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The Gippsland Lakes: management challenges posed by long-term environmental change

Paul I. Boon[A,C], *Perran Cook*[B] *and Ryan Woodland*[B]

[A]Institute for Sustainability and Innovation, Victoria University, Footscray Park, Vic. 8001, Australia.
[B]Water Studies Centre, School of Chemistry, Monash University, Clayton, Vic. 3800, Australia.
[C]Corresponding author. Email: paul.boon@vu.edu.au

Abstract. The Gippsland Lakes, listed under the Ramsar Convention in 1982, have undergone chronic salinisation since the cutting in 1889 of an artificial entrance to the ocean to improve navigational access, exacerbated in the mid–late 20th century by increasing regulation and extraction of water from inflowing rivers. Both developments have had substantial ecological impacts: a marked decline in the area of reed (*Phragmites australis*) beds; the loss of salt-intolerant submerged taxa such as *Vallisneria australis*, causing a shift to a phytoplankton-dominated system in Lake Wellington; and, nearer the entrance, an expansion in the area of seagrasses. Mangroves (*Avicennia marina*) first appeared in the late 1980s or early 1990s. Since 1986 recurring blooms of *Nodularia spumigena* have led to loss of recreational amenity and to the periodic closure of recreational and commercial fisheries. Changes to hydrological and salinity regimes have almost certainly shifted the contemporary fish community away from the pre-entrance state. Rises in eustatic sea levels and increases in storm surges will exacerbate the issue of chronic salinisation. Whether or not managers choose to intervene to prevent, or at least minimise, ongoing environmental change will inevitably prove controversial, and in some cases no socially or technologically feasible solutions may exist.

Additional keywords: cyanobacteria, estuary, eutrophication, fish, salinisation, wetland.

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Introduction

Nearly one-third of Australia's 65 Ramsar-listed wetlands are in the coastal south-east of the country, and in total these 18 sites cover an area of more than 400 000 ha (Department of the Environment 2014). The Coorong (140 500 ha, South Australia) is the largest of the south-eastern Ramsar sites, followed by three sites in Victoria: Corner Inlet (67 186 ha), Gippsland Lakes (60 015 ha), and Western Port (59 297 ha). The Gippsland Lakes Ramsar site is the topic of this paper. It presents a particularly difficult conundrum for coastal-zone managers, as the site encompasses high aesthetic, landscape and biodiversity values, includes geomorphological sites of international, State and regional significance, is a critical region for tourism and recreation in south-eastern Australia, and at the same time lies in a catchment increasingly modified by agricultural, industrial, and urban development, the rivers of which are subject to increasing extraction demands as sources of potable, irrigation, and industrial water. To complicate the situation further, the hydrological and physico-chemical characteristics of the Gippsland Lakes were altered fundamentally in 1889, when an artificial and permanent entrance was cut to Bass Strait in order to improve navigation and the safety of ships passing into and out of the Lakes. The creation of this permanent entrance shifted the Lakes from an open-and-closed coastal lagoon system to one permanently linked to the ocean, and resulted in an immediate change in water levels and in chronic salinisation of a formerly fresh- or brackish-water system. These pressures will be further exacerbated by chronic sea-level rise and increasing storm-surge impacts, and the near certainty of on-going salinisation. Sea-level rise and the salinisation of formerly freshwater systems is an issue faced by several other Ramsar sites in Australia, most obviously Kakadu National Park in the Northern Territory (Winn *et al.* 2006; Cobb *et al.* 2007; Finlayson *et al.* 2013).

The aim of this paper is to review the diverse factors that influence the environmental characteristics of the Gippsland Lakes and to determine whether there are consistent trends in the long-term ecological condition of this important Ramsar site. We focus on fringing and submerged angiosperms and wetlands, on phytoplankton and algal blooms, and on fish; water birds are not a focus because of the great role that conditions elsewhere in the continent have on bird abundances at a specific location (Kingsford and Norman 2002). Perhaps surprisingly, given the very high environmental, social, and economic values of the Gippsland Lakes and the multitude of threats they face, such a synthesis has not been undertaken recently in the published literature: the last broad-reaching assessment of, and prognosis for, environmental change in the Gippsland Lakes was published nearly 50 years ago, in a book chapter by the geomorphologist E. C. F. Bird, in 1966. (Unpublished accounts of environmental change include Harris *et al.* (1998) and Webster *et al.* (2001). The Ramsar site's Ecological Character Description (Department of Sustainability, Environment, Water, Population and Communities 2010) is also notable.) In our review, we bring together disparate sources of information on the ecology of the Gippsland Lakes, including not only scientific works in the peer-reviewed published literature but also a large number of previously unpublished and/or consultants' reports, which contain valuable – but otherwise largely unavailable – information.

The Gippsland Lakes – a brief description

Physical setting

The Gippsland Lakes are located on the south-eastern coast of Australia, latitude 37°49′ to 38°12′S and longitude 147°04′ to 148°08′E, and consist of a system of coastal lagoons and fringing wetlands, sheltering behind a series of sandy barriers that developed during the Late Pleistocene. Three sets of barriers were recognised by Bird (1978): a 'prior' barrier stand to the north of the Lakes, beneath the former sea cliff; an 'inner' barrier, north of Lake Reeve; and an 'outer' barrier, on the seaward side and which carries the Ninety Mile Beach. Each barrier is surmounted by beach ridges and dunes. Development and maintenance of this system of coastal barriers is promoted by the small tidal range in south-eastern Australia ($<$2 m), an abundant supply of sand moving along the coast and, until recently, slow rates of relative sea-level change (Sloss *et al.* 2007).

The lagoon system consists of four large, shallow coastal lakes (Lake Wellington – 148 km^2; Lake Victoria – 78 km^2; Lake King – 97 km^2; and Lake Reeve – 52 km^2), fed by five river systems: the Latrobe–Macalister–Thomson system, flowing into the western side of Lake Wellington; the Avon–Perry system, flowing into the northern side of Lake Wellington; and the Mitchell, Nicholson and Tambo Rivers, all eastern rivers flowing into Lake King (Fig. 1). Associated with these rivers and the shoreline of the four lagoons is a complex mosaic of fresh, brackish, and hypersaline wetlands; the largest of these are the brackish-water Lake Coleman ($\sim$20 km^2), Dowd Morass ($\sim$15 km^2) and Macleod Morass ($\sim$5 km^2), and the ephemeral and often hypersaline Lake Reeve.

The lagoons have a combined shoreline of $\sim$320 km, and the rivers drain a catchment of 20 600 km^2, just over one-tenth of the State of Victoria (Bird 1978). The catchment includes several large towns and cities (Sale towards the west; Warragul, Moe, Morwell, Traralgon, and Bairnsdale; plus Lakes Entrance to the east), Victoria's major electricity-generating facilities in the Latrobe Valley industrial area, extensive dry-land and irrigated farmland (e.g. the Macalister Irrigation District), and a significant proportion of Victoria's hardwood (native) and softwood (plantation) timber resources.

Economic and social values

The Gippsland Lakes support Victoria's largest commercial fishing fleet and the single largest recreational fishery in the State for the iconic black bream (*Acanthopagrus butcheri*). The value of the commercial fish catch in the Gippsland Lakes and the adjacent Lake Tyers was AU$1 138 000 in 2009/10, with black bream accounting for 38% of the total value (Department of Environment and Primary Industries 2012). The remainder included commercially valuable species such as dusky flathead (*Platycephalus fuscus*), tailor (*Pomatomus saltatrix*), and silver trevally (*Pseudocaranx georgianus*), as well as less valuable species such as carp (*Cyprinus carpio*) and yellow-eye mullet (*Aldrichetta forsteri*). Recreational catch of black bream in the Gippsland Lakes is thought to have accounted for $\sim$42% of the total black bream harvest of 237 tonnes during 2000/01 (Kemp *et al.* 2013). Recreational fishing in the Gippsland Lakes targets not only black bream, but also estuary perch (*Macquaria colonorum*), snapper (*Pagrus auratus*), and various species of flathead, whiting, squid, and prawns.

The social value of the Lakes for recreation, visual amenity, and in providing habitat for wildlife and biodiversity is reflected in the economic value of tourism to the region. It has been estimated that in 2006, the Lakes attracted $>$4 577 000 total visitor-days, including 2 326 000 spent in overnight visits and 1 436 000 in local day visits (URS 2008). If an expenditure of AU$200 per day is (conservatively) assumed for overnight visitors and AU$50 per day for local day visitors, the tourism value of the Gippsland Lakes is at least AU$550 million each year. It is likely that this value has increased over the past decade, given population and other demographic changes outlined in the Discussion. Non-market values are also important in an estimation of the total value of the Gippsland Lakes, and URS (2008) noted that four of the large wetlands (Clydebank Morass, Dowd Morass, Heart Morass, and Sale Common) were worth AU$1.15 million (in 2006 dollars) for their biodiversity value alone.

