Kingsford 2011; Pittock and Hartmann 2011). These include "piggy-backing" environmental flows on delivery flows for irrigation as well as mimicking the pulses of flows in rivers when providing water downstream for extractive use. There will also be an increasing focus on the management of environmental flows to meet specific conservation objectives (Kingsford *et al.* 2011a) and this is likely to drive increasingly innovative ways of maximizing the outcomes in the management of environmental flows. This management will need to meet the principle of maintaining or improving connectivity between aquatic ecosystems. With increasing drying of freshwater ecosystems, there is a danger that connectivity will be lost between wetlands or refugia. This includes lateral connectivity which is already compromised on many river systems in the Murray-Darling Basin (e.g., Macquarie Marshes, Steinfeld and Kingsford 2011).

Temperature increases and effects on dependent alpine freshwater ecosystems will be difficult to counteract (Table 2). Revegetation of riparian areas will reduce temperatures and provides a possible adaptation although it may be difficult to establish in alpine areas. For key aquatic species, translocation may be the only option (Turak *et al.* 2011), but this will be costly and may impact on biota in ecosystems that receive translocated species. For coastal wetlands, sea level rise will be difficult to combat

Table 2. Climate change drivers, physical impacts, risks, and adaptation opportunities for the six case study freshwater ecosystems in this review.

Climate change drivers	Physical impacts	Risks	Adaptation opportunities	Case studies - Freshwater ecosystems impacted
Increased temperatures	Higher water temperature, reduced dissolved oxygen	Species loss as thresholds for tolerance to higher temperatures exceeded	Revegetate riparian landscapes to shade freshwater ecosystems, focus on low flow management to protect refugia, protect habitats from other impacts (i.e. livestock), expand protected areas, protect free-flowing rivers, maintain or improve connectivity (especially between refugia)	All freshwater ecosystems
Reduced rainfall, extended drought	Reduced flooding, increased time between floods,	Species loss as thresholds for tolerance to drying exceeded	Environmental flows (water buy-back, carry over, piggy-back on irrigation flows), improve dam management, focus on low flow management to protect refugia, protect free-flowing rivers update water sharing plans to manage during drought, reduce threats from regulation, maintain or improve connectivity (especially between refugia), minimise impacts from invasives	Macquarie Marshes and wetlands in dry regions
Rising sea level	Elevated salinity	Species loss as thresholds for tolerance to salinity exceeded	Levee banks, migration of tidally structured communities up rivers assisted by habitat restoration and creation, reduce threats such as mangrove harvesting	New Zealand coastal wetlands, Kakadu wetlands, Pacific Island wetlands
Increased temperatures, reduced rainfall	Reduced snowfall and melt	Species loss as flow regime disrupted and temperatures increase	Migration or translocation to higher elevations, minimize impacts from invasives, use regulation structures when available to improve flow/thermal regime, maintain or improve connectivity (especially between refugia)	Australian and New Zealand alpine and snow melt streams
	Melting glaciers	Species loss as glacial habitat retreats	Reduce greenhouse gas emissions to slow temperature rise	New Zealand alpine

with relatively few adaptation options that will guard against the loss of biodiversity (Table 2). Where coastal freshwater ecosystems are not able to migrate inland because of elevation or human infrastructure, they will be lost. In some places, adaptation to salt intrusion in coastal rivers may be achieved by building levees but this will be spatially limited and would probably need to coincide with human requirements for protection.

In addition to these adaptations, there should be a focus on active reduction of other threats that affect long term conservation of aquatic biodiversity (e.g., pollution, invasive species). For example, removal of introduced trout in some streams may buffer impacts of increasing temperature on cold adapted freshwater biota. Flushing flows may be utilized to maintain water quality but equally there needs to be a focus on mitigating the pollution of waterways, whether from industrial, urban, agricultural, or mining waste or natural pollutants exceeding natural levels (e.g., salinity, blue-green algal blooms).

CONCLUSIONS

There are a range of impacts of climate change on freshwater ecosystems and these often exacerbate current threatening processes. In particular, the most serious issues are increasing temperatures for alpine systems which will change flow regimes and also drive some cold-adapted species to extinction. For inland river and wetland systems that are already heavily affected by water resource development, climate change is projected to increase drying particularly in places such as the Murray-Darling Basin. This may erode hard won gains of environmental flows bought back to rehabilitate wetlands. For coastal systems in the Pacific, Australia and New Zealand, sea level rise will cause a change of state from freshwater to marine or estuarine ecosystems, radically altering the composition of biotic communities. There are clear imperatives for nations. At a global scale, reduction in carbon emission remains a critical goal that will reduce long-term impacts of a "business-as-usual" approach to carbon emissions. Increasing environmental flows and protecting flows of unregulated rivers and streams will improve the resilience of ecosystems but this will be challenging in our region with burgeoning human populations requiring water. Other adaptations are likely to be local in scale and expensive, including levees, translocations and revegetation. Climate change presents a major challenge for the freshwater ecosystems of Oceania. It is one that will be difficult to meet given the limited adaptations available but it is critical that we tackle the problem, given the long-term consequences on biodiversity and ecosystem services that will occur with the increasing loss and reduction in function in freshwater ecosystems.

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