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From the Editors

The papers published in this issue of the journal were presented at the 2013 FNCV Biodiversity Symposium, the theme of which was 'Water and biodiversity'. The issues explored by the presenters, within the theme of the symposium, covered a wide span of research into ecology, human–wildlife interaction, and attitudes to wetlands in an historical context.

The FNCV series of biodiversity symposia fulfils an important function, for both the club and research in natural history generally. For the past 12 years the club has been promoting the value of research into biodiversity issues; in hosting the symposia, it creates opportunities for scientists and other researchers to meet with a public audience and speak about their research. The resulting spread of information assists in the club's primary goal — to understand our natural world.

It is well recognised that wetlands have a vital place within natural environments. Among many functions they help to maintain good water quality in rivers, replenish groundwater, and store carbon. In addition they are sites of high biodiversity value. Regrettably, they are threatened often by human activity and need to be protected. The more we know about the value of these landscape features, the better.

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Front cover: *Limonium hyblaeum*, a weed threatening some coastal wetlands of Victoria. Photo by Maria Gibson.
Back cover: Bird watching from mobile bird hides at Western Treatment Plant. Photo by Michael A Weston. See page 150.

A boggy question: differing views of wetlands in 19th century Melbourne

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Abstract
The site of European settlement in the Port Phillip region was a place of many swamps. For the Indigenous population these features were essential to their way of life, because of the wide diversity of foodstuffs and raw materials they provided. They were the main support for the meetings of large numbers of the Kulin nation that occurred regularly around the top of the Bay. As the immigrant population of Melbourne increased so too were the indigenes excluded from their customary haunts, and were thus eventually unable to maintain their traditional ways. The immigrant settlers viewed the swamps in a different light: they were a source of disease, and of such little regard they could be used as dumping grounds. As Melbourne grew and the need to improve commercial facilities increased, these areas were progressively transformed into a range of other, more culturally useful forms. The ways in which these wetlands played a part in the histories of both Indigenous and settler populations are examined. (*The Victorian Naturalist* **131** (4) 2014, 96–105)

Keywords: wetlands, swamps, Eastern Kulin, Melbourne

Introduction

Swamps or wetlands[1]—whatever term one applies, terrestrial features that comprise more or less permanent but relatively shallow water—have had an interesting conceptual place in western cultures. Perhaps this is because, unlike other parts of the landscape, they did not fit easily into the natural classification of environment as either land or water (Giblett 1996). Wetlands could not be farmed or built upon; but neither could they be put to the same uses as rivers, lakes or the ocean. Historically, this ambivalence has shaped European ways of thinking about such areas and been a major influence in the ways Europeans have interacted with them.

Perceptions of swamps as sources of airborne disease, and as unproductive areas with no potential for useful development were embedded in European culture. Such attitudes had evolved in the countries of origin over many generations but they were given new ground within which to grow following European intrusion into Aboriginal environments. There, the immigrants' perceptions of wetlands—indeed, of the whole of nature—came face to face with a completely different set of beliefs, held by the people whose land was being invaded. Although none of these newcomers realised it, in all parts of the continent Indigenous people viewed all landscape features within their estates, including wetlands, as fundamental and necessary parts of their very existence. Over thousands of years Indigenous Australians had developed a view of their world that saw nature and culture as indivisible parts of the same series of creative acts (Maddock 1974).

With these markedly contrasting perspectives in mind, this paper will consider the uses to which the many wetlands in the Melbourne area have been put. This examination aims at elucidating the place and impact of wetlands in the history of the area, and of providing insights into the differing ethnic perspectives on Melbourne's many wetlands.

The immediate area of Melbourne is a useful historical context in which to consider culturally-based differences in environmental perspective. On the one hand it is a region in which the activities of both Indigenous people and immigrant settlers are well documented; on the other, at the time of settlement wetlands were a common feature in the immediate vicinity. Indeed, much of the surface in the general area was covered by swamps and lagoons.

The specific location chosen for a settlement by Europeans in the Port Phillip area in 1835 was determined largely by the presence of a reliable source of potable water — the Yarra River (Presland 2008). An area on the northern bank of the river, adjacent to a rocky bar that was the limit of tidal reach, represented the most suitable site available for settlement, lightly timbered and sufficiently raised above the flood level. On a visit to the settlement in March 1836 Governor Richard Bourke described it as 'a beautiful and convenient site' (Bourke 1981: 101). As the town grew in size and importance, however, some features of the landscape came to be seen as less than convenient. In the early years of settlement a number of large water-laden swampy areas, adjacent to the river and in its estuary, had virtually determined the shape and spread of the expanding township (Fig. 1). But as Melbourne's population grew and the urban centre began to be transformed into a major mercantile hub, these areas increasingly posed problems for future commercial expansion, and also for the health of the increasing population.

For perhaps thousands of years before the arrival of Europeans, the site of Melbourne and its surrounding area had also presented a convenient place for major gatherings of the Indigenous population. Members of the two local language groups, *Woi wurrung* and *Boon wurrung*, were part of the Eastern Kulin nation, an interconnected cultural bloc of language groups that occupied the area of central Victoria, from the Murray River to Bass Strait (Barwick 1985). At regular intervals people from many Kulin clans had gathered in their hundreds in the very area that was to become the site of European settlement. Such meetings were not uncommon across Aboriginal Australia but all relied on the availability of sufficient resources to support such unusually large numbers for a period of three to four weeks. At the top of Port Phillip Bay those resources were to be found in the same numerous and extensive wetlands that were subsequently viewed with disfavour by the European settlers.

Kulin country

At the time of European settlement south of the Murray River, the central part of what is now Victoria comprised the collective estates of Aboriginal clans that formed the Eastern Kulin language groups. The Kulin Nation was made up of 22 clans, members of which were closely connected by kin, thought and tongue (Barwick 1985; Clark 1990; Presland 2010).

Within the Kulin world every individual was connected, by family and spiritual bonds, with members of clans whose estates were some distance away. These relationships, developed by marriage ties and shared beliefs, needed to be affirmed and confirmed through regular meetings of the clans concerned. Such meetings were occasions to create and re-create alliances and connections to kin and clan; to settle disputes; to exchange goods; and to conduct the necessary business of their world. Because they involved people from across the whole of the Kulin world, these periodic gatherings brought together hundreds of individuals normally resident in estates as far apart as the Yarra River and the lower reaches of the Goulburn River as much as 200 kilometres distant.

The places within the collective Kulin territory at which clans gathered were time-honoured and generally areas of abundant seasonal resources. By an unfortunate chance, one of these localities was the area where Europeans first established themselves in 1835, the area that was to become the township of Melbourne.

Whenever Kulin clans gathered in the region of the Yarra River ahead of a series of meetings, each group set up camp in a particular location, favoured by tradition. In each case the preferred area was adjacent to an extensive wetland. Thus *Watha wurrung* speakers from the Bellarine Peninsula and further west generally camped on the rising ground at the western end of what is now Lonsdale Street, an area that overlooked West Melbourne Swamp. Members of the *Daung wurrung* speaking clans, approaching the Bay from the Goulburn River area, camped to the north of the river, in the area now called Clifton Hill. The *Boon wurrung* took up their customary position in the high ground of the future Botanic Gardens, adjacent to a large billabong on the Yarra; and the *Woi wurrung* set up in the area that would one day include the MCG, from where they had ready access to a number of wetlands along the northern bank of the Yarra.

While these various wetlands were the mainstays of the Kulin world at times of major

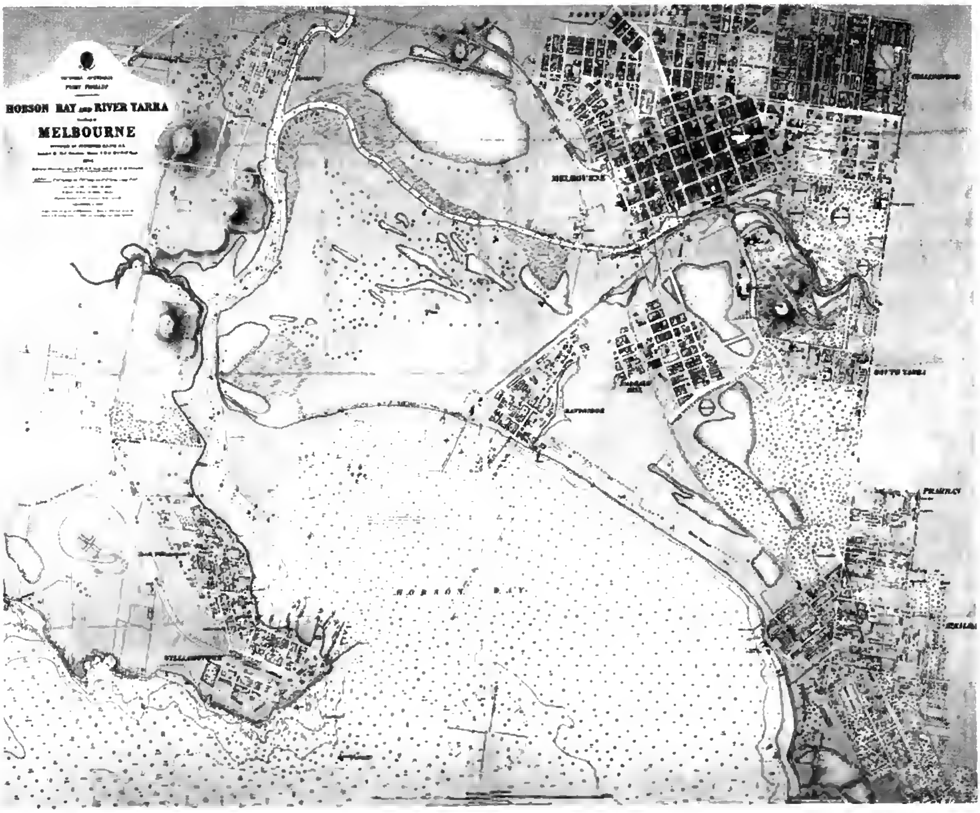

Fig. 1. 1863 chart of the estuary of the Yarra and Maribyrnong Rivers, showing locations of wetlands.

meetings, a number of other wetlands a little further from the river also figured in the traditional seasonal round of local clans. For example, a large area on the southern side of the Yarra estuary, in what is now Port Melbourne, Albert Park, Middle Park and the major part of South Melbourne, was low-lying and consisted of a boggy marshland. This area included a couple of sometimes-connected lagoons that subsequently formed the basis of Albert Park Lake. This and the other swampy areas provided a wide range of seasonally-available resources for the *Yalukit willam* clan of *Boon wurrung*, as did an extensive freshwater wetland in the Elwood area further south along the shore of Port Phillip Bay.