The Gippsland Lakes Ramsar site

Almost all of the Gippsland Lakes and its fringing wetlands were listed under the Ramsar Convention in 1982 (Department of Sustainability, Environment, Water, Population and Communities 2010) on the grounds that they met four criteria: Criterion 1 – a particularly good representative example of a natural or near-natural wetland characteristic of the appropriate biogeographical region; Criterion 3 – regularly supports substantial numbers of waterbirds from particular groups, particularly grey teal (*Anas gracilis*), chestnut teal (*Anas castanea*), black swan (*Cygnus atratus*), Australasian grebe (*Tachybaptus novaehollandiae*), Eurasian coot (*Fulica atra*), and great cormorant (*Phalacrocorax carbo*); Criterion 5 – regularly supports 20 000 or more waterfowl; and Criterion 6 – regularly supports $>$1% of the individuals in a population of little tern (*Sterna albifrons*), common tern (*Sterna hirundo*), black swan, great cormorant, great crested grebe (*Podiceps cristatus*), and Australian pelican (*Pelecanus conspicillatus*).

Critical to the maintenance of these bird populations are fringing fresh, brackish, and hypersaline wetlands (Bird 1962*b*; Corrick and Norman 1980; Cowling and Lowe 1981). The brackish-water wetlands are the largest and are typically woodlands of swamp paperbark (*Melaleuca ericifolia* Sm), interspersed with swards of common reed (*Phragmites australis* (Cav.) Trin. ex Streud) and open water, previously colonised by

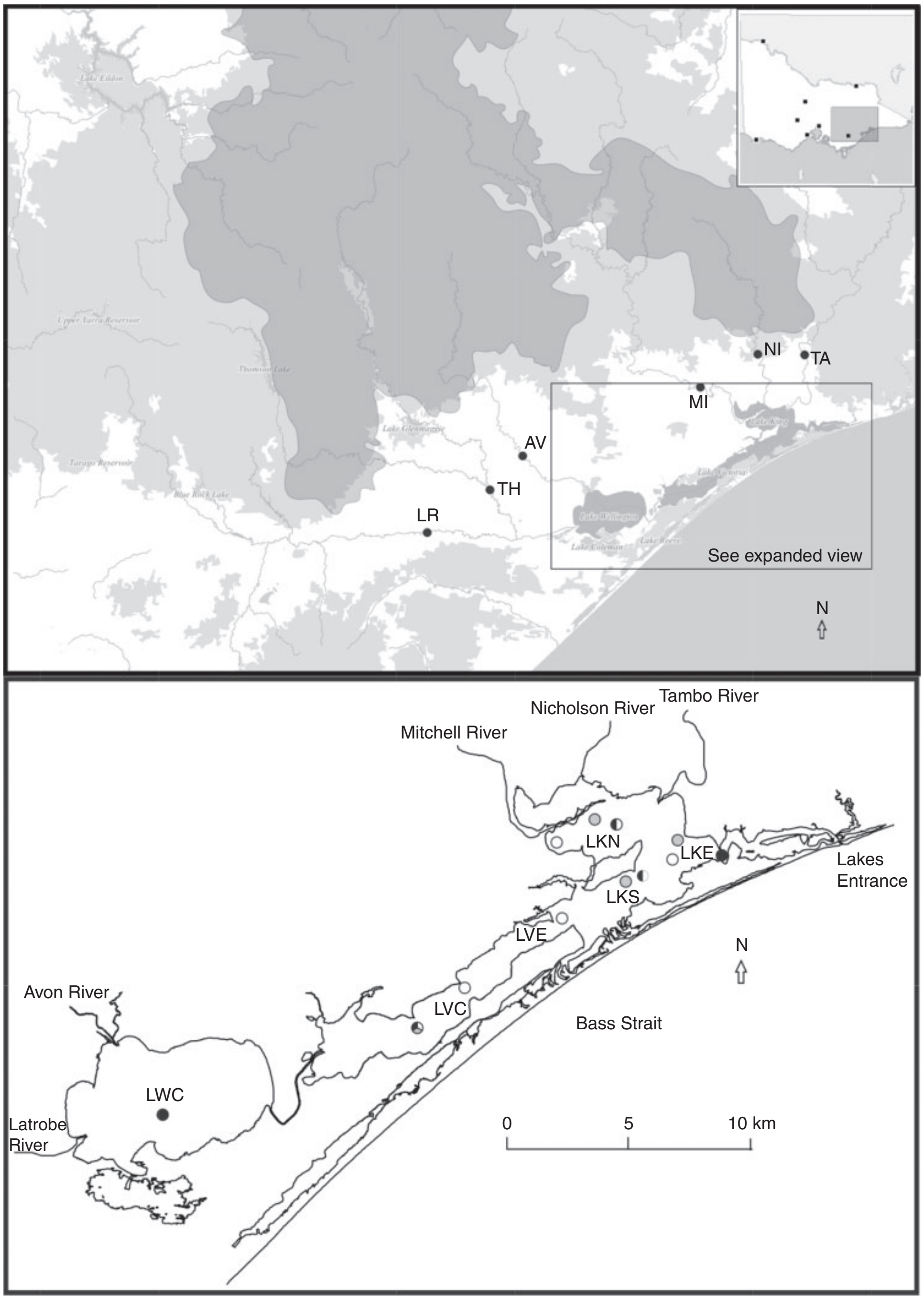

Fig. 1. The Gippsland Lakes, south-eastern Australia. The top panel shows the Gippsland Lakes catchment, with the area burned in the 2006–07 bushfires (dark grey), and the location of gauging stations on the Latrobe (LR), Thomson (TH), Avon (AV), Mitchell (MI), Nicholson (NI) and Tambo (TA) rivers. The bottom panel shows environmental monitoring sites deployed across the Gippsland Lakes: black circles indicate long-term EPA Victoria monitoring sites; white circles are sites used for benthic flux measurements; and grey circles are recently installed EPA Victoria continuous-monitoring stations. Note that some locations are combinations of the above three site types.

the submerged and mostly salt-intolerant angiosperm, eel weed (*Vallisneria australis*). These vegetation communities have been described most recently by Boon *et al.* (2008), Raulings *et al.* (2010, 2011), and Boon (2014); earlier descriptions include those of Bird (1961, 1962*b*), Aston (1977), Ducker *et al.* (1977), and Clucacs and Ladiges (1980). The hypersaline wetlands associated with Lake Reeve are typically saltmarsh, dominated by chenopods such as beaded glasswort (*Sarcocornia* *quinqueflora*): see Sinclair and Boon (2012) and Boon *et al.* (2014). Of the remaining wetlands, only Sale Common (~2 km^2) is fresh, although the lower sections of Macleod Morass are maintained in a freshwater state by the (deliberate) ingress of treated sewage from Bairnsdale Sewage Treatment Plant and the positioning of a regulatory structure at the end of the wetland, nearest Lake King, to prevent saline intrusions.

The Gippsland Lakes Ramsar site also contains one site of international geological or geomorphological significance (the Mitchell River silt jetties: see Bird 1970; Bird and Rosengren 1974), two of national significance (Sperm Whale Head to Boole Poole Peninsula, including the Outer Barrier and Ninety Mile Beach, relict tidal channels and tidal delta, Rotomah Island, Boole Poole Peninsula, and Sperm Whale Head; and Cunninghame Arm), and seven of State significance (Lake Reeve and the Outer Barrier; the Tambo River delta; Macleod Morass; Point Turner and the Banksia Peninsula, the Outer Barrier near Seaspray; McLennans isthmus and McLennans Strait, and the Latrobe River delta).

In addition to the main lagoonal complex with its associated fringing wetlands, the Ramsar site includes two smaller adjacent estuarine systems: Lake Tyers, an intermittently-open-and-closed estuary of 12 km^2 ~12 km east of the township of Lakes Entrance, fed by Stony and Cherry Tree Creeks (on the Toorloo Arm), and Boggy and Ironstone Creeks (on the Nowa Nowa Arm); and Lake Bunga, a small (0.5 km^2) intermittently-open-and-closed estuary (formerly part of the Gippsland Lakes, before the opening of the artificial entrance) ~6 km from Lakes Entrance. This review addresses only the Gippsland Lakes *sensu stricto*, and does not consider either Lakes Tyers or Lake Bunga.

The changing physico-chemical environment of the Gippsland Lakes

The colonisation of the Gippsland region by Europeans was described by Watson (1984), Wells (1986) and Synan (1989); critical activities include the cutting of forests for timber, the development of the catchment for dryland and irrigated agriculture, the discovery and extraction of precious metals (e.g. gold), the damming of rivers, particularly those in the western parts of the catchment, and the modification of the Lakes environment itself for shipping. All have resulted in changes to hydrological and salinity regimes and to loads of nutrients and sediments carried by the rivers that debouch into the main lakes.

The permanent connection to Bass Strait at Lakes Entrance – impacts on salinity regimes

When the Gippsland region was colonised by Europeans in the 1840–50s, the Gippsland Lakes were linked with the sea only by a shifting and intermittent outlet through the sand barriers at the easterly part of Lake King. Shipping access to the ocean was dangerous and constrained by the often-closed entrance. In response to navigational limitations posed by the intermittent entrance, a permanent (artificial) entrance was – after many attempts – successfully cut to Bass Strait in 1889 at Lakes Entrance, ~5 km from the natural entrance (Bird and Lennon 1989). Sand deposition soon sealed off the old natural outlet, and navigation through the artificial entrance has been maintained ever since by dredging (Bird 1966, 1978; Wheeler *et al.* 2010).

As a consequence of the creation of a permanent entrance to the sea, mean water levels in the two most easterly lagoons – Lake King and Lake Victoria – now correlate closely with mean water levels in Bass Strait on time scales of ~1 week or longer (Webster *et al.* 2001). Variations in ocean levels in Bass Strait occur in response to long-term changes in atmospheric pressure and the set-up or set-down of storms. The resulting longer-term variation in ocean water levels then dominates the observed pattern of variations throughout the two eastern lagoons, and result in fluctuations typically of ±0.2 m about mean sea level. During large ocean surges in Bass Strait the eastern lagoons respond with variations in mean water level of as much as 1 m, and, as noted below, these can have major implications for saline intrusions into the western parts of the Lakes complex.

In contrast, Lake Wellington, the western-most lagoon in the Lakes complex, is not tidal. It receives saline influences as brackish water moves through McLennan Strait, primarily as a result of variations in water level in the eastern lagoons (Brizga *et al.* 2011). These can be induced not only by changes in mean ocean levels but also when storms over the eastern catchments result in flooding in the eastern rivers (especially the Mitchell River), which in turn causes saline water to back-up into the western lagoon and lower reaches of the inflowing rivers. Because of the restricted connection to the sea at Lakes Entrance, these large inflows of fresh waters during floods down the eastern rivers cause an increase in water levels in Lakes King and Victoria, and flooding in nearby low-lying areas. Lake water levels are modelled to increase by up to 1.8 m and 2.2 m at Lakes Entrance and Lake Wellington, respectively in the event of a 1-in-100-year flood (Moroka 2010). Elevated water levels last for several days, until discharge through the entrance returns water levels to normal.

The opening of the permanent entrance at Lakes Entrance had two major consequences for hydrological and salinity regimes in the Gippsland Lakes. The first – and immediate – consequence was to modify the range of fluctuating water levels in the Lakes; prior to 1889, water levels would increase by ~2 m when the entrance was closed, due to on-going river discharge and precipitation (Bird 1966). Such high water would persist until the sand berm was breached and the water escaped to the ocean. Following the cutting of the permanent entrance, water levels vary over only a slight range, largely driven by fluctuations in mean sea level as described earlier. Water levels now do increase in times of flood, when water backs-up through the lagoons and causes localised flooding, but the minor variations induced by tidal influence extend only a short distance up the lower-most parts of Lakes King and Victoria.

The second consequence has been – and continues to be – the progressive salinisation of the Gippsland Lakes, which previously were relatively fresh, or at best episodically brackish, because of their intermittent linkage with the ocean and the large discharge of freshwaters down the inflowing rivers (Bird 1966; Bird and Rosengren 1974; Harris *et al.* 1998). According to Ducker *et al.* (1977), the first eyewitness account of the Gippsland Lakes and their surroundings was made by W. A. Brodribb, one of a group of stockmen who in 1841 crossed the Latrobe River and Avon River and followed the latter '… along broad marshes covered with reeds … to a broad sheet of water extending right and left as far as the eye could see'. Brodbribb's party named this body of water Lake Wellington, and recorded the waters to be '… fresh at least drinkable by man and beast'. Oral histories of local families in the late 19th century tell of commercial fishermen not taking drinking water with them whilst netting, as the lakes were fresh enough to provide drinking water merely by placing a pannikan over the side of the boat (Ellis and Lee 2002).

With the creation and maintenance of the permanent opening, low-frequency sea-level variations in Bass Strait, combined with tidal forcing and the impacts of storms, now affect water levels and salinities across all of the Gippsland Lakes, although the effect decreases with increasing distance from the entrance (Webster *et al.* 2001; Brizga *et al.* 2011). Over recent decades, salinities in the water column of Lake King have typically ranged from 8–26 (EPA Victoria 2013). Salinities in Lake Wellington, the lagoon most distant from the ocean, typically range from <1 to 10 (EPA Victoria 2013), but this is the most variable of all the lagoon systems, and since 1976 salinities have ranged from <1 after floods to >15 and >20 during the 1998 and 1982 droughts, respectively (Grayson 2003).

Thus not only has there been a chronic increase in salinity since the end of the 19th century, but the permanent opening allows for episodic intrusions of saline water into the Gippsland Lakes following storm surges or via differential discharges down the eastern and western rivers, which can push saline water into other lagoons and into their fringing wetlands, even if they are not directly affected by flooding (Parks Victoria 1997). In contrast to pre-entrance conditions, when ingress of seawater was a temporary phenomenon dependent on intermittent breaches in the barrier dune, the artificial permanency of the current entrance has created a constant boundary condition for the Lakes. As a result, variability in salinity is now controlled also by variability in freshwater inflows from the main rivers, and periods of low river discharge are thus strongly correlated with periods of high lake salinity, and vice versa.

River regulation and reductions in freshwater inflows

The Latrobe–Macalister–Thomson River system (44% of mean annual inflow) and the Mitchell River (35% of mean annual inflow) are the largest contributors of fresh water to the Gippsland Lakes; the Avon–Perry (8%), Tambo (11%) and Nicholson Rivers (2%) make up the balance (Moroka 2010). These rivers have been variously developed to provide potable water for the city of Melbourne (primarily via the Thomson Dam, on the Thomson River), to support irrigated agriculture (e.g. Glenmaggie Dam, on the Macalister River), and to provide cooling water (from the Latrobe River) for thermal electricity generation in the Latrobe Valley industrial area (e.g. for the 1600-MW Hazelwood power station, constructed in 1964). The Thomson Dam, opened in 1984, has a storage capacity of 1068 GL and is complemented by smaller storages on various tributaries of the Latrobe–Macalister–Thomson river system, such as Moondarra Reservoir (1962, 30 GL). At 177 GL, Lake Glenmaggie, opened in 1926 and increased in capacity in 1958, is the largest of the irrigation storages. Lake Narracan (1961, 7 GL) was originally constructed to provide cooling water for brown-coal-burning power stations, but is used now almost entirely for recreation, and on-site storages now provide cooling water for this industry (e.g. Hazelwood Pondage, on the Latrobe River). Also extracting water from the Latrobe–Macalister–Thomson river system is the Maryvale paper mill, constructed in 1937.

The consequence of the construction and operation of this infrastructure is that ~20% of the average annual discharge of rivers that flow into the Gippsland Lakes is extracted for agricultural, industrial and domestic purposes before it reaches the Lakes (Moroka 2010). This represents a marked increase in extraction over recent decades, and even as late as 1980 Clucacs and Ladiges (1980, page 11) reported that only 8% of water carried by the rivers was diverted for human use; at that stage (i.e. before the completion of the Thomson Dam), water was extracted mainly from the Macalister River, for irrigation. Clucacs and Ladiges (1980) predicted that, if all planned diversions were implemented, 24% of flows previously entering the Lakes would be intercepted by 2000.

The western rivers – the Latrobe–Macalister–Thomson system – are the most regulated, and they currently supply ~96% of total volume of water extracted for human use from the rivers that flow into the Gippsland Lakes. In contrast, the eastern rivers rise and flow through relatively steep land, much of which is protected as National Park or as State or Forest Park and are currently subject to little extraction. The eastern rivers are largely unregulated and thus extraction represents a markedly smaller proportion of their flows (although most extraction occurs in summer and autumn, when flows are lowest and the rivers are possibly flow-stressed already). Just over 30% of average annual flows of the Latrobe–Thomson–Macalister system are currently captured for storage or are extracted for nearby use. If existing water entitlements were exercised, the additional extraction of water would reduce riverine discharge by a further 8% (of current inflow), which would equate to a total reduction of ~44% from natural inflows for the Latrobe River system (Moroka 2010). An additional consequence of regulation and extraction of water from the Latrobe–Thomson–Macalister system, and from the Latrobe River in particular, is that the wetlands fringing these rivers are less likely to be inundated by small to medium-sized floods, even if the passage of very large floods is almost unaffected by river regulation.

The Latrobe–Macalister–Thomson system also receives several licensed discharges of waste, the major contributions being treated sewage from the townships of Warragul, Moe and Morwell, and industrial wastewater from thermal power stations. Within the Tambo catchment there are former mining areas around Cassillis that have eroded in the past, creating large slugs of sand within the lower reaches of the river near Bruthen and Tambo Upper and erodible agricultural areas, particularly in granitic areas. Both the Avon and Nicholson Rivers drain from vegetated upper catchments into areas that are now dominated by cleared agricultural land along their lower reaches. The Mitchell River is currently unregulated.

Although groundwater inflows make a relatively small direct contribution to the water balance of the Lakes system, there is strong evidence that discharge from groundwater to the various tributary streams is likely to make a significant indirect freshwater contribution to the Gippsland Lakes (Moroka 2010; Unland *et al.* 2013). Groundwaters are recharged from rainfall and by irrigation, with some minor upwards leakage from deeper aquifers such as the Boisdale Aquifer. Groundwater discharge contributes 24–36% of annual average flow in the Avon River during periods of average rainfall; similar contributions are expected for the other major rivers, particularly the Mitchell, Tambo and Nicholson Rivers.

Nutrient and sediment loads

External nutrient inputs to the Gippsland Lakes are dominated by riverine inputs (as opposed to marine sources), and, of these inputs, those from the Latrobe River dominate (Table 1). It is likely that nutrient loads have changed substantially in the past ~150 years, as agriculture and urbanisation within the Gippsland Lakes catchment have expanded and become more intensive, as reported for elsewhere in Australia by Harris (2001) and Davis and Koop (2006). More generally, water-quality issues have been identified as a threat to Ramsar-listed wetlands on a global scale in the recent review by Verhoeven (2014).