Ecologically, these areas were highly productive and thus the haunt of abundant birdlife (Corrick 1981). At the right time of year—from late spring through to early autumn—the wetlands were favoured spots for the local clans. How productive these areas were can be gauged from an early European description of West Melbourne swamp. Mattingley (1916: 85), writing in the context of what had been lost in the making of North Melbourne, described the swamp in the following terms:

> on the waters of a large marsh or swamp graceful swans, pelicans, geese, black, brown and grey ducks, teal, cormorants, water hen, sea gulls ... disported themselves, while curlews, spur winged plover, cranes, snipe, sandpipers and dotterels either waded in the shallows or ran along its margin, and quail and stone plover ... were very plentiful ... eels, trout, perch inhabited its waters.

Add to this animal abundance a plant regime consisting of dozens of useful plants and it is not difficult to see why it was an attractive place to the Indigenous people. Such areas have been likened by some (e.g. Department of Conservation 2003; Bataluk Cultural Trail 2013) to contemporary supermarkets. Given the range and variety of resources available in a relatively small area this may be a helpful simile. But if we want to continue with this analogy we should perhaps also refer to wetlands as pharmacies and hardware stores, since a number of plants found in such localities were used for purposes other than food. While many plants have parts that can be eaten—rhizomes, flowers, bulbs, tubers and seeds—often, the same species have other parts that can be put to a range of uses. For example, while Cumbungi (*Typha* sp.) was a major food source in the Melbourne area, its leaves and fronds, when dried, could be split into thin lengths for weaving baskets, or making nets and items of personal adornment (Gott 1999; Zola and Gott 1992). Further, a number of aquatic plant species, such as Marsh watercress *Rorippa palustris* and River clubrush *Schoenoplectus tabernaemontani* were gathered, when needed, for their medicinal properties (Lassak and McCarthy 2011).

Settler views of wetlands

Where Kulin clans saw natural provision, white settlers viewed with some disquiet what most saw as unproductive swamps, areas of little value. More tellingly, they saw these elements of the natural world as something that could be amended. For many, the Biblical injunction in Genesis 1:28 to 'subdue the earth' was to be taken literally; indeed, many Europeans took it as the role of humans to improve upon what God had made (Coleman 1996).

Initially, because the wetlands around the settlement were of no practical use to the immigrants, they could be ignored. Although the town quickly achieved city status, its development, particularly on the southern side of the river, was relatively slow. Here a collection of smaller urban areas occupied the higher ground that surrounded the low-lying wetlands (Fig. 1). But following the discovery of gold in Victoria in 1852, Melbourne grew rapidly in size and importance, and attention became increasingly focused on those wetlands. Two concerns were paramount regarding the presence and location of these areas: the health hazard they posed to nearby residents; and the impediments they had become to urban and industrial development.

The former concern stemmed in large part from the prevailing view of the spread of disease. It was a common belief in European society, first voiced by the ancient Greeks and reinforced during the Middle Ages as a result of the widespread occurrence of the plague, that diseases such as cholera and malaria derived from bad air emanating from swamps. From early in the 18th century onwards this belief came to be called the miasma theory. It held sway well into the 1860s, until disproved by John Snow in London, at about the same time that (independently) Louis Pasteur and other researchers demonstrated that germs, not foul air, were the principal agents in the spread of disease (Giblett 1996; Halliday 2001). In Melbourne, the influence of miasma theory continued to be felt for some time. A proposal by John Watson (reported in *The Argus* of 10 February 1866) to drain West Melbourne Swamp suggested that such action would diminish 'the risk of epidemic by destroying what is now a copious source of miasma'.

Over and above their belief that marshes and swamps were the sources of disease, for many of Melbourne's residents these areas were places of little regard. The prevailing attitude to such features is clear from the way in which the wetlands of inner Melbourne were misused. By the mid 1870s, Sandridge Lagoon, for example, had become a cesspit, filled with dumped rubbish, and sewage from adjacent councils (Buckrich 2002). Although such blatantly insanitary use of wetlands only contributed to the threat to public health, it was difficult putting a stop to it. The usual means of policing the regulations set out in legislation were generally inadequate (Dunstan 1984). More than 20 years later, although drainage into Sandridge Lagoon from surrounding areas had been improved, manure from thousands of horses docked nearby was being dumped into the water of the remaining part of the Lagoon (U'ren and Turnbull 1983).

The parlous state of Sandridge Lagoon was mirrored by most of the wetlands within the Yarra's estuary; it was a function of the low esteem with which such areas were regarded, and the fact that they had been neglected for decades. This was easy to do when there was no pressure to make use of the areas. By the early

1870s, however, there was a push for more urban and industrial development close to the city, and the low-lying areas that were previously ignored by governments became the subject of a Royal Commission on the Low Lands (Dunstan 1985).

The Commission was an indication of the growing concern that had arisen within the city regarding wetlands and their future. The enquiry became the first step in a lengthy process of improving those areas that had to date been not worth worrying about. Wetlands that were being encroached on by urban development were seen now as potential impediments to the expansion of further industry or housing. In the succeeding 30 years the landscape of Melbourne was changed dramatically, as all of the wetlands were drained, or transformed into an amenable form. The result of these endeavours takes a number of forms, varying according to the location of each area and a range of determining social factors. These elements are detailed briefly in the remainder of this paper, along with the means and methods employed in the process of creating a desirahle urban landscape.

The reclamation of swamps

In the period from the mid 1870s through to the earliest years of the 20th century almost all of the swampy areas within a two kilometre radius of central Melbourne were reclaimed. About a dozen wetlands were affected, all within the combined estuary of the Yarra and Maribyrnong Rivers, or along the Yarra itself. With a few exceptions, nothing now remains to indicate the former presence of any of these wetlands. In three cases, however, although the wetlands are long gone there are hints that point to the fact that they were once part of the Kulin world. The fate of these three features, plus that of the largest of the swamps in the inner Melbourne area, are considered here, and used as examples of the range of outcomes achieved as a result of European exploitation of the Port Phillip environment.

West Melbourne Swamp

This swamp lay on the northern side of the Yarra within the estuary of the river, immediately to the west of the settlement. At about 30 hectares, it was the largest wetland in close proximity to the settlement (Presland 2011). Because of its proximity to Batman's Hill, this wetland was often called Batman's Swamp. In the early years of the settlement it was seen by a young boy as an attractive feature in the landscape. About 70 years later Gordon McCrae (1912: 117) wrote in glowing terms of what had he had seen in the 1840s but had long since disappeared:

> ... stretching away from the base of the Flag-staff Hill, lay a beautiful blue lake ... a real lake, intensely blue, nearly oval, and full of the clearest salt water; but this by no means deep. Fringed gaily all round by mesembryanthemum (*vulgo*, "pig's face") in full bloom, it seemed in the broad sunshine as though girdled about with a belt of magenta fire.

Most of the area in this part of the estuary was covered by a Brackish Grassland (EVC 934)[2] that was subject to tidal inundation and regular flooding. This grassland enclosed a Brackish Lake Aggregate (EVC 636), which was rimmed by an area of Coastal Saltmarsh (EVC 9).

Away from the edge of the saltmarsh the ground surface of the surrounding grasslands was firm enough to be used as the venue for the first race meeting in the settlement. This took place on 8 February 1837 (Boys 1959).

Although plans to drain the swamp had been made as early as 1849 (*The Argus* 24 April 1849: 2), little work actually took place before the late 1870s. Through the 1850s and 60s a part of the area was given over to cow-herders, and boiling down works and a bone mill were established along the river on the southern portion of the wetland. There were periodic complaints over many years about the offensive smells from the fellmongery and boiling-down works (*The Argus* 13 September 1864: 7) and the stench of liquid mud in the swamp (*The Argus* 5 June 1871: 4). Pollution was also coming into the Swamp from tannery yards in Flemington, via Moonee Ponds Creek (Lack 1985). The fact that an area of several acres of the north eastern corner of the wetland continued to be used as a refuse depot into the 1890s, did nothing for the cause of clean air or a healthy environment (Dunstan 1984).

The biggest impetus to the reclamation of West Melbourne Swamp was the creation of a Harbor Trust in 1877. Although the area was not within the jurisdiction of the Trust (Buckrich 2002), it was soon impacted by the Trust's primary purpose, which was to plan the improvement of Melbourne's port facilities (Dunstan 1984). One of the earliest actions of the Trust was to engage an English engineer, Sir John Coode, to advise on how to solve the associated problems of Melbourne's narrow and circuitous river and a lack of adequate docking facilities (Bentley and Dunstan 1996). Coode's scheme, when put into effect, shortened the river by about 1500 metres and led to the use of a part of the wetlands for the new Victoria Dock. The Coode canal opened in September 1886 and the docks in 1893. The reclamation of the swamp was still going on in 1905 when the Melbourne Harbor Trust established a depot in the old stream bed of the Yarra to store material it had dredged from the Bay, for use in raising the level of the West Melbourne Swamp. (Melbourne Harbor Trust 1905).

Today the area formerly covered by West Melbourne Swamp contains a number of urban features of comparatively recent construction, including the Docklands Precinct; the extension of Moonee Ponds (constructed in the early 1890s to facilitate the barging of coal to ships moored in Victoria Dock); the Melbourne Freight Terminal; and Melbourne Wholesale Fruit and Vegetable Market.

Sandridge Lagoon

During its survey of the perimeter of Port Phillip Bay in 1803 the Grimes exploratory party 'came to a salt lagoon about a mile long and a quarter of a mile wide; had not entrance to the sea' (Flemming 1972: 27). Following settlement in 1835 and the subsequent development of a village adjacent to the sand ridge on the shore line, this feature—possibly a remnant of a former course of the Yarra—became known as Sandridge Lagoon. (Fig. 2) Because of its shape and closeness to the Bay, in the earliest period of the urban development of Sandridge (later Port Melbourne) the lagoon was a defining and determining feature of settlement in the immediate area (U'Ren and Turnbull 1983).

During times of flood the lagoon filled through runoff from the surrounding area, and sometimes (as in December 1849) the water broke through to the Bay (*The Argus*, 4 December 1849: 2). This passage through the sand ridge allowed small boats to find shelter in the lagoon but this was usually a temporary thing. With an entrance to the Bay, the water level within the lagoon was subject to tidal influence; at times of high tide the extent of the lagoon could be a much as 27 acres (U'Ren and Turnbull 1983).