Table 1. Summary of long-term loads of suspended solids and nutrients for six rivers that discharge into the Gippsland Lakes
Source: Grayson *et al.* (2001)

River	Total estimated load (tonnes $year^{-1}$)		
	Suspended solids	Total nitrogen	Total phosphorus
Latrobe River	93 670	1277	132
Thomson River	36 230	346	56
Avon River	35 970	321	32
Mitchell River	26 800	436	47
Nicholson River	6670	59	7
Tambo River	10 540	218	15

Loads of total nitrogen (TN) and total phosphorus (TP) to the Gippsland Lakes from the western catchments have increased by a factor of 3–5 since European colonisation, compared with a factor of <2 for the less-developed eastern catchments. Taken together, these values generate Lakes-wise anthropogenic increases in nutrient loading by a factor of 2–3 (Grayson *et al.* 2001). On a catchment-wide basis, most of the increased nutrient loads arise from dryland grazing and from irrigated agriculture (Cottingham *et al.* 2006). Monitoring of flow and nutrients since 1979 has enabled load estimates to the Gippsland Lakes to be constructed over a three-decadal time span (Cook and Holland 2012), and this analysis shows that annual loads are highly variable and that the principle driver for this variability is river flow. Accordingly, loads of TN and TP entering the Lakes are highly correlated with river flow (Fig. 2).

Fire is also an important driver of nutrient loads, and there was a marked increase in nutrient export relative to catchment inflow following a flood in June 2007 that occurred after bushfires had burned ~40% of the catchment of the Gippsland Lakes (Cook and Holland 2012). These combined events led to a massive increase in loads of TN, and of particular note a four-fold increase in nitrate loads. Interestingly, nutrient loadings from the catchment rapidly returned to their long-term relationship with flow in the years following this catastrophic event (Fig. 2).

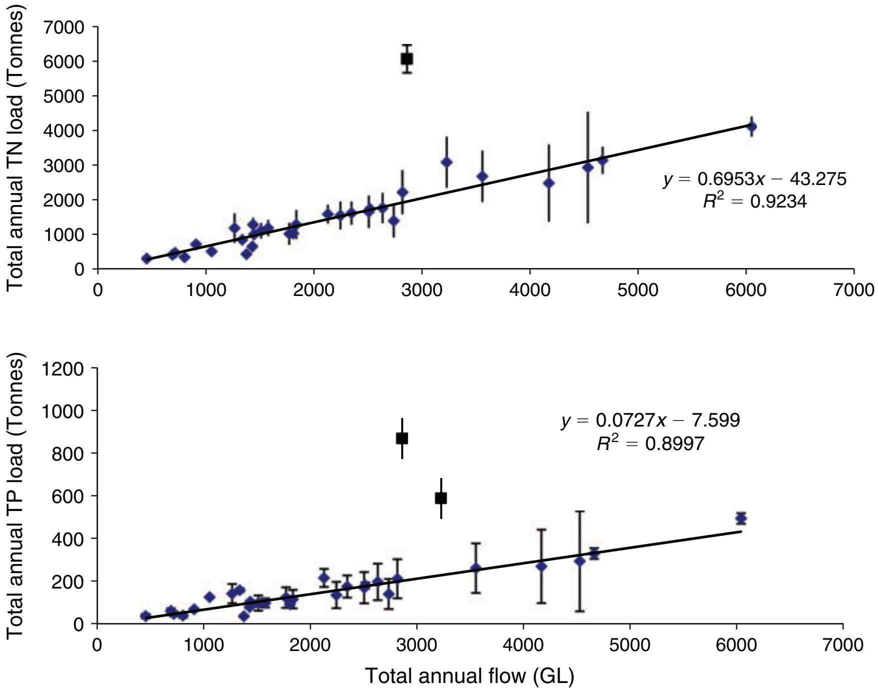

Fig. 2. The relationship between flow and annual (1 June–31 May) nutrient loads to the Gippsland Lakes over the period 1978–2011 (Cook 2011). The outlying points (black squares) are excluded from the regression and occurred in the year 2007–08 (TN: total nitrogen, upper diagram) and 2007–08 and 1995–96 (TP: total phosphorus, lower diagram). The cause of the 2007–08 outlier was the 2007 bushfires, and the cause of the 1995–96 outlier is unknown.

Ecological consequences of altered physico-chemical conditions

Fringing vegetation and shoreline dynamics

The creation of the permanent entrance at Lakes Entrance initiated an ecosystem-wide cascade of environmental consequences, starting with the whole of the Gippsland Lakes developing into a permanently estuarine environment (*sensu* Tagliapietra *et al.* 2009). As shown earlier, before the cutting of the artificial entrance the lagoons and their fringing wetlands would have been relatively fresh because of their intermittent linkage with the ocean and the large volumes of fresh water that discharged into the Lakes from the then-unregulated rivers (see also Saunders *et al.* 2008). Salinity increases would have been more gradual than the near-instantaneous effect on water levels, but salinity-mediated impacts of chronic seawater inputs on salt-intolerant fringing vegetation (e.g. *Phragmites australis*) were probably evident within the first few decades after the entrance was artificially opened (Bird 1966; Bird and Rosengren 1974).

That there would be marked ecological consequences caused by making a permanent opening to the ocean at Lakes Entrance were first explicitly acknowledged by Bird (1966), who predicted that the existing fringes of *P. australis* would be replaced by the putatively more salt-tolerant *Melaleuca ericifolia* and, ultimately, even the latter would be replaced by coastal saltmarsh. An analysis of post–World War 2 aerial photographs confirmed the progressive loss of reed beds and their replacement by swamp paperbarks in Dowd Morass, one of the largest of the fringing wetlands (Boon *et al.* 2008). There is also evidence for an increase in the area of saltmarsh around the Gippsland Lakes more generally, although the spatial and temporal patterns are exceptionally complex (Sinclair and Boon 2012). The submerged, freshwater angiosperm *V. australis* was formerly common in the lagoons and fringing wetlands (Aston 1977), including in Lake Wellington (Ducker *et al.* 1977; Bird 1978), but is now precluded by high salinities and unstable sediments from these areas (Salter *et al.* 2010*a*). High salinities also adversely affect the condition of adult paperbark trees and their ability to recruit sexually (Bird 1962*b*; Ladiges *et al.* 1981; Robinson *et al.* 2006, 2008, 2012; Salter *et al.* 2007, 2008, 2010*b*, 2010*c*).

The loss of reed beds and other types of fringing vegetation has had serious impacts on shoreline stability (Bird 1983). In fact, the first written record of this impact was by Hart (1921), who reported the cut-back and erosion of shorelines previously densely vegetated by swamp paperbarks. The progressive loss of reed beds and the associated retreat of the geomorphologically significant silt jetties along the Mitchell River has been documented for over 50 years (e.g. Bird 1961, 1962*a*, 1970, 1978, 1983; Bird and Rosengren 1974; Sjerp *et al.* 2002). Several other shorelines around the Gippsland Lakes are similarly subject to increased erosion, especially those on unconsolidated sediment (e.g. sand, silt, and clay), on organic deposits, or on other poorly consolidated material, such as weakly indurated sandstone or mudstone (Neville Rosengren, pers. comm., September 2014). Bird (1962*a*, 1970, 1983) reported that examples of highly vulnerable shorelines included the Mitchell and Tambo River deltas; in a later study, Sjerp *et al.* (2002) identified also the Latrobe and Avon River deltas, and parts of McLennan Strait as showing evidence of continuing erosion. Other areas of substantial erosion, identified by Sjerp *et al.* (2002), include Roseneath Point, Swell Point, Storm Point, west of the Avon River/Clydebank Morass, Marlay Point, around Loch Sport, Luff Point, Harrington Point, northern Raymond island, Point Fullarton, Tambo Bluff, and the northern shores of Jones Bay.

The impact of progressive salinisation has not been limited to the loss of fringing reed beds and related increased rates of shoreline erosion; it has been manifest also in changes in condition and/or state in almost all the wetlands that fringe the perimeter of the Gippsland Lakes. As a result of the combination of the creation and maintenance of the permanent opening to the ocean and the marked reductions in freshwater inputs from the rivers, low-frequency sea-level variations in Bass Strait, combined with tidal forcing and episodic storms, now control water levels and salinities across all of the Gippsland Lakes. Water-column salinities in Lake Wellington now vary over a wide range, largely according to incidences of drought and flood, but over recent decades have often exceeded 15–20 (Grayson 2003). The Lake Wellington wetlands are nominally classified as 'permanent deep-water freshwater wetlands' in the State-endorsed system used to classify wetlands in Victoria; 'freshwater' in this typology is indicated by salinities of $<$1–3; even so, water-column salinities in one of the largest brackish-water wetlands (Dowd Morass) over the period 2003–06 have regularly exceeded 15 (Boon *et al.* 2008; Raulings *et al.* 2010, 2011).

Currently only two of the extensive mosaic of fringing wetlands around the Gippsland Lakes remain fresh: the relatively small Sale Common ($\sim$2 km^2), the most upstream of the wetlands included in the Ramsar listing, and the larger Macleod Morass, which is maintained in a fresh condition only because of near-continuous inputs of tertiary-treated sewage from the Bairnsdale Sewage Treatment Plant into its upper portion and the construction of barrage gates at the lower end, which prevent intrusions of saline water from the lower reaches of the Mitchell River and Lake King. Attempts to rehabilitate the now-salinised wetlands have been only partly successful (Raulings *et al.* 2007, 2011; Boon 2014), in large part because of the risk of further saline intrusions and the possibility that potential acid sulfate soils would be activated should water levels be dropped sufficiently to allow flushing flows of fresh waters from the Latrobe River.