Fig. 2. Detail of the 1863 Cox chart, showing the location and extent of Sandridge Lagoon.

By the mid 1860s pollution in the Lagoon had begun to be a concern to residents. The Sandridge Council felt that a channel cut from the Lagoon into the Bay would solve the issue of accumulated pollution, much of which, it was recognised, came as run-off from Emerald Hill (U'Ren and Turnbull 1983). As well as the health issues raised by the presence of noxious material in the Lagoon, the area posed a danger to people who accidently wandered into the marsh. In November 1869, two men drowned in the Lagoon in the space of as many days. But while these and other human bodies were always removed from the water, such was apparently not the case with goats and dogs that drowned there (*The Record and Emerald Hill and Sandridge Advertiser*, 19 November 1875: 3).

There had been repeated calls since 1854 for a dock and harbour to be created in the lagoon (*The Argus* 16 March 1854: 5), with little progress to that end. In 1869 Sandridge Council offered a prize of £100 for the best and cheapest solution to the unsanitary condition of the lagoon (U'Ren and Turnbull 1983). This produced no workable scheme and by 1879 the only change to the lagoon was that the northern end had been filled in as far as Bridge Street; the rest was still something of a cesspit. With the creation of the Melbourne Harbor Trust in 1877, the lagoon became a political football, its fate bouncing around between the Trust, the colonial administration and the Councils of Sandridge and neighbouring Emerald Hill. In July 1886 an agreement was reached between the players that saw the lagoon mouth opened to the Bay as far as Graham Street and drainage from streets diverted away from that portion (Hoare 1927).

Pollution continued to flow into the lagoon, however, and in 1890 a pumping station was constructed to pump the sewage and drainage past the lagoon to assist in keeping it clear (U'Ren and Turnbull 1983). This had limited success but ultimately contributed to the shoaling of the small harbour (Hoare 1927).

In 1921 a breakwater was built adjacent to the outlet of the lagoon, providing shelter for small vessels. The lagoon was thus made redundant and was finally closed (Hoare 1927). The only easily visible clue to the previous existence of the lagoon is the appropriately-named Lagoon Reserve, which lies between two of Port Melbourne's streets that are suggestively named Esplanade East and Esplanade West.

Albert Park

Albert Park is perhaps best known today as the venue of the Australian Formula I Grand Prix, staged around the perimeter of the Albert Park Lake. That the area still exists as a public reserve, albeit one that is annually given over to well-heeled people with fast cars, when most other similar areas in the inner city have been given over to urban or industrial use, is interesting in itself.

On Robert Hoddle's map of North and South Melbourne in 1842, the area of Albert Park was shown as containing not one piece of water, but several swampy water-holes with a bordering of teatree (*Leptospermum* sp.) (Sutton 1912). The vegetation in these perennially wet areas comprised Brackish Lake Aggregate (EVC 636), which was enclosed by Brackish Wetland (EVC 656). Perhaps because it was an open area and there was little urban development in the vicinity, it had an appeal not normally accorded to other wetlands. Late in 1844 a petition was made by the Melbourne City Council to the Lieutenant Governor, CJ La Trobe, to have an area south of the Yarra set aside as a public park.

Nothing happened immediately but the area was permanently reserved by the government in 1864 and given the name Albert Park, to commemorate the recently-deceased Consort of Queen Victoria (Barnard and Keating 1996). Previously, the area was known as South Park and was administered by the adjacent Emerald Hill and St Kilda councils.

Albert Park was destined to be a public park, so the lagoons were transformed into a permanent lake, rather than being drained as was done in other wetland areas. The impetus to create this lake was twofold: firstly it was a way of dealing with the increasingly unhealthy state of the swamps; and secondly it was intended to create a place where boating activities could be pursued. The two major lagoon areas were deepened and joined to make the lake. It was found, however, that over the summer months the water level in the lake became too low for sailing. Boating enthusiasts petitioned parliament and from 1877 until 1892 the level was kept topped up by piping water from the Yarra River via a steam pump installed in the Domain (Lamb 1996). This pump was superseded in 1892 by a newer installation at Dights' Falls, and was the same that supplied fresh water to the Botanic Gardens, over the same period.

By the latter part of the 1870s, although all the land between the park and St Kilda Road was in private hands (Barnard and Keating 1996), the park itself was reserved for public use in essentially the form it retains today.

Yarra River Lagoon

In March 1846, when CJ La Trobe selected a site for Melbourne's Botanic Gardens, the area he chose on the south side of the Yarra River was an 'indefinite swampy tract' (Gross 1956: 76). The major feature of the area was a billabong, which was connected to the Yarra River (Fig. 3). Unlike the Sandridge Lagoon and other swamps within the estuary of the river, this was a freshwater lagoon, rejuvenated through periodic flooding from the Yarra River. The

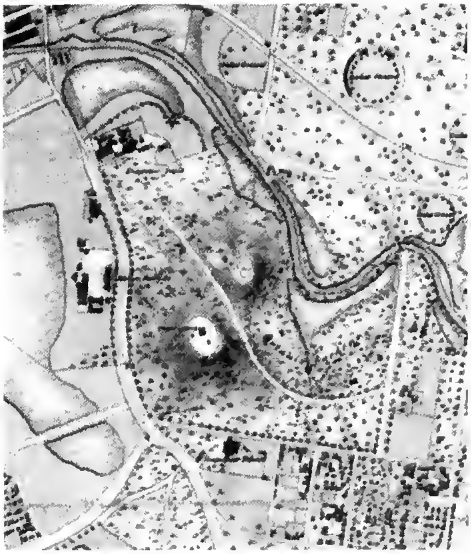

Fig. 3. Detail of the 1863 Cox chart, showing the location and extent of Yarra River lagoon.

vegetation within the billabong was that of a Reed Swamp (EVC 300).

Between 1837 and 1841, Langhorne's Anglican mission had operated immediately to the east of the chosen site, to take advantage of the regular gatherings of *Boon wurrung* who made use of the lagoon (Cannon 1982; Pescott 1982). In addition to a range of plant resources the lagoon was noted for its abundant birdlife. A count in the mid-1850s listed 63 land- and 19 water species of birds (Wilson 1857). Because it was connected to the river, the lagoon also held a regular supply of eels and other fish species. Eels were taken by Aboriginal men using spears, the hunter having detected their presence while wading in the water. George Robinson recorded in his journal (29 January 1841) seeing two Aborigines catch about forty pounds (18.2 kg) of eels in a very short time.

As the lagoon was connected to the river it was prone to flooding, but from the beginning it was incorporated into the general plan of the Gardens. It was subject also to silting up through run-off from the adjacent slope; as a result it needed to be deepened in 1850 (Pescott 1982). Ferdinand Mueller, who was appointed Director of the Gardens in August 1857, left it essentially unchanged except for the installation of a fountain on an island in the lagoon. His successor William Guilfoyle developed the area in ways that eventually led to its becoming a lake. The lagoon was to be deepened again, the height of one of its islands was raised, and the lagoon's edges were shaped (Pescott 1974). Although the work to give effect to these plans was begun in the late 1870s it was delayed and spread over a couple of decades through government work on the river itself. Much of the surface along the southern side of the river in this area, including parts of the Gardens, was low-lying and subject to inundation when the river rose. A government enquiry recommended the straightening of the river in the stretch adjacent to the Gardens. This led to the passing of the *Yarra Improvement Act* 1896 and, in turn to the carrying out of 'improvements' to the river. These were completed in 1901, breaking the nexus between the lagoon and the river, since when it has effectively become a lake (Pescott 1974).

Conclusion

The area at the top of Port Phillip Bay, chosen as the site of European settlement, was one that contained a number of wetlands. For the local Kulin clans these places were essential sources of a wide range of foodstuffs and materials; for the immigrant settlers they were generally disregarded as places of little economic or social value and increasingly used as dumping grounds. Such contrary views of these wetlands reflected the nature of the connection each group had with its physical environments. Notwithstanding this disparity, however, these landscape features can be seen as a significant factor in the local history of both groups.

The enduring presence of Europeans across Victoria ultimately spelt the end of a way of life for the Indigenous people. In the area of Melbourne, as increasing numbers of whites settled so more and more were Aboriginal people excluded from the areas that had sustained them over thousands of years. The wetlands of the Port Phillip region had been a significant resource base during the customary seasonal round. Of greater importance, the presence of these wetlands had supported the regular meetings of clans from across the entire territory of the Kulin. These gatherings were an essential element in maintaining the Kulin world, providing opportunities for a range of necessary activities such as exchange of marriage partners and the enacting of religious and initiatory ceremonies. Denial of access to these areas of abundant resources was thus a blow against not only the material underpinnings of the Kulin world but also against its spiritual and conceptual bases. The exclusion of Aboriginal people was a major factor in the collapse of local Indigenous culture.

The rapid expansion of the city during the 1850s following the discovery of gold put enormous pressure on the wetlands areas around Melbourne. On the one hand decades of neglect had allowed many of these areas to become the repositories of sewage and waste; on the other, the location of some of them was pivotal in the future development of Melbourne's shipping facilities and urban expansion. Europeans beliefs regarding nature in general, and the character of swamps in particular, lead to these areas becoming a focus of attention in efforts to create a landscape that was amenable to the commercial and cultural interests of the city. Significant parts of the inner Melbourne area have been shaped as a result of the countermeasures enacted to achieve these ends.

Although many of the wetland areas in the inner Melbourne area were drained and filled in to be completely lost to view, a few were transformed and continue to have a presence in the urban environment. Of the four wetland areas considered here, ultimately three were turned into areas of public access and recreation. Today they survive as markers of a past, but not-quite-forgotten, landscape form. It could be argued too, that despite the vastly different ways in which wetlands were appreciated and used by the immigrant population, in some fundamental sense these same areas continue to function as they did before settlement. For example, today when large numbers of people get together at Etihad Stadium, to take part in social or ritual activity, it should be remembered that the stadium is located close to where West Melbourne Swamp once stretched. And that in Aboriginal times large numbers of people also gathered there, for not entirely dissimilar purposes (Presland 2002). The swamps of Melbourne may be gone but, in effect, something of their social function remains.