Some wetlands on the western side of Lake Wellington also receive increased salt loads from their surrounding catchments, via groundwater (Sinclair Knight Merz 2001). These salts are mobilised by elevated watertables driven by excess recharge from irrigation and catchment clearing. The relative significance of this salt source to individual wetlands is likely to vary with their proximity to Lake Wellington and other areas of terrestrial secondary salinisation, but quantitative information on groundwater discharges and their impacts on aquatic systems are lacking for the Gippsland region.

The appearance of mangroves

Only one taxon of mangrove, *Avicennia marina* subsp. *australasica*, occurs in Victoria (Duke 2006). Most texts show that it has a discontinuous distribution, from the Barwon River in the west (38°17′S, 144°30′E) to McLoughlins Beach, in the Corner

Inlet–Nooramunga region of South Gippsland (~38°40′S, 146°52′E), in the east (e.g. Barson and Calder 1981; Harty 1997). *A. marina* occurs along the Victorian coast mostly as a dense, monospecific shrubland, with individuals growing as shrubs or small trees from 0.3 to 4 m tall. Their densities, however, can vary from individual plants growing sparsely on the shoreline to dense, near-continuous belts of vegetation.

There is a small stand of mangroves at the distal end of Cunninghame Arm, in the most eastern part of the Gippsland Lakes. A field inspection undertaken by one of us (PIB) in September 2014 indicated an isolated specimen also near Bullock Island (Lakes Entrance). It is possible that other specimens occur elsewhere in the most saline parts of the Lakes. The origins of these mangroves is a topic of debate. According to Harty (2011), the specimens in Cunninghame Arm were planted, probably in the late 1980s or early 1990s. No evidence was provided for the assertion as to their origins, but Harty's estimate of the date of the arrival of mangroves in the Gippsland Lakes is generally agreed upon. It is quite possible that, instead of being artificial, mangroves entered and established within the Gippsland Lakes as a belated consequence of the permanent opening at Lakes Entrance. The potential for mangrove propagules to spread (either from the west, from Corner Inlet–Nooramunga; or from the north, from stands in southern New South Wales) suggests their establishment in the Lakes may be a natural phenomenon, occurring in response to the creation of a new and vacant niche in the intertidal zone as a result of chronic salinisation. Clarke and Allaway (1993) and Clarke and Myerscough (1993) showed that the establishment of *A. marina* was limited within existing mangrove stands only by the amount of propagules, but in mangrove-free areas by light, salinity, and sediment suitability.

As an expansion of mangroves has been reported for many parts of south-eastern Australia (Saintilan and Williams 1999; Rogers *et al.* 2005; Saintilan *et al.* 2009), it is perhaps not surprising that they are found now also in the Gippsland Lakes. The limiting factor may be dispersal from existing mangroves, and the sole report on the topic (Clarke 1993) suggests a limited capacity for *A. marina* propagules to move more than ~10 km from their parent tree. Regardless of the mechanism by which they came to be established in the Gippsland Lakes, the current presence and likely spread of mangroves down Cunninghame Arm is a good indication of the progressive, and ongoing, shift from a formerly freshwater or intermittently brackish water system into an estuarine, and in places fully marine, coastal system.

Seagrass beds

Seagrass supports a wide range of ecosystem functions, including the provision of habitat, sediment stabilisation, and nutrient cycling, in the Gippsland Lakes (Waycott *et al.* 2009). Seagrasses cover ~8% of the area of the main lagoons of the Lakes; *Zostera nigricaulis* and *Zostera mulleri* are the dominant species, but *Ruppia spiralis* is also present. Seagrass habitats in the Gippsland Lakes support high diversities and densities of benthic invertebrates (Poore 1982) and fish (Warry and Hindell 2012), and also provide nutritional support for some fish and invertebrates (Warry *et al.* 2013). Seagrass in the Lakes is vulnerable to nutrient enrichment, sedimentation, and reductions in water clarity (Waycott *et al.* 2009). These threats increase with modification of land and hydrology in coastal catchments, particularly in estuarine ecosystems like the Gippsland Lakes where seagrasses inhabit shallow waters close to human activity (Roob and Ball 1997; Warry and Hindell 2012).

The extent and condition of seagrass beds in the Gippsland Lakes is highly dynamic on decadal (Roob and Ball 1997) and interannual (Warry and Hindell 2012) scales. The specific mechanisms generating these temporal and spatial patterns have not yet been unravelled, but variations in freshwater inputs and of loads of nutrients and sediments from the catchment are likely to influence seagrass condition and extent. From 2009 to 2012 inclusive the condition of seagrass in the Gippsland Lakes was measured via underwater videography and the footage was used to rank seagrass physical condition, on the basis of coverage and blade density (Warry and Hindell 2012). This analysis showed that interannual variability of seagrass physical condition was highest at sites closest to the outlet of the Mitchell River (i.e. the major freshwater input to the Lakes) (Warry *et al.* 2013). Variation in the nutrient and light conditions was likely higher at sites closer to freshwater inputs than those closer to the entrance, where marine exchange dilutes and flushes catchment inputs, potentially generating more constant physico-chemical conditions.

Relatively poor physical condition of seagrass has followed periods of extended algal blooms in the Gippsland Lakes (Warry and Hindell 2012). Algal blooms reduce light availability for benthic communities and poor seagrass condition was observed in September 2008 immediately following the prolonged 2007–08 bloom of the cyanobacterium *Synechococcus* spp. Declines in physical condition were observed also in April 2012 compared with April 2011, following a bloom of the cyanobacterium *Nodularia spumigena* in the summer of 2011–12. Elemental and isotopic compositions of seagrass leaves sampled in April 2012 further indicated that seagrass plants were light- rather than nutrient-limited during this period (Warry *et al.* 2013).

How then have seagrass beds changed since European colonisation of the region or, from a more limited perspective, since the Gippsland Lakes Ramsar site was listed in 1982? Whilst the mechanisms generating variation in seagrass growth and decline in the Gippsland Lakes remain to be fully understood, it is emerging that light availability and proximity to catchment inputs influence plant condition. Thus it can be concluded in general terms that modification of catchment land-use and the discharge and nutrient loading of tributaries that discharge into the Lakes will influence seagrass growth, extent and condition. Indeed, the permanent entrance may have facilitated the maintenance of salinity and light conditions needed to maintain environments conducive to seagrass persistence, particularly near Lakes Entrance.

It likely, but difficult to tell unequivocally, whether the progressive salinisation of the Lakes resulted also in a shift away from freshwater angiosperms (such as *V. australis*) towards seagrasses. As noted above, *V. australis* was formerly common in the main lagoons and in the open-water areas of the fringing wetlands, but is no longer present in these areas, probably because of high salinities (Salter *et al.* 2010*a*). In cases such as these, oral histories can often provide useful leads, as has been demonstrated by the work of Scott (1998) with the Tuggerah Lakes on the mid-central New South Wales coast. The problem in deciphering these changes is that the early oral histories from the Gippsland Lakes containing references to beds of submerged plants (e.g. Ellis and Lee 2002) describe them in broad terms (e.g. 'weed') and it is unclear whether this refers to angiosperms with freshwater or with marine affinities (or, indeed, to benthic macroalgae). Stable carbon isotope data suggest that phytoplankton dominates the carbon deposited in the central basin of Lake King, and that this has been the case for the past 200 years, indicating that seagrass distribution has also probably been similar over this period for Lake King (Holland *et al.* 2013*a*).

Phytoplankton, nutrient loads and algal blooms

Non-toxic phytoplankton blooms are an annual occurrence in the Gippsland Lakes, and are driven primarily by catchment-derived nutrient inputs associated with high runoff. Many blooms typically go un-noticed and are a critical part of the Lake's food web. Although toxic blooms of *N. spumigena* have been reported anecdotally since European colonisation of the region (Holland *et al.* 2013*a*), it seems that blooms – at least those that have been reported – were infrequent before 1986. Since then, blooms of *N. spumigena* have been reported frequently and, as they can be toxic, have led to closure of the lagoons to contact-based recreation, such as swimming, as well as to closing of commercial and recreational fishing, as occurred in 2011–12.

Although nutrient inputs fundamentally drive the productivity of estuaries, excessive loads can drive eutrophication and the formation of algal blooms and poor water quality (Harris 2001; Conley *et al.* 2009). Algal primary productivity in estuaries is typically limited by nitrogen and/or phosphorus. In the case of the Gippsland Lakes, bioassays and nitrogen-to-phosphorus ratios strongly suggest that phytoplankton are nitrogen-limited most of the time (Holland *et al.* 2012). The Gippsland Lakes have a relatively long residence time of ~1 year (Webster *et al.* 2001), and so once nutrients enter the system recycling processes play a critical role in the ecological and biogeochemical response. Inflows, which are typically at their maximum in winter and spring, stimulate rapid blooms of diatoms and dinoflagellates, which are generally not toxic and are rapidly grazed by the resident zooplankton (Holland *et al.* 2012). Phytoplankton cells sink to the sediment and are remineralised, leading to a loss of bioavailable nitrogen from the ecosystem through microbially mediated denitrification. The input of organic matter into the bottom waters in combination with strong stratification during inflow events often leads to anoxia in the bottom waters of Lakes King and Victoria. It is under these conditions that phosphorus stored within the sediments associated with iron oxyhydroxides is released from the sediment (Cook *et al.* 2010). This flux of phosphorus is significant, and can be equivalent to up to ~15 times the phosphorus loads from the catchment, underscoring the importance of internal recycling versus new inputs (Cook *et al.* 2008).

It is this sequence of events that sets the stage for periodic blooms of nitrogen-fixing *N. spumigena*. Low concentrations of dissolved inorganic nitrogen, in combination with relatively high concentrations of phosphorus, provide ideal conditions for the growth of *N. spumigena*, which are able to satisfy their nitrogen requirements via fixation of atmospheric N_2. The growth of *N. spumigena* is also increased by the presence of grazing organisms in the water column, and this is likely to be caused by the liberation by grazers of nutrients as they consume other phytoplankton taxa, making them available to *N. spumigena*, which themselves are unpalatable and remain largely ungrazed (Holland *et al.* 2012). Once established, blooms of *N. spumigena* lead to considerable loss of public amenity owing to their toxicity; if large enough, blooms lead to the lakes being closed to contact (e.g. swimming) and non-contact (e.g. angling) activities.

The ability of *N. spumigena* to fix 'new' nitrogen has raised the question as to the importance of this process as a source of *de novo* nitrogen in relation to catchment inputs. It has recently been shown that N_2 fixation by cyanobacteria may contribute as much as 20% of the total catchment-derived nutrient load into Lakes King and Victoria (Woodland and Cook 2014). In 2007–08, the aforementioned extreme nitrogen loading caused by bushfires in the catchment led to an extensive and long-lived bloom of *Synecococcus* sp. Although non-toxic, this cyanobacterial bloom caused a AU$20 million loss to the Gippsland tourism industry (Connolly *et al.* 2009), as well as seagrass losses. In other estuaries (e.g. Chesapeake Bay, USA: Kemp *et al.* 2005), fish kills have been associated with algal blooms. Over the period 1998–2007, seven fish kills were reported for the Gippsland Lakes (EPA Victoria 2007) and each was ascribed to increased salinity or to increased turbidity; any relationship between algal blooms and fish kills is yet to be established.

From the perspective of water quality and algal blooms within the Gippsland Lakes, the impact of European colonisation is not clear. *N. spumigena* blooms typically occur during low-salinity periods (salinity 10–22), and so the cutting and subsequent maintenance of the artificial entrance may have altered the frequency of blooms of this species from their pre-European incidence. The answer to this question may lie with palaeolimnological studies. Two such studies have been undertaken. In the first, Saunders *et al.* (2008) reported that Lake King has experienced extensive environmental change since European colonisation and the opening of the entrance, but that (on the basis of diatom–salinity transfer functions) algal blooms were likely to have been a natural feature of the Gippsland Lakes. With increased marine influences originating from the permanent opening to Bass Strait, there was a shift from a brackish-water, planktonic diatom flora towards a benthic-dominated, marine-affinity diatom flora. More recently, Holland *et al.* (2013*a*) have argued that cyanobacterial blooms may have been more frequent before the opening of the artificial entrance to Bass Strait than they are at present. The latter study, however, found also a clear increase in indicators of eutrophication since the 1940/1950s, coinciding with an intensification of agriculture and population density within the Gippsland Lakes catchment.

Has there been a change in ecological state in Lake Wellington?

Combined, the above sections provide strong evidence for a switch having occurred in the aquatic vegetation of the

Gippsland Lakes, and especially in Lake Wellington. Submerged angiosperms, including the salt-intolerant *V. australis*, were formerly common across the Lakes, but disappeared from Lake Wellington in 1968 and have been absent ever since (Aston 1977; Ducker *et al.* 1977; Bird 1978). Phytoplankton is now the dominant plant type. The loss of *V. australis* followed a short but severe drought, which undoubtedly increased water-column salinities, and a subsequent bushfire and flood, which, on the basis of detailed studies of more recent events, would have increased nutrient loads and decreased water-column clarity. It is unclear which of these events was the most critical, but it seems that the combination resulted in a switch from a system dominated by rooted, submerged angiosperms to one dominated by phytoplankton. Lake Wellington has remained devoid of vascular plants, and it continues to have high nutrient and chlorophyll *a* concentrations. Whether the changes represent a shift along a simple state-and-transition gradient, or a more complex shift across stable states with complex feedback loops and strong hysteresis in responses, is unclear (see Petraitis (2013) for a description of these alternatives; cf. Scheffer *et al.* 2009). Regardless of the interpretation, it is evident that the previous angiosperm-dominated community has been replaced by a phytoplankton-dominated system and the prior vegetation type is unlikely to return, even if salinities were to fall to pre-European conditions.

Fish and commercial and recreational fisheries

The Gippsland Lakes support a diverse fish assemblage of >60 species, which includes resident, diadromous, and seasonally transient freshwater and marine species (Warry and Hindell 2012). The resident assemblage consists of a variety of species, including small-bodied epibenthics (e.g. eastern bluespot goby, *Pseudogobious* sp. 9; Tamar River goby, *Afurcagobius tamarensis*), pelagic schooling species (e.g. smallmouth hardyhead, *Atherinosoma microstoma*), and large predators (e.g. black bream and estuary perch). Diadromous species use estuaries as a spawning habitat or as a movement corridor through which they gain access to marine or freshwater spawning habitats. Some of the more common diadromous species found in the Gippsland Lakes include southern shortfin eel (*Anguilla australis*), tupong (*Pseudaphritis urvillii*), and common galaxias (*Galaxias maculatus*). Opportunistic marine species such as Australian anchovy (*Engaulis australis*), smooth toadfish (*Tetractenos glaber*), silver trevally (*Pseudocaranx dentex*) and several flathead species (e.g. dusky flathead, yank flathead, *Platycephalus speculator*) are also abundant (Warry and Hindell 2012). The Gippsland Lakes serve also an important nursery function for early life stages of species that inhabit marine or freshwater habitats as adults. Among other species, juvenile tailor (*Pomatomus saltatrix*), eastern and western Australian salmon (*Arripis trutta* and *A. truttaceus*, respectively), greenback flounder (*Rhombosolea tapirina*), and cobbler (*Gymnapistes marmoratus*) recruit to nursery habitats in the Gippsland Lakes.

The diverse community of ecologically and economically important fish species in the Gippsland Lakes is supported, in part, by the structural, bathymetric and physicochemical habitats present within the Lakes. Structural habitats such as rocky reefs, high-relief bathymetry, woody debris, and macrophytes provide physical refugia and foraging habitat for fish, as well as supporting the production of important invertebrate prey species. The relative importance of these habitats for estuarine fishes is species-dependent and can also differ among life stages (Gillanders 1997; Guidetti 2000; Able 2005). Elsewhere in Victoria, seagrass beds have been identified as important nursery habitat for a variety of fish species (Jenkins *et al.* 1997; Jenkins and Wheatley 1998; Hindell 2006), and there is evidence that seagrass beds serve a similar function in the Gippsland Lakes (MacDonald 1992; as cited in table 3 of Gillanders 2006; Warry and Hindell 2012). Seagrass habitat is considered an important nursery habitat for post-settlement juvenile black bream (Norriss *et al.* 2002), whereas macrostructural habitat (i.e. large woody debris) might be more important to adult age-classes within the Gippsland Lakes and its tributaries (Hindell 2007).

The close association between many juvenile-stage fish species and seagrass habitat suggests that historical and recent fluctuations in the extent of seagrass habitat within the Gippsland Lakes has likely influenced community dynamics within the estuary. For example, there is some evidence that commercial fisheries' catches increased during the early to mid-1900s following a historical recovery of seagrass habitats in Gippsland Lakes. Reviewing an earlier study by MacDonald (1992), Harris (1995) stated: '… he (MacDonald) noted the recovery of commercial catches coinciding with the progressive recovery in seagrasses that followed an extensive loss of meadows'. Regrettably, more recent studies overlapping with the 1982 Ramsar listing of the Gippsland Lakes are not available and the net effect of changes in seagrass habitat availability on community composition, species-level productivity, and food web dynamics remains unclear.

The Gippsland Lakes estuarine food web is supported by a diverse resource base derived from both pelagic and benthic production (Warry *et al.* 2013). Recent trophic analyses from the Gippsland Lakes and Port Phillip Bay (also in Victoria) have indicated that seagrass, particularly *Zostera* spp., are an important basal resource for many fish species in these ecosystems (Hindell 2007; Warry *et al.* 2013). For glass shrimp (*Paratya* spp.), tupong and some goby species, stable isotope-based modelling suggests that ~20–40% of the biomass of these species is derived from *Zostera* spp. at some locations within the Gippsland Lakes (Warry *et al.* 2013). The same study found an increased contribution of pelagic nutrition sources to two pelagic fish species, river garfish (*Hyporhamphus regularis*) and smallmouth hardyhead, although these species still appear to assimilate carbon from a variety of sources (e.g. epiphytes, macroalgae).