Notes

[1] While distinctions (based on various analyses) can be drawn within a range of landscape features, variously referred to as 'fens', 'marshes', 'swamps', and 'aquatic wetlands' (see for eg. Corrick 1981; Greb and DiMichele 2006), in this paper the more general terms 'wetland' and 'swamp' are used throughout. Both of these terms are taken to refer to 'any area of low-lying land where the water table is at or near the surface most of the time, resulting in open-water habitats and waterlogged land areas' (Jones *et al.* 1992: 441).

[2] Three of the four wetlands examined in this paper—West Melbourne Swamp, Albert Park and the Yarra Billabong—have been categorised within the system of Ecological Vegetation Classes (EVCs) devised by Oates and Taranto (2001). The terms used to describe the vegetation in these wetlands is drawn from that work.

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One Hundred and Two Years Ago

EXCURSION TO COODE ISLAND.

By C. FRENCH, JUN., AND J. R. TOVEY

INTERESTED by the remarks on the flora, &c, of Coode Island in the Naturalist for July last (Vict. Nat., xxviii., p. 57), a party of eighteen boarded the motor boat at Queen's Bridge on Saturday afternoon, 23rd March, en route to the island. Shortly after arrival there two more members joined the party. All expressed surprise at the large amount of foreign vegetation that was growing there. It was noted that most of the plants found flowering there during previous trips made by the leaders in such periods of former years as October to December, also June, were found in flower, thus evidently showing that the exotic plants had not as yet settled down to our seasons. It was also pointed out that, whilst the foreign shells collected there were mostly of North American origin, the plants were mostly South African, and in a few instances European or Asiatic, but none of North American origin, thus showing that the North American ballast came from the seashore, and that from South Africa apparently from further inland.

About twelve plants which are either recognized as native or naturalized aliens in other parts of the State, but not previously recorded for the island, were collected. Amongst these might be mentioned a variegated form of the Red Goosefoot, *Chenopodium rubrum*, L., and the Caltrops, *Tribulus terrestris*, L. Four species not previously recorded as introduced in Victoria were obtained—*i.e.*, *Mercurialis annua*, L., Annual Dog's Mercury, Euphorbiaceae; a native of Europe and North Africa. *Aizoon rigidum*, L., var. *angustifolium*, Sond., Rigid Aizoon, Ficoideae; indigenous to South Africa. *Hermannia velutina*, D.C., Velvet Hermannia, Sterculiacess; another stranger from South Africa. *Abutilon indicum*, Sweet., Indian Lantern-flower, Malvaceae; a native of the tropical regions, also found in South Africa.

From *The Victorian Naturalist* **XXIX,** p 5, May 9, 1912

Rehabilitating wetlands in the Gippsland Lakes Ramsar site: the benefits and limitations of community involvement

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Abstract
The Gippsland Lakes are one of Australia's most important and highly used estuarine complexes. Development pressures have caused the degradation of many of the Ramsar-listed, brackish-water wetlands that fringe the Gippsland Lakes. Two developments stand out as critically important: i) the creation of a permanent opening to Bass Strait in 1889 at Lakes Entrance; and ii) the increasing extraction of freshwater from the Latrobe, Macalister and Thomson Rivers for use by irrigators and industry and as a supply of potable water for Melbourne. One of the largest of the Ramsar-listed wetlands in the Gippsland Lakes—Dowd Morass—was the subject of an intensive rehabilitation project from 2003 to 2007. The project benefited greatly from community involvement, particularly with revegetation efforts and in the re-establishment of more natural wetting and drying regimes. The benefits and limitations of community involvement are discussed, with a focus on community perceptions of wetland values and degradation; the impacts of carp and of exotic weeds are two cases where there was little congruence between the perceptions of community groups and those of professional aquatic ecologists. The wider use of conceptual models is one way to improve understanding and enhance collaboration across different groups in large, multi-disciplinary projects that seek to rehabilitate high-value coastal wetlands. (*The Victorian Naturalist* **131** (4) 2014, 106–114)

Keywords: *Melaleuca*, *Phragmites*, restoration, salinity, water regime

The Gippsland Lakes: their economic, social and environmental values

The Gippsland Lakes, currently Australia's largest navigable inland waterway, are located on the south-eastern coast of Victoria in southern Australia. They consist of three large coastal lagoons (Lake Wellington – 148 km^2; Lake Victoria – 75 km^2; and Lake King – 98 km^2), fed by seven rivers, plus an extensive mosaic of fringing wetlands that line the edges of the lagoons and the lower reaches of the inflowing rivers. The lagoons have a shoreline of 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 entire complex of lagoons, wetlands and the lower reaches of the main rivers is listed under the Ramsar Convention as the Gippsland Lakes Ramsar site, which covers an area of 60 015 ha.

The Lakes provide a wide range of ecosystem services and are a vital component of the Victorian economy. They support Victoria's largest commercial fishing fleet and the single largest recreational fishery in the State for the iconic Black Bream *Acanthopagrus butcheri* Munro 1949 (Department of Primary Industries 2006). 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 the tourism industries they support. It has been estimated that in 2006, the Lakes attracted over 4 500 000 visitor days, including nearly 2 400 000 spent in overnight visits and nearly 1 500 000 in local day visits (URS 2008). URS (2008) calculated the non-market value of four of the large wetlands in the Gippsland Lakes Ramsar site (Clydebank Morass, Dowd Morass, Heart Morass and Sale Common) at $1.15 million (in 2006 dollars) for their biodiversity value alone.

Nevertheless, there are significant threats to these very high environmental, social, and economic values. The decline in Black Bream catches over the past 40 years has been stark (Department of Environment and Primary Industries 2010), critical water-quality objectives, for example for Dissolved Phosphorus and for Dissolved Oxygen, regularly exceed the objectives set out in the State Environment Protection Policy (Waters of Victoria), and recurrent algal blooms are a worrying phenomenon (Environmental Protection Authority [EPA] Victoria 2013). Threats to water quality come from a number of sources, including high nutrient and sediment loads from the inflowing rivers, bank erosion, and the decline of fringing vegetation. One process leading to these changes in environmental condition has been progressive salinisation of the lakes and their fringing wetlands, which in turn is due to two factors. The first is the construction of a permanent opening to Bass Strait in 1889 at Lakes Entrance, which converted the Lakes from an intermittently open and closed lagoonal system to one with a permanent (and dredged) opening to the sea (Bird 1966). The second is the marked reduction in flow in the rivers that discharge into the western parts of the Gippsland Lakes (Moroka 2010). Over one-third of all flows is currently extracted from the Latrobe River (and its tributaries, the Thomson and Macalister Rivers), for agricultural, industrial, and potable use. As a result of both processes, many of the wetlands that fringe the lower parts of these rivers have become increasingly saline, with marked changes to plant species composition and to the ecological condition of vegetation communities (Boon *et al.* 2008; Raulings *et al.* 2010, 2011; Robinson *et al.* 2012; Salter *et al.* 2007, 2008, 2010a, b; Sinclair and Boon 2012). That such changes would occur as a consequence of increasing salinisation were predicted nearly half-a-century ago by Bird (1966).

From 2003 to 2007 I led a large, multi-disciplinary research and development (R&D) project to rehabilitate Dowd Morass, one of the largest of the Ramsar-listed, brackish-water wetlands that fringe the western parts of the Gippsland Lakes. The selection of Dowd Morass was predicated on three requirements: i) the wetland had to be of demonstrable high value (e.g. Ramsar-listed); ii) it had to be amenable to a landscape-scale manipulation of water regimes; and iii) the undertaking had to be explicitly embedded within a strong program of regional natural-resource management, including the involvement of local management agencies and communities in the project and in the implementation of its major findings.

Dowd Morass was ideal because it had been subject to very tight control of its wetting and drying cycles over the past 30 years: since the mid-1970s, it had been kept almost permanently inundated to provide for waterfowl hunting, to allow for breeding of colonial-breeding waterbirds, and to prevent intrusions of saline water from the adjacent Lake Wellington. Moreover, a series of levees had been constructed inside the morass in the mid-1970s when it was in private ownership; these allowed us to implement a landscape-scale manipulation of water regime to determine whether the ecological condition of the wetland could be improved by re-introducing more natural patterns of wetting and drying. Finally, the local community, both at the level of government agencies, such as Parks Victoria and the East and West Catchment Management Authorities, and in terms of local organisations such as Field and Game Australia and the Field Naturalists Club of Victoria, was deeply involved in the project. This paper analyses some of the complex interactions that occurred among community groups, government agencies, and scientifically trained ecologists, and draws out the strengths and weaknesses of community involvement in such wetland-rehabilitation projects.

Dowd Morass

A brief history of Dowd Morass and alterations to its hydrology

Dowd Morass, at 1500 ha, is one of the largest wetlands within the Gippsland Lakes Ramsar site and is located on the southern bank of the Latrobe River, near where the river debouches into Lake Wellington, at the western end of the Gippsland Lakes. Scottish Highlanders were among the first Europeans to explore and settle Gippsland (Watson 1984) and were probably responsible for naming many of the wetlands as 'morasses'.

Land around Dowd Morass was being privately developed by 1888, and by 1942 almost all the wetland was in private ownership (State Rivers and Water Supply Commission 1972). A series of levee banks approximately 0.9-1.9 m Australian Height Datum (AHD) was constructed within the wetland in 1973 'with a view to drainage and development for agricultural purposes' (State Rivers and Water Supply Commission 1972: 1), as well as to prevent overbank flows from the Latrobe River and to prevent brackish water from Lake Wellington entering the western side of the morass (Keith Heywood, local landowner, pers. comm.). The largest of the levees—Heywood's embankment

—runs north–south and almost completely separates the eastern from the western sections of the wetland. Two artificial drains were constructed in the early 1970s to establish a hydraulic connection between Dowd Morass and the Latrobe River.

In 1975 the State Government purchased Dowd Morass as a State Game Reserve, and breaches were created in the levees to improve internal water circulation. The morass was flooded in 1975 from the Latrobe River, through the larger of the two artificial drains dug earlier in the decade. This flooding was intended to reduce saline intrusions from Lake Wellington as well as to provide better opportunities for waterfowl hunting (A Schulz, Parks Victoria, pers. comm.). The drains at that time did not have structures to allow control of the direction or volume of water flow from the river.

Except for three short periods, the wetland has been kept fully flooded almost continuously since this flooding in 1975. The first was in 1983, during a nationwide drought, when the morass would have dried completely but for the digging of a channel by persons unknown and the flooding of the wetland with moderately saline water from the Latrobe River. The second was during an experimental draw-down of water levels in 1997/98 by Parks Victoria, the state government agency that manages the wetland. The third, during the summers of 2003/04 and 2004/05, was the intervention we implemented as part of the trial to reintroduce more natural wetting and drying regimes into the wetland (Raulings *et al.* 2010, 2011).

In 1987 the existing drains were repaired by Parks Victoria with help from the regional Field and Game Australia hunting club, and the structures were further improved in 2000 when stainless steel flaps were fitted to the culverts to allow water movement into and out of the wetland from the river to be controlled. From 1987 to 1997 the morass was managed primarily to maintain stable water levels for colonial breeding waterbirds, by not drawing down water levels from September to December each year. The resultant water regime was not inconsistent with earlier intentions to maximise opportunities for duck hunting and to prevent intrusions of saline water from Lake Wellington, but is markedly different from what was probably the natural hydrology of the wetland. It is thought that prior to European settlement Dowd Morass would have filled with sediment- and nutrient-laden fresh water from the adjacent Latrobe River during winter-spring floods, and with less turbid water of variable salinity from Lake Wellington when wind conditions were suitable and when the water level of Lake Wellington exceeded the shoreline level of Dowd Morass (Sinclair Knight Merz 2003). Surface-water levels in the wetland would therefore have fluctuated in response to complex seasonal patterns of precipitation, runoff and evaporation, as well as with the state of the intermittent opening to the ocean, just north of today's township of Lakes Entrance. It has been proposed too that prior to European settlement Dowd Morass and other wetlands in the Lake Wellington wetlands complex would have dried out about once every five years (Parks Victoria 1997). To my knowledge, no palaeobotanical studies have been undertaken to lend support to this notion (the palaeoecological work of Saunders *et al.* (2008) was concerned with Lake King).

Impact of altered hydrology and salinity on wetland vegetation

The dominant vegetation in Dowd Morass is a woodland of Swamp Paperbark, *Melaleuca ericifolia* Sm, which covers about 500 ha (Boon *et al.* 2008). Swards of Common Reed *Phragmites australis* (Cav.) Trin. ex Streud, cover about 350 ha, and the remainder of the wetland is open water or bare mudflat. *Melaleuca ericifolia*, a clonal wetland shrub or tree, is the dominant woody plant in many coastal and near-coastal wetlands across southern Australia, where it forms a vegetation community known as Swamp Scrub which is critically important as nesting and roosting habitat for colonially breeding waterbirds (Bird 1962; Corrick and Norman 1980; Cowling and Lowe 1981).

A large rookery of colonially breeding waterbirds, consisting mostly of Australian Ibis *Threskiornis molucca* Cuvier 1829, Straw-necked Ibis *Threskiornis spinicollis* Jameson 1835 and Royal Spoonbills *Platalea regia* Gould 1838, is located in the south-west corner of the morass. Ibis have bred in Dowd Morass since at least 1961, with notable breeding periods occurring in 1965 to 1970 inclusive, 1974, and

1978 (Cowling and Lowe, 1981). The birds roost and breed almost entirely in the Swamp Paperbark woodlands.

In the mid-1990s, Parks Victoria staff were concerned that the near-permanent inundation of Dowd Morass since 1975 was having adverse impacts on the condition of *M. ericifolia* and thus on the breeding success of ibis and spoonbills. It is likely that the altered water regime would have contributed to declining condition of *M. ericifolia* woodlands, as *M. ericifolia* seed cannot germinate under flooded conditions (Ladiges *et al.* 1981; Robinson *et al.* 2006); nor can young seedlings establish (Salter *et al.* 2007, 2010b). The growth and survival of established plants is also compromised by deep, permanent water (Morris *et al.* 2008; Salter *et al.* 2010a). Concerns that the existing water regime was responsible for widespread degradation of *M. ericifolia* in the critical ibis rookery area prompted Parks Victoria to partly drain the morass in March 1997. In the winter of 1998, however, the eastern rivers that flow into the Gippsland Lakes were in flood and their large discharge caused saline water to back up in Lake Wellington, as the entrance to Bass Strait at Lakes Entrance was too small to let flood water pass from the swollen Lakes Victoria and King out to the sea. Saline water was thus pushed into the western parts of the lakes and eventually up into Dowd Morass via the shallow opening between it and Lake Wellington ('The Dardanelles') and via over-bank flow along the lower Latrobe River once it too had backed up from Lake Wellington (Sinclair Knight Merz 2003). This saline water then flowed into the partly drained Dowd Morass, exacerbating the prior hydrological problem with a possibly even more severe salinity problem.

Water levels in the morass have been kept high since the temporary draw-down of 1997/98, in order to prevent a repeat occurrence of saline intrusions, to provide stable water for waterfowl hunting, and to provide habitat for the colonial breeding waterbirds. But the problem remains, that the *M. ericifolia* woodlands are likely to degrade further because of the inappropriate hydrological regime, probably exacerbated by chronic and/or acute secondary salinisation processes. Regional natural-resource managers also face the complication that Dowd Morass, like many other wetlands along the Gippsland coast, overlays potential and/or active acid sulfate soils created as a result of the wetlands' coastal position and recent (10 000 years BP) history of post-Holocene sea level changes. The presence of these soils makes any attempt to reintroduce more natural wetting and drying regimes even more fraught.

Community involvement in rehabilitating Dowd Morass

Community attitudes to wetlands of the Gippsland Lakes

Immediately after the R&D project started in early 2003, a series of workshops were conducted with two of the most relevant community groups in the region: Sale and District Field Naturalists, and Field and Game Australia (Sale and District branch). The purpose of these workshops was three-fold: i) to inform the regional community about the project; ii) to determine whether they saw the need for, and supported, the type of research being undertaken in the Gippsland Lakes; and iii) to determine whether they would be interested in participating directly in it. The two groups broadly covered the interests of amateur biologists and conservationists, and hunting and fishing interests respectively, but the division does not necessarily mean that their interests were always incompatible.

At the conclusion of the introductory workshops, a short questionnaire of 12 Likert-type questions was distributed. The Likert scale is a psychometric measure commonly used in questionnaires and surveys, where participants are invited to rank their agreement or disagreement to a series of statements or questions (Likert 1932). Typically a five-level response is invited, ranging from strong agreement (score of 5) to strong disagreement (score of 1), via an intermediate of 'neither agree nor disagree' with a score of 3. The return rate for each group was 85% (18 from 21 at the Field Naturalists' meetings; 29 from 34 at Field and Game meetings). The results are shown in Fig. 1, 2 and 3.

Both community groups were highly aware of the wetlands fringing the Gippsland Lakes, with Likert scores exceeding 4 out of the highest possible score of 5 (i.e. a score of 1 = not aware and a score of 5 = highly aware). Almost all respondents thought the wetlands were integral to the larger Gippsland Lakes (Likert score >4.5 out of 5). This is an important finding because until recently there was a tendency among some natural-resource managers, and even among some scientists, to consider the lagoonal components of the Gippsland Lakes separately from the fringing wetlands.

One question addressed the topic of why local residents valued the wetlands of the Gippsland Lakes. As expected a wide range of reasons was expressed, but among the most common reasons were aesthetics, recreation, and environmental values (Fig. 1). The main difference in responses between the two community groups was that 'sport' was rated highly by Field and Game members but very lowly by Field Naturalist members. Field and Game members also ranked the provision of 'ecosystem services' more highly than did Field Naturalists.

Subsequent questions addressed wetland condition (Fig. 2). Field and Game members thought the wetlands were in poorer condition than did the Field Naturalists, although both groups tended to the view that the wetlands were in only 'fair' condition. Field and Game members perceived a decrease in wetland condition over time, whereas Field Naturalists returned a Likert score of 2.5 out of 5, indicating that they perceived no great change in condition (data not shown). A wide range of factors was proposed as causing wetland degradation: the presence of Carp *Cyprinus carpio* L., an introduced fish species, was an almost ubiquitous explanation, followed by an inappropriate water regime, salinity impacts, and the generic term 'plant loss'. Weed invasions were problematic for Field Naturalists but less so for Field and Game members. 'Excess bird numbers' was identified by 21% of Field and Game members as contributing to wetland degradation. It is likely, however, that ibis rather than huntable species such as waterfowl, were the birds identified as being 'too abundant'. What might seem to be an esoteric cause of environmental degradation to many people not familiar with wetlands – acid sulfate soils – was identified by 17% and 24% of Field and Game and of Field Naturalists members, respectively. We deliberately left the meaning of the terms 'condition' and 'degradation' open, so that respondents could reply in terms of what they perceived as a change in wetland condition and the factors responsible for any degradation.

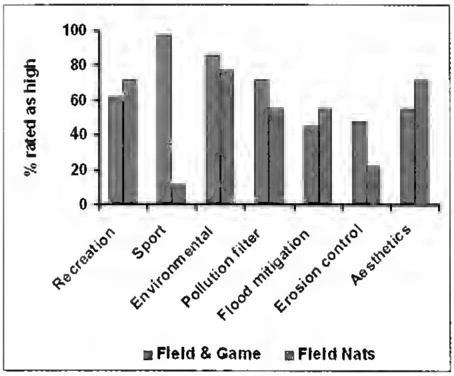

Fig. 1. Comparison of responses from two contrasting community groups (Field and Game Victoria Sale and District branch, and Field Naturalists Club of Victoria) on the value of Dowd Morass. The y axis shows the percentage of respondents who placed a rating of 'high' for the various values indicated along the x axis.

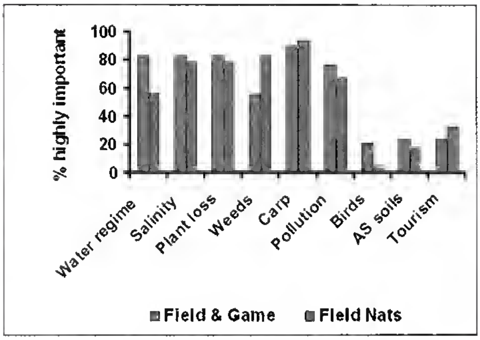

Fig. 2. Comparison of responses from two contrasting community groups (Field and Game Victoria Sale and District branch and Field Naturalists Club of Victoria) on the factors believed to cause degradation of Dowd Morass. The y axis shows the percentage of respondents who placed a rating of 'highly important' for the various threats indicated along the x axis. AS soils = acid sulfate soils.