In addition to pelagic fish species, pelagic primary production in the Gippsland Lakes can be important for demersal and benthic-foraging fish species, via filter-feeding prey. This pathway was shown in a recent study that documented the transfer of atmospheric nitrogen fixed following an intensive, estuary-wide cyanobacterial bloom (*N. spumigena*) to adult black bream via a series of trophic transfers (Woodland *et al.* 2013). There is also some evidence that fixed nitrogen is assimilated by fish during smaller and locally isolated *N. spumigena* blooms; this, however, appears restricted to small-bodied species that are presumably less mobile and

Table 2. Five-year average catch with minimum and maximum annual catches (parentheses: minimum–maximum) in commercial fisheries production in the Gippsland Lakes and Lake Tyers

Note: data have not been corrected for interannual changes in fishing effort. Source: Department of Environment and Primary Industries (DEPI) commercial fishery catch statistics for Gippsland Lakes and Lake Tyers (www.depi.vic.gov.au)

Species	Catch over five-year period (tonnes per year)					
	1981–1986	1986–1991	1991–1996	1996–2001	2001–2006	2006–2011
Carp (*Cyprinus carpio*)	179 (60–367)	358 (262–401)	362 (334–379)	520 (397–658)	409 (251–469)	64 (25–105)
Black bream (*Acanthopagrus butcheri*)	298 (235–446)	212 (167–277)	145 (118–178)	146 (89–181)	53 (26–137)	64 (36–144)
Dusky flathead (*Platycephalus fuscus*)	36 (26–65)	16 (7–28)	7 (4–13)	3 (1–7)	20 (12–48)	24 (16–35)
Australian salmon[A] (*Arripis trutta* and *A. truttaceus*)	9 (2–18)	11 (5–16)	14 (1–25)	14 (8–19)	17 (5–30)	5 (2–10)
Tailor (*Pomatomus saltatrix*)	24 (13–45)	36 (20–58)	19 (8–35)	24 (9–46)	46 (14–71)	22 (12–35)
Luderick (*Girella tricuspidata*)	21 (13–31)	35 (21–49)	24 (13–37)	11 (9–14)	17 (13–21)	19 (8–43)
Yellow-eye mullet (*Aldrichetta fosteri*)	90 (77–106)	112 (79–157)	86 (62–95)	51 (26–82)	42 (22–57)	16 (11–24)
Silver trevally (*Pseudocaranx georgianus*)	19 (11–30)	34 (10–63)	19 (9–39)	14 (11–19)	18 (12–31)	10 (7–13)
Other species[B]	84 (61–115)	139 (45–258)	111 (50–224)	48 (29–62)	57 (46–78)	74 (19–194)
Total[C]	762 (576–934)	955 (741–1178)	786 (654–938)	832 (726–993)	679 (476–805)	286 (212–352)

[A]Includes Western and Eastern Australian salmon.
[B]Includes sea mullet, leather jacket spp., river garfish, estuary perch, Australian anchovy, blue mussel, and others.
[C]Total does not include catch from fishery species with fewer than five licence holders, as per DEPI reporting of commercial fishery catch statistics.

restricted in terms of their spatial foraging (Holland *et al.* 2013*b*). The reticulate structure of the trophic linkage between pelagic toxic cyanobacteria and the benthivorous black bream highlights the complex and tightly interconnected food web present in the Gippsland Lakes. The interconnected nature of the Gippsland Lakes food web indicates that perturbations to the food web resulting from chronic changes in nutrient availability, water clarity and habitat availability (e.g. seagrass beds) are likely to have non-linear and unpredictable effects on fish populations.

As noted in the Introduction, the Gippsland Lakes continue to support highly productive – and valuable – commercial and recreational fisheries. Black bream, yellow-eye mullet, tailor, dusky flathead, and luderick (*Girella tricuspidata*) form the bulk of the commercial harvest (Table 2). Recreational fishing targets these same species and includes other species such as estuary perch, King George whiting (*Sillaginodes punctatus*), and snapper. The most economically important finfish species in the Gippsland Lakes is black bream – commercial catch during 2011–12 was 96 tonnes, representing 87% of the total commercial black bream harvest in Victoria (Kemp *et al.* 2013). Recreational fishing for black bream is also intense in the Gippsland Lakes, and accounts for 20–50% of the state-wide harvest on an annual basis (Department of Environment and Primary Industries 2010).

Even so, the commercial catch of black bream has declined markedly in recent decades; there was a simultaneous reduction in the number of commercial fishing operators, from 38 in 1987 to 10 in 2012, arising from a voluntary licence buy-back scheme (Kemp *et al.* 2013). It is thus unclear to what extent the apparent decline in the fishery reveals a *bona fide* decline in stocks versus a simple decrease in fishing effort. The black bream stock is currently believed to be stable, albeit at markedly lower productivity than in previous decades. The population in the Gippsland Lakes is sustained by periodic strong year-classes that recruit to the adult population (Morison *et al.* 1998). These year-classes are critical for maintaining sufficient spawning stock biomass during unfavourable reproductive years, a phenomenon termed the 'storage effect' (Warner and Chesson 1985; Secor 2007). Black bream recruitment success has been linked closely to flow conditions during the egg and larval stages, and it has been hypothesised that autumn flow conditions are an important determinant of juvenile settlement and survival (Jenkins *et al.* 2010; Williams *et al.* 2012, 2013). Changes to the hydrological regime in the western tributaries of the Gippsland Lakes could affect local reproductive success of species like black bream that have developed life-history strategies that rely on the timing and magnitude of seasonal freshes.

The uncertainty surrounding both the status of the black bream population as well as the effects of environmental conditions on black bream population dynamics is indicative of the wider state of knowledge regarding many of the fisheries resources in the Gippsland Lakes. A recent analysis identified several critical knowledge gaps facing fisheries managers in the Gippsland Lakes, including: (1) primary environmental determinants of spawning and recruitment success, (2) critical habitat identification, (3) threshold freshwater flow volumes needed to maintain suitable water quality, (4) effect of the loss of ecological connectivity between estuarine and upriver reaches, and (5) effects of estuarine constriction on taxonomic and genetic diversity (Department of Environment and Primary Industries 2010).

It is unclear precisely what effect the permanent opening has had on the juvenile production of marine species that recruit to the Gippsland Lakes (e.g. Australian salmon, tailor, flathead spp., greenback flounder). Potential benefits include: (1) an increase in the availability of suitably saline habitats within the estuary; (2) a permanent corridor allowing continuous recruitment to the estuary by marine spawning species; and (3) the creation of a permanent dynamic mixing zone within the channel. Despite these potential benefits, access to productive shallow estuarine nurseries does not necessarily equate to increased population-level productivity and the availability of a permanent opening can also facilitate the presence of piscivorous predators (e.g. dolphins, seals, fish). For example, there is evidence that little terns (*Sterna albifrons*) preferentially forage within the channel near Lakes Entrance, targeting small pelagic fish such as pilchard (*Sardinops neopilchardus*), southern anchovy (*Engraulis australis*), and blue sprat (*Spratelloides robustus*) entering the Gippsland Lakes on the flood tide (Taylor and Roe 2004). Stomach contents and stable isotope data from crested terns (*Thalasseus bergii*) indicate that birds roosting within Gippsland Lakes were also feeding on marine, rather than estuarine, fish (Holland *et al.* 2013*b*). It is not clear whether the crested terns were foraging in the Lakes Entrance channel or the coastal ocean. Increased predation pressure within the estuary or the entrance channel is only one of several possible mechanisms by which the potential advantages of the increased accessibility of estuarine habitats may be offset for species that would otherwise be restricted to coastal marine habitats.

Synthesis and prognosis

Long-term changes in the ecological condition of the Gippsland Lakes

The Gippsland Lakes and their fringing wetlands have changed markedly since Europeans colonised this region of south-eastern Australia in the early–mid 19th century. Driving many of the changes has been the process of progressive salinisation, which in turn has been caused by two fundamental changes to the Gippsland Lakes and the rivers that flow into them. The first is the construction of a permanent opening to Bass Strait in 1889, which converted the Lakes from an intermittently open and closed lagoonal system to one with a permanent (and dredged) opening to the sea. The second is the marked reduction in flow of all of the western rivers that discharge into the Gippsland Lakes, as well as a smaller and possibly trivial reduction in the discharge of the eastern rivers. As a result of these developments, almost all of the formerly freshwater wetlands that fringe the main lagoons and the lower parts of the inflowing rivers have become salinised, with marked changes to plant species composition and to ecological condition (Boon *et al.* 2008; Raulings *et al.* 2010, 2011; Sinclair and Boon 2012). That such changes would occur as a result of the opening to the ocean were predicted nearly 50 years ago by Bird (1966), and indeed the first reports of rapid shoreline erosion taking place with the loss of fringing vegetation was provided nearly a half a century earlier, by Hart (1921).

Impacts are also likely to have occurred on seagrass beds, although these are not as well documented as those for the fringing vegetation of reed beds, swamp paperbarks, coastal saltmarshes, and the formerly freshwater wetlands. The opening of the artificial entrance may have increased the distribution of *Zostera* spp., which is now dominant, possibly at the expense of *Ruppia spiralis* and certainly at the expense of *V. australis*. Some of the variability in seagrass distribution can be explained by obvious events such as algal blooms, which lead to a measurable decrease in seagrass cover. Other factors, such as overgrowth by epiphytes in the nutrient-rich water of the lagoons, may further limit their primary productivity. Although there is anecdotal evidence of the long-term loss of seagrass from the Gippsland Lakes, rigorous monitoring has only commenced in the past decade and the long-term significance and scale is hard to assess.