The penultimate question sought feedback on the project's relevance to the better management of wetlands in the Gippsland Lakes Ramsar site, and the final question sought to determine whether members were willing to be involved in various aspects of the project. The two groups returned very high Likert scores (4.6 and 4.7 out of 5) for the 'relevance' question, which indicated they thought the R&D project was strongly relevant to the better understanding and management of the Gippsland Lakes and its fringing wetlands. As discussed below, this was an important conclusion because it allows us to further justify the project to its financial backers in Commonwealth and State Governments. Over three-quarters of respondents (76% for Field and Game; 83% for Field Naturalists) wanted to be involved in some aspect of the project. When asked what activities were of most interest, revegetation and bird counts, followed by water quality monitoring and frog counts, were common responses (Fig. 3). We took advantage of these positive responses in later aspects of the project, as discussed next.

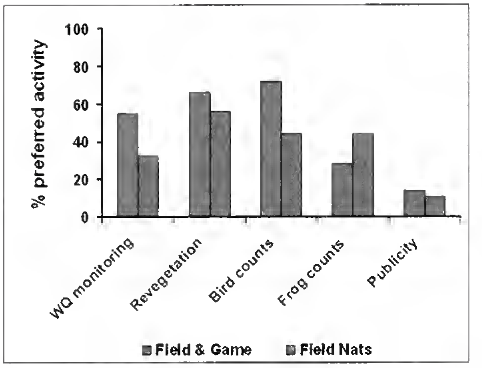

Fig. 3. Comparison of responses from two contrasting community groups (Field and Game Victoria Sale and District branch and Field Naturalists Club of Victoria) on the activities they would prefer to be involved in as part of the R&D project on Dowd Morass. The y axis shows the percentage of respondents who placed a rating of 'would like to be involved' for the various activities indicated along the x axis. WQ monitoring = Water quality monitoring

Examples of on-ground community involvement

Members of Field and Game and the Field Naturalists, as well as other community groups such as local schools, were heavily involved with active revegetation of degraded parts of the wetland with *M. ericifolia* seedlings. The results of these revegetation trials have been reported in Raulings *et al.* (2007). Over 2000 root-stock seedlings were planted during the conduct of the revegetation experiments, and it would have been impossible for the research team to plant this many seedlings in the complex array of experimental areas without the help of the many volunteers from local community groups.

Volunteers from community groups also were instrumental in the two experimental drawdowns of water levels attempted over 2003 to 2005. With the assistance of Parks Victoria staff and volunteers from Field and Game, repairs were made to gaps that had developed over the past three decades in the internal levees subdividing Dowd Morass. Repairs to Heywood's embankment in particular were required so that the east and west sides of the wetland could be physically isolated and their water levels manipulated independently in a planned Before-After-Control-Impact (BACI) experimental design, even if a different technique—gradient analysis—was used for the final statistical analysis: see Raulings *et al.* (2010, 2011) for details.

Community-group volunteers not only repaired the levees, but were active in monitoring water quality in the wetland (through the regional Waterwatch initiative, membership of which was common within the Field Naturalist group) whilst water levels were being manipulated, and in providing surveillance against vandalism. Perhaps most importantly, volunteers from Field and Game maintained the two pumps that maintained contrasting water levels in the two parts of the wetland: large, diesel-fuelled agricultural pumps were used to pump water from various parts of Dowd Morass and they had to be maintained regularly (e.g. fuel supply, periodic changing of filters etc). The costs of maintaining the engines and centrifugal pumps from our base in Melbourne were prohibitive, but diesel engineers who happened to be members of Field and Game donated their services to maintain the pumps and, as far as possible, ensure they were kept under periodic surveillance to guard against vandalism. (Vandalism was initially a problem, as outlined in Raulings *et al.* 2011).

The final benefit of collaboration with community groups was that it further demonstrated to the government funding agencies (which provided most of the financial support) that the project had significant local backing. Such a demonstration was particularly important for the gaining of Commonwealth support for the project. Salt *et al.* (2008) also found that strong public support was needed to secure the legislative funding necessary for successful rehabilitation of the Everglades wetlands in Florida (USA).

Comparisons of community and scientific understanding

One area of community involvement and the utilisation of local knowledge that has become increasingly valued in the rehabilitation of degraded landscapes is the construction of historical narratives, which provide a long-term perspective of environmental change in a given region (Robertson *et al.* 2000; Bart 2006). These historical perspectives are especially powerful if they include repeat photographs of chosen sites to complement verbal descriptions (Pickard 2002). McClanachan (2008) provides a striking example of how historical photographs can be used to track environmental degradation in aquatic systems. Some information of this nature is available for the Gippsland Lakes (e.g. Ellis and Lee 2002), but it is not as detailed as the oral and photographic history that has been documented for the Tuggerah Lakes, a broadly comparable coastal lagoonal system in central New South Wales, by Scott (1998). Given the lack of a well-described community narrative, we sought to draw out what the community perceived as the factors most responsible for wetland degradation and how those views compared with the understandings of the professional aquatic ecologists involved in the R&D project.

During the project we became aware of several cases where the oral tradition (expressed as community beliefs summarised in Figs. 1, 2 and 3) was in close agreement with scientific understanding of wetlands and their ecology. In other cases there were clear mismatches. One example is the perception by many community members that carp were largely responsible for the ecological degradation of Dowd Morass (Fig. 2). This belief does not match with the scientists' understanding of the factors primarily responsible for the degradation of Dowd Morass: firstly an inappropriate water regime, exacerbated by secondary salinisation; followed by nutrient enrichment; and finally, the presence of acid sulfate soils. The identification of carp by community groups as the main culprit in the degradation of wetlands is not surprising, given that carp are highly visible exotic fauna that have been the topic of several regional workshops and that they are episodically abundant throughout fresh and brackish water parts of the Gippsland Lakes. But many community perceptions of carp-mediated impacts are simply wrong: Koehn *et al.* (2000), for example, identified 12 common perceptions about carp that were not supported by scientific facts, including that carp can remain alive in mud for long periods and that they undermine riparian trees and river banks. Many of these putative abilities were expressed during the workshops of 2003.

A second example of a mismatch between community (and even management) beliefs and scientific understanding is weed infestations, which were seen as especially problematic by Field Naturalists but less so by Field and Game members (Fig. 2). Indeed, the draft management plan for the Lake Wellington wetlands (Parks Victoria 1997) identified Brazilian Milfoil *Myriophyllum aquaticum* Vell.Verdc.; syn. *Myriophyllum brasiliense* Cambress. as the critical aquatic weed species in wetlands of the western parts of the Gippsland Lakes. An analysis of the salinity and flooding tolerance of *M. aquaticum* undertaken as part of our rehabilitation assessment concluded that the species was too salt-sensitive to pose a major threat to brackish-water wetlands in the Gippsland Lakes, especially given salinity regimes over the past decade (Toomey and Boon 2007).

Conversely, however, the identification of acid sulfate soils as a major problem by ~20% of respondents agreed very well with scientific understanding of the critical role soil acidification can play in degrading *Melaleuca*-dominated coastal wetlands (and their adjacent receiving waters) along the eastern seaboard of Australia (e.g. Johnston *et al.* 2003; Eyre *et al.* 2006).

Conceptual models as a communication tool

One way to achieve congruence across the different groups involved in the management and rehabilitation of Ramsar-listed wetlands is to create a suite of conceptual models that show how wetlands respond to changing environmental conditions, such as altered water and salinity regimes. Conceptual models can be developed to illustrate a range of ecological structures, processes, responses and interactions, and are highly effective in capturing the current scientific knowledge of an ecosystem and showing likely ecological responses to natural and anthropogenic stresses (e.g. Gentile *et al.* 2001). They are an essential – albeit usually neglected – component of the adaptive management framework (Walters 1986), and are especially valuable because they should prompt continued re-evaluation of what is known about a given ecosystem, as well as making explicit the ecological principles that underlie the rehabilitation of degraded areas (Weiher 2007).

The creation of a conceptual model of Gippsland Lakes' wetlands would serve a valuable role in promoting collaboration across the different parties in large, multi-disciplinary R&D projects: such a model should integrate knowledge from the oral tradition, scientific understanding, management expectations, and community perceptions about wetland uses, rehabilitation and water use. The irony here is that in his pioneering book chapter of 1966, Eric Bird included a simple and elegant predictive conceptual model of how vegetation in the Gippsland Lakes would respond to progressive salinisation (Bird 1966). Perhaps it just shows that important advances in the resolution of the environmental issues we face today were often made many years ago, by an older generation of scientists. It also indicates that it pays not to neglect this older literature, nor the perspectives of the 'elders' (scientific and community) in our society, many of whom have valuable scientific and natural history perspectives on matters such as the health of the Gippsland Lakes and its fringing wetlands, and the best ways degraded sites can be rehabilitated.

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One Hundred Years Ago

PAPER READ

By Dr. T. S. Hall, M.A., D.Sc, entitled "Notes on the Gippsland Lakes."

The author, in the course of an interesting paper, illustrated by lantern slides, traced the extent of the Gippsland Lakes, which were formerly estuaries of rivers, but had become lakes by the action of sand-dunes, which still continue to form along their seaward margins. The position of portion of- the old coast-line was still marked by cliffs of marine Tertiaries in several places. He said that it seemed probable that the altered conditions which now exist could be traced to the breaking-down of the land-bridge which once existed between Australia and Tasmania and the probable change in the ocean currents which followed the formation of Bass Strait. The chairman said the thanks of the Club were due to the author for his very interesting and instructive paper.

Messrs. A. D. Hardy, G. A. Keartland, and A. H. E. Mattingley took part in a short discussion which followed.

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The influence of cover on nesting Red-capped Plovers: a trade-off between thermoregulation and predation risk?

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Abstract
Some ground-nesting birds adopt a mixed strategy of nesting in the open, or under cover (e.g. vegetation). This may represent a trade-off between thermally favourable nest sites (covered) and those that enable the early detection and avoidance of predators (open). This study examined whether such a trade-off exists for Red-capped Plover *Charadrius ruficapillus*, whose eggs are preyed upon principally by Little Raven *Corvus mellori*. For real and artificial nests, nest temperatures under cover (real, 25.9 ± 0.1°C; false, 16.2 ± 0.5°C) were cooler than those in the open (real, 26.8 ± 0.1°C; false, 17.4 ± 0.9°C). Covered nests had more visual obstructions than open nests (covered, 65.5% ± 11.4%; open, 7.4% ± 2.8%) and a standardised measure of incubator escape distance, initiated by experimental human approaches, indicated incubators fled open nests at longer distances than for covered nests. Nests under cover showed a slightly (non-significant) higher probability of surviving one day (Daily Survival Rate [DSR] = 0.978) than those in the open (DSR = 0.950). For false nests containing model eggs, covered nests exhibited better survival to 10 days compared with open nests (20.4% vs. 4.7%). Thus, covered nests are associated with enhanced thermal environments and egg survival, but predators can approach the incubator more closely. Overall, the proposed trade-off between thermal and predation risks associated with nest sites appears to exist and explains the ongoing occurrence of nests in open and covered locations. (*The Victorian Naturalist* 131 (4) 2014, 115–127)

Keywords: shorebird, wader, microhabitat, *Charadrius ruficapillus*

Introduction
Depredation is among the most important factors influencing the rate of clutch loss for many avian species and accounts for up to at least 80% of all clutch failures across a variety of species, habitats and geographic locations (Ricklefs 1969; Martin 1993). High rates of clutch loss may result from the occurrence of superabundant generalist egg predators that opportunistically prey upon nesting birds and their eggs (Marzluff and Neatherlin 2006). Corvids are a common avian nest predator and are the most prominent egg predator of many ground-nesting birds (Angelstam 1986; Dwernychuk and Boag 1972; Yahner and Wright 1985). These generalist omnivores are highly adaptable and are increasing in numbers, taking advantage of the anthropogenic resources available to them in a highly urbanised world (Angelstam 1986; Marzluff 2001; Marzluff and Neatherlin 2006; Wallander *et al.* 2006).

Nesting birds employ a range of anti-predator strategies including crypsis (hiding the nest and its contents) and active defence, including aggression and distraction (Byrkjedal 1989; Geering *et al.* 2007). Nest site selection (the choice regarding where a bird places its eggs) is one of the most important reproductive decisions (Thyen and Exo 2005; Smith *et al.* 2007) because it has a major influence on nest outcome (Angelstam 1986; Martin 1993; Gotmark *et al.* 1995; Santisteban *et al.* 2002; Tieleman *et al.* 2008). Once birds decide on a nest site and eggs are laid, the eggs cannot be moved and must survive the incubation period *in situ* if young are to hatch. Nest location also influences the prevailing thermal conditions for incubation, particularly for ground-nesting species whose nests do not feature structures which aid thermal insulation (Amat and Masero 2004a; Tieleman *et al.* 2008).

Cover and temperature
Incubation, the use of parental body heat to thermoregulate eggs (Blanken and Nol 1998)

is essential for the growth and development of avian embryos (Purdue 1976; Clutton-Brock 1991; Tieleman *et al.* 2008). Adults both warm and cool eggs as required (Grant 1982; Weston and Elgar 2007). For adult shorebirds that typically nest on the ground, thermoregulatory behaviours such as egg shading, heat transfer via the brood patch or belly soaking (Purdue 1976; Downs and Ward 1997; Geering *et al.* 2007) are employed to cool or warm the eggs to maintain optimal egg temperatures. In most avian species, optimal egg temperature for embryo growth and development ranges between 32°C and 35°C and lower and upper lethal temperature limits range from 25°C to 27°C and 43°C to 44°C, respectively (Tieleman *et al.* 2008). Many shorebirds nest on the ground with little or no insulation in nests (Amat and Masero 2004a). In areas with little or no cover, incubating birds and their eggs can be exposed to air temperatures in excess of 40°C and ground temperatures in excess of 50°C (this study). As egg temperature is positively correlated with ambient temperature (Purdue 1976; Weston and Elgar 2005), in areas where air temperatures can reach extremes, nesting adults and their developing embryos can be exposed to conditions that threaten egg viability through overheating (Alrashidi *et al.* 2010). Nesting under vegetation provides protection from the thermal extremes of wind and solar radiation (Amat and Masero 2004a; Kim and Monaghan 2005). Vegetation surrounding a nest acts as an insulator by blocking wind, and shade created by overhanging vegetation helps to create a more stable and thermally favourable microclimate (Amat and Masero 2004b; Kim and Monaghan 2005; Smith *et al.* 2007; D'alba *et al.* 2009).

Cover, crypsis and predator defence

In addition to providing thermal protection, cover may also provide protection from predators (Tieleman *et al.* 2008). Vegetative cover can be defined as any form of vegetation that provides crypsis, concealing or protecting an animal. There is great variation in the costs and benefits of nesting under cover or in the open (Table 1). With the risk of clutch depredation high in many avian species, the ability to detect the approach and proximity of potential predators during nesting is likely to enhance defence of clutches, especially for passively defending species, which optimise nest crypsis by distancing themselves from the nest location in the proximity of predators. Nesting in the open also allows an incubating adult to better detect an approaching predator and maximise its own survival. An exposed open nest, however, also allows predators to better detect incubating adults and their nests from a distance. Cover may obscure the vision of an incubating parent from the nest, thus hindering its ability to detect approaching predators (Gotmark *et al.* 1995; Javurkova *et al.* 2011). This can prove particularly costly when the species requires early detection for escape (Smith *et al.* 2007) because cover can impede a bird's ability to escape the nest undetected (Wiebe and Martin 1998) and increase the risk of adult depredation (Wiebe and Martin 1998; Amat and Masero 2004a; Low *et al.* 2010).

Many birds that nest in open habitats (e.g. waders) have several adaptations to avoid detection by predators. Among these, some species move to environments where they can avoid terrestrial predators: Banded Stilts *Cladorhynchus leucocephalus*, for example, breed on isolated islands in ephemeral inland lakes (Geering *et al.* 2007). Incubating adult waders often exhibit cryptically coloured dorsal plumage presumably as camouflage, decreasing the likelihood of nest detection and clutch depredation (Wallander *et al.* 2006). Species may also lay cryptically coloured eggs (especially effective when the nest is left unattended) and some hide their nest under vegetation as a form of defence (Wallander *et al.* 2006). Parents employ distraction, aggression or crypsis in defence of their clutches (Geering *et al.* 2007; Ekanayake and Weston 2011).

Study aims

Species that can nest in either the open or under cover, may effectively trade-off between thermally favourable nest sites, and their view from the nest (i.e. overall safety from predators), when selecting nest sites (Gotmark *et al.* 1995; Wiebe and Martin 1998; Thyen and Exo 2005; Tieleman *et al.* 2008). This study investigated the benefits and costs associated with nesting under cover or in the open in the ground-nest-

Table 1. A partial review of studies investigating the influence of nest cover on the nesting success of bird species. NA means not applicable.

Authors	Species	Nest type	Climate	Most successful real nest cover type	Most successful false nest cover type
Amat and Masero (2004a)	Kentish Plover *Charadrius alexandrinus*	Ground	Hot	No effect of nest cover	NA
Tieleman et al. (2008)	Hoopoe Lark *Alaemon alaudipes*	Ground	Hot	No effect of nest cover	No effect of nest cover
Gotmark et al. (1995)	Song Thrush *Turdus philomelos*	Tree (low cup nests)	Temperate	No effect of nest cover	Covered nests
Dwernychuk and Boag (1972)	Ducks (six species)	Ground	Unknown	No effect of nest cover	Covered nests
Wiebe and Martin (1998)	White-tailed Ptarmigan *Lagopus leucurus*	Ground	Alpine	Covered nests	NA
Ludwig et al. (2010)	Black Grouse *Tetrao tetrix*	Ground	Cold	Covered nests	NA
Colwell (1992)	Wilson's Phalarope *Phalaropus tricolor*	Ground	Unknown	No effect of nest cover	NA
D'Alba et al. (2009)	Common Eider *Somateria mollissima*	Ground	Cold	Covered nests	NA
Kim and Monaghan (2005)	Herring Gull *Larus argentatus* and Lesser Black-backed Gulls *L. fuscus*	Ground	Temperate	Covered nests	NA
Schieck and Hannon (1993)	Willow Ptarmigan Lagopus lagopus	Ground	Cold	No effect of nest cover	NA

ing Red-capped Plover *Charadrius ruficapillus*, using real eggs and nests and model eggs in artificial (henceforth 'false') nests. Specifically, this study aimed to determine whether:

1. covered nests present incubating adults with a more favourable microclimate for nesting by maintaining more favourable (i.e. less extreme) nest temperatures; and
2. cover restricts the view from the nest of an incubating adult, affecting its ability to detect and react to potential approaching predators. These metrics represent indices of predation risk to adults, and also to eggs because the defence of eggs relies on the incubator's departure from the cryptic eggs before a predator approaches too closely.

Methods

This study was conducted at two neighbouring coastal wetland sites, in an urbanising landscape, in southern central Victoria, Australia: Cheetham Wetlands (37°53'56"S, 144°47'33"E; 420 ha) and Truganina Swamp (37°52'07"S, 144°48'12"E; 148 ha) (Antos *et al.* 2007). The area experiences a Mediterranean climate, with temperatures during the study period reaching a minimum of -1.7°C and a maximum of 47.5°C (Bureau of Meteorology 2012). Due to the close proximity of the two sites and the fact that Red-capped Plovers and their primary predator (Little Raven *Corvus mellori*) move between these sites (unpubl. data), data is pooled across sites. The study areas are substantial and apparently suitable nest sites abound (both covered and in the open).

Nest monitoring

Nests were located between July 2011 and February 2012 by searching regularly in and around ponds containing suitable nesting habitat (for seasonal variation in the occurrence of nests under cover and in the open, we draw upon data that was collected starting in 2010, and for response distances we draw upon nests recorded from mid-2011 to mid-2014). Red-capped Plovers almost always nest in new locations (within or between pairs), and there is no 'traditional' use of specific nests. Once a nest was located, the expected hatch date was estimated to enable success rates of nests to be estimated. Eggs were aged using the flotation method (Liebezeit *et al.* 2007) and estimated hatch date was calculated assuming a 30-day incubation period. Nests were then visited around this estimated date to determine hatching success. Incubating adults were caught by using walk-in nest traps (see for example, Cardilini *et al.* 2013). Captured individuals were banded with a metal band placed on the tarsus and an orange flag engraved with a unique two-letter combination on the tibia. This combination was used later to identify individual adults and often aided in establishing whether some nests were successful or had failed by observing a brood accompanying the flagged adult/s (see Lees *et al.* 2013).

Following hatching or failure, nest site characteristics were recorded. Nest cover (any vegetation dead or alive directly above the nest) was indexed for each nest by placing a circular 10.5 cm diameter quadrat in the nest scrape, and counting the number of 12 x 12 mm grids visible (88 grids in total) from directly above. The percentage of visible grids was calculated and a cover type (open or covered) allocated. A covered nest was defined as having ≥ 10 % of the grids covered.