The same does not hold for algal blooms, for which there is a reasonable monitoring record over the past ~30 years. It is critical to note that algal blooms have most likely always been a part of the ecology of the Gippsland Lakes. The opening of the entrance seems to have initially led to a reduction in blooms. Since the 1940s, however, there is evidence of the re-eutrophication of the lakes, most likely driven by increased catchment nutrient inputs (Holland *et al.* 2013*a*). Since 1986 recurring blooms of *N. spumigena* have led to loss of recreational amenity and periodic closure of the fishery. Such blooms are likely to continue into the future and possibly intensify initially with climate change (Paerl and Paul 2012). Longer-term, rising sea levels, and reduced river flow from the western catchments will most likely see a reduced frequency of blooms as the system becomes more marine.

The diverse fish community of the Gippsland Lakes supports highly productive commercial and recreational fisheries in addition to the trophic demand of local piscivorous birds, mammals and fish. This productivity is supported by a complex and highly reticulate food web as well as a mosaic of biogenic and structural habitat types. Changes to the physical, physico-chemical, chemical and biological environment of the Gippsland Lakes have almost certainly shifted the contemporary fish community away from the pre-entrance state; however, there is very little historical information from which to construct an accurate baseline of pre-entrance conditions. Currently, the fish community of the Gippsland Lakes supports a wide range of anthropogenic and natural functions, and further work is needed to quantify the influence of salinity, river flow, nutrients, habitat availability, harvest, and predation on the dynamics of this fish community.

Implications for management of the Gippsland Lakes Ramsar site

The difficulties to be faced by those charged with managing the Gippsland Lakes Ramsar site over coming decades are sizeable, some might say insurmountable. Rises in eustatic sea levels and increases in the incidence and severity of storm surges are inevitable (McInnes *et al.* 2005). They will increase salinisation pressures within the Lakes and will exacerbate current issues with shoreline erosion and the loss of fringing vegetation (Sjerp *et al.* 2002). As only a narrow sand barrier currently separates the lagoons of the Gippsland Lakes from Bass Strait and the Southern Ocean, overtopping or localised erosion of this outer barrier will have severe consequences for the current ecological structure, function and value of the lagoons and their fringing wetlands. Even if this catastrophic event were not to occur, simple rises in eustatic sea level alone will increase salinities throughout the lagoons of the Gippsland Lakes, up into the lower reaches of the rivers, and into the fringing wetlands (Brizga *et al.* 2011). The severe impact of a deeper penetration of seawater intrusions into coastal floodplains and their wetlands has been demonstrated repeatedly for broadly similar paperbark-dominated wetland systems in northern Australia (Winn *et al.* 2006; Cobb *et al.* 2007; Finlayson *et al.* 2013) and is likely to be just as severe for those of the Gippsland Lakes.

Concurrent with these forces come anthropogenic pressures for intensification of agriculture in the catchment, partly as a response to irrigated agriculture becoming less viable in drier parts of the State to the north and the west. An increase in the intensity and extent of dryland or irrigated agriculture would have major implications for loads of nutrients and sediments to the Lakes. Webster *et al.* (2001) estimated that a 40% reduction in phosphorus loading was required to reduce the incidence of *Nodularia* blooms in the Gippsland Lakes by 20%, and that load reductions of 50–70% would be required to return the Gippsland Lakes to a mesotrophic condition. It has been estimated that a 40% reduction in phosphorus loading would cost ~AU$1.3 billion (Roberts *et al.* 2009). Reductions in nutrient loading of these magnitudes will likely be unattainable without large-scale reductions in the area of land devoted to agriculture in the catchment. A recent study undertaken on the nearby Corner Inlet–Nooramunga marine system of Victoria (also Ramsar-listed) has shown that even with the catchment-wide implementation of current world-best-practice in irrigated and dryland farming, nutrient loading will continue to pose a threat to ecosystem integrity and, especially, to seagrasses (West Gippsland Catchment Management Authority 2013).

Rapidly increasing population growth, especially in the metropolis of Melbourne, will continue to increase demand for water to be extracted from the western Latrobe–Thomson–Macalister river system, which already sees >30% of flows extracted for agricultural, industrial, and domestic purposes. Proposals for a dam on the Mitchell River have been aired for at least four decades, and at least one political party has announced that it wants to again explore options to dam the river, the largest dam-free river in south-eastern Australia. It is likely that calls for such a dam to be constructed will persist into the future, especially if water availability to urban centres decreases as a consequence of climate change and population growth.

Closer to the Gippsland Lakes themselves, urbanisation and development of the shoreline is likely to occur as a result of the sea-change phenomenon that has gripped Australia for recent decades. Already 85% of Australians live within 50 km of the coast and nearly 25% within 3 km of the coast (Australian Bureau of Statistics 2004, 2010). The coast continues to attract residents much faster than does non-coastal Australia, and the Gippsland coast is one of the favoured regions in the State for permanent residence and for holidaying. During the 2013–14 summer, the population of Lakes Entrance and environs was estimated to increase from 6000 to 50 000 (*The Age*, 23 December 2013). These episodic but intense population pressures create an economic environment conducive to additional coastal development and ongoing estuarine degradation.

Fishing pressure from recreational angling is also likely to increase, as populations increase and people move increasingly to the coast to live and to recreate. Already recreational angling is the nation's most common recreational activity, and more than 3.4 million Australians include recreational fishing as part of their lifestyle (Creighton 2013). Blair (2009) summarised the status of the recreational black bream fishery in the Gippsland Lakes and reported the perception of anglers targeting this species in terms of the process of shifting baselines, whereby the present-day experience of the (degraded) fishery is construed to be the 'natural' state, and prior conditions, when fish stocks were far superior, are discounted (Pauly 1995).

Notwithstanding this suite of management challenges, the fundamental problem facing those who manage the Gippsland Lakes is that of ongoing salinisation. On the one hand, increasing salinity may well have advantaged – and will probably continue to advantage – some elements of the biota of the Gippsland Lakes: seagrass beds and the fish species that they support are examples. On the other, it has had a demonstrably negative influence on fringing water-dependent vegetation (e.g. reed beds), on previously freshwater wetlands, on the benthic vegetation of Lake Wellington, and on shoreline erosion. Sea-level rise will probably exert an overwhelming effect on water quality if the Boole Poole barrier system were to be breached by storm surges, and this would lead to a further increase in salinity and reduced residence time. Both may have a generally positive impact on water quality, as salinity-induced stratification of the water column would be reduced and nutrients increasingly washed out of the lagoonal system to the ocean. If salinities increase sufficiently, conditions will become unsuitable for the previously troublesome bloom-forming cyanobacterium *N. spumigena*. Increased forest fires under a drying climate-change scenario may, however, lead to increased blooms of other cyanobacteria such as *Synechococcus* sp., as was observed in 2008. Increased salinity and tidal flushing would likely have a positive influence on water quality, algal blooms, seagrass condition and extent, and probably on fish.

Conclusions

The critical issue facing those charged with managing the Gippsland Lakes is that environmental conditions have changed in the past, are currently changing, and will continue to change into the future. Managers have essentially two options. The first is to accept the altered baseline(s) and let the Gippsland Lakes change in accordance with current and future anthropogenic forces. If this course of action is taken, over long time frames there will inevitably be major shifts in the ecological structure and function of the Gippsland Lakes. These changes will have implications for the site's Ramsar status, as the ecological character that led to the 1982 listing will almost certainly have changed, and likely in a very marked way. The second option is to reject the new baseline(s), and attempt to maintain current conditions or even to revert to some prior ecological condition.

Whether or not managers do intervene to prevent, or at least minimise, change, their actions will inevitably prove controversial. They seem to be caught in a situation that social scientists identify as a 'wicked problem', with no clear means of general resolution (Churchman 1967). In some cases, such as with reducing the load of nutrients received from the catchment, there seems to be no technical solution that will be socially acceptable, even if current best practices of land management are invoked. If more severe intercessions are chosen, the ultimate 'solution' will probably involve some form of whole-of-lakes engineering, at least for the control of salinity. Some have suggested that salt ingress into Lake Wellington could be controlled by placing a lock or similar structure across McLennan Straight, the narrow channel that links Lake Victoria with Lake Wellington, but a recent analysis indicated this was impracticable at an engineering level and was likely to have other – and adverse – ecological consequences (Sinclair Knight Merz 2010). Salt ingress into the Lakes more generally may be controlled by a structure across the entrance to the ocean, but that is likely to prove either unbuildable or unacceptable to users of the port at Lakes Entrance. There are recurring calls for a second entrance to be built west of the current entrance, around Ocean Grange, with the view to reducing flooding and flushing nutrients from the Lakes. This option was investigated by Webster *et al.* (2001), who found that it would lead to only little flushing of nutrients. The intervention, however, would increase water column salinities rapidly following floods, which would likely mitigate *N. spumigena* blooms. Such an option would be costly, possibly lead to the localised loss of migratory bird habitat through scouring (Sinclair Knight Merz 2005), and be highly contentious. Salinisation of the lower reaches of the Latrobe River system and its wetlands might be limited by environmental flows to push the salt wedge further down the estuary, but the use of water in this way is likely to reduce the supply available to other users and, in any case, the volumes required are too great to be delivered with any conceivable infrastructure upstream (Brizga *et al.* 2011).

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