Thermal environment

Temperature loggers (Thermodata™ thermochron iButtons) were used to index nest temperature. They were placed in real nests just under the surface of the scrape and were programmed to record temperature once every hour for 10 days. Loggers were deployed only in nests ≥ 10 days old, ensuring laying had ceased. Ambient temperatures were recorded using iButtons suspended in shade 50 cm above the substrate. To test for an effect of incubation and cover on nest temperature, temperature loggers were also deployed in false nests (see below). For analysis of day and night temperatures, night was defined as the hours of 2100–0600 and day 0700–2000; this reflected periods of light and dark during the study period.

Indexing incubator detection and response to predators

Flight Initiation Distance (FID), the distance between predator and prey when escape begins (Weston *et al.* 2012) was used in this study as a reliable measure of predator detection by incubators (see Guay *et al.* 2013a). To obtain FID, the investigator approached incubators from a distance that maximized the Starting Distance (SD) (e.g. Guay *et al.* 2013a, b, c). For open nests minimum SD was the distance at which the incubating adult was visible on the nest and could clearly see the investigator; for covered nests SD was defined as the distance at which the vegetation covering the nest could be seen by the investigator from the direction of approach. For FID, we supplemented our data for the 2011/12 season (collected by SL) with estimates collected during the 2013/14 breeding season (collected by LXT). A General Linear Model (GLM) revealed no difference between logSD between observers/seasons ($F_{1,42} = 0.534$, P=0.135) but a significant difference in logSD between covered and open nests ($F_{1,42}$=9.314, P=0.004). Longer SDs for open nests may obscure the effect of cover on FID (see McGriffin *et al.* 2013). Thus, we elected to adjust FID for SD, where FID_{adj} is the FID at the average SD (both logged), derived from a linear regression of logFID against logSD ($F_{1,45}$=32.900, $P <$ 0.001, R^2=0.428). The adjustment: (1) used the slope of the relationship to obtain the model estimate of FID at average SD, and (2) then involved the addition or subtraction of the appropriate residual values. Nests were then approached directly at a constant walking pace until the bird fled the nest (i.e., FID). Distances were recorded using a laser rangefinder (after Glover *et al.* 2011).

Each nest for which an FID was recorded was assessed for visibility of the incubator from the nest (henceforth 'nest visibility') after the nest had hatched or failed. Visibility was measured by taking photographs (Lumix FT2, 14.1 Mega Pixel [MP] camera) from the nest scrape facing an 89 × 60 cm panel placed at each compass point (four in total to survey a bird's field of vision) and above the nest, each at a distance of 1 m from the nest. Each panel had 54 square 4 × 4 cm grids marked on it and photographs were examined to determine the number of grids obscured by cover.

Egg survivorship and predation risk

False nests with model eggs were deployed across the study site as a method of indexing depredation rates. Although depredation rates of model eggs may not mimic that of real nests, they may be used to determine relative differences in depredation between habitats (Angelstam 1986; Willson *et al.* 2001). To determine any seasonal variation in depredation rates, the experiment was repeated five times (September to February excluding October), but all repeats involved new nest locations. The experiment was a simple single-factor design to determine depredation rates on open versus covered nests. The first experiment (September) incorporated a second factor, namely the use of remote sensor cameras (Scoutguard™: 5MP ultra Compact digital scouting/trail camera, DTC-530 V, HCO Outdoor Products) to monitor half of the false nests (stratified across the cover and open treatments) to identify egg predators. Due to the overwhelming depredation rate on camera-monitored nests, subsequent deployments did not involve the use of cameras.

The number of nests and eggs deployed, and in some cases the occurrence of nest checks, was influenced by unanticipated flood events. On average 74 false nests were deployed at least 100 m apart and at random locations throughout the study site (no site was reused) and were randomly allocated a cover treatment (open or covered). Each nest, a small depression in the substrate, contained two Japanese Quail *Coturnix japonica* eggs that were chosen because they best resemble those of Red-capped Plovers. Nests were monitored for a 30-day period to mimic plover incubation duration and checked at regular 10-day intervals after deployment to establish nest depredation rates and to retrieve any deployed temperature loggers.

Real nests were visited weekly to determine nest success or fate (here, 'success' describes the likelihood of hatching and 'fate' refers to the cause of failure; after Cardilini *et al.* 2013) and was classified as either depredated, flooded, abandoned or successful. Due to the imbalance between clutches depredated and those that were successful (87.5% failed), the application of many statistical approaches such as logistic regression was not possible. Alternatively, to determine any differences in clutch survival between open and covered nests, daily survival rate of nests (DSR, the probability that a nest would survive one day) was calculated using Mayfield's method (Mayfield 1961, 1975).

Statistical analysis

Standard statistical procedures were followed for all analyses as outlined in Quinn and Keough (2002). Mixed modeling, correlation analysis and general linear models (GLMs) were performed in Statistical Package for the Social Sciences (SPSS), version 18.0 (SPSS Inc. Chicago, Illinois), Daily Survival Rate (DSR) calculations and contingency table (χ^2) analysis were performed manually in Microsoft Excel, version 2010. To calculate exposure days required for DSR analysis, GLMs were used within the package MASS in the statistics software package R, version 2.11.1. (R Development Core Team). Summary statistics are presented as means ± one standard error (unless otherwise stated). Graphs, temperature data and percentage data are all presented with raw untransformed data to enhance readability and do not imply normality of data.

Results

Eighty-nine nests were discovered of which 29 (33%) were under cover and 60 (67%) were in the open (Fig. 1). Of those, 72 clutches (81%) were depredated, nine (10%) were successful, five (6%) were abandoned and three (3%) were flooded. Sixteen chicks hatched from the nine successful nests; however, none survived to fledge. An examination of data from 2010 to 2012 (n=191 nests), revealed that covered and open nests were located throughout the breeding season (Fig. 2). For those months in which at least 10 nests were recorded, the percentages of covered nests were: 72.0%, September; 77.8%, October; 58.6%, November; 54.3%, December; 58.2%, January; and 75.0%, March.

Thermal monitoring

Residuals of the relationship between false nest temperature and ambient temperature were derived from a linear regression which was selected using the curve fit procedure on SPSS; $\log_{10}$(Ambient °C)=0.730 * $\log_{10}$(False Nest °C) + 0.348 (R^2=0.676, $F_{1,12051}$=25118.420, $P < 0.001$). Thus, high residual values (absolute) indicate greater deviation from this model, positive values indicate hotter temperatures than modelled, and negative residual values indicate cooler temperatures than modelled. Residuals were used as the response variable for subsequent mixed modelling of the influence of cover on false nest temperatures. The mixed model included a random factor of nest identity to control for repeated sampling of the same nest. This mixed model indicated that open false nests were warmer than those under cover (open nests compared with covered, coefficient=0.0182, t=10.255, $p \leq 0.001$; Fig. 3). In terms of actual temperature this represents a mean difference of 1.2°C (open, 17.4 ± 0.9°C, n=17 nests; covered, 16.2 ± 0.5°C, n=26 nests).

Some real nests being thermally monitored were preyed upon or hatched before collection of the loggers, necessitating truncation of the sequence of the thermal data, such that it represented only those periods during which eggs were being incubated. To truncate sequences, thermal data from nests were inspected for minimum temperatures recorded across a 24-hour period. Of the 17 nests surveyed, incubation had ceased prior to logger collection (temperature ≤ 18°C) in five nests and the thermal data preceding this date were excluded. Of these nests, one data set was excluded entirely due to incorrect temperature readings. Residuals of the relationship between real nest temperature and ambient temperature were derived from a linear regression which was selected using the curve fit procedure on SPSS; $\log_{10}$(Real T °C)=0.474 * $\log_{10}$(Ambient T °C) + 0.775 (R^2=0.546, $F_{1,3936}$=4733.363, $P \leq 0.001$). Residuals were used as the response variable for subsequent mixed modeling of the influence of cover on real nest temperatures. The mixed model included a random factor of nest identity to control for repeated sampling of the same nest. This mixed model indicated that open real nests were warmer than those under cover (open nests compared with covered, coefficient=0.006, t=3.504, $p \leq 0.001$; Fig. 3). In terms of actual temperature this represents a mean difference of 0.9°C (open, 26.8 ± 0.1°C, n=9 nests; covered, 25.9 ± 0.1°C, n=7 nests). The presence of an incubating parent warmed the nest (Fig. 3), indicating that the nest temperatures indexed the temperatures experienced by the eggs. With nest temperatures around 9.4°C warmer in the open and 9.7°C warmer under cover than temperatures recorded in false nests, real nests were maintained at a higher temperature both during the day and night (day, 27.4 ± 0.1 °C; night, 23.5 ± 0.1 °C, n=16 nests) than false nests (day, 18.4 ± 0.1 °C; night, 13.6 ± 0.1 °C, n=43 nests).

Depredation risk

A potential cost of nesting under cover is that the ability to detect incoming predators may be reduced by visual obstructions around the nest. The number of panel grids visually obstructed from the nest per panel was highly correlated between panels ($r_{Pearson}$ 0.504 – 0.946; all *Ps* ≤ 0.012; n=24) and therefore were summed to characterise overall visibility from the nest (henceforth the total number of panel grids visually obscured ($Obs._{Total}$). $Obs._{Total}$ was also highly correlated with the estimate of cover over a nest from the circular quadrat ($r_{Pearson}$=0.888, $P \leq 0.001$, n=24) so $Obs._{Total}$ was selected for analysis. $Obs._{Total}$ was substantially and significantly higher for covered nests (79.2 ± 3.0 % grids covered, n=11 nests) than open nests (5.2±1.2 %, n=13 nests) (t=17.95, df=22, $p \leq 0.001$).

Forty-six nests were approached to examine whether covered nests were associated with higher FIDs than open nests. The low clutch survivorship of real nests limited the number of FIDs obtained and therefore the number of nests used in FID analysis. Covered nests had more visual obstructions compared with open nests (panels facing the direction of investigator approach [$Obs._{Approach}$], 2011/12 data only; covered, 65.5 ± 11.4%, n=11; open, 7.4 ± 2.8 %, n=13; t=5.367, df=15.418, $p \leq 0.001$). The influence of cover on FID was investigated by running a GLM of FID_{adj} on the two-level factor of 'covered' or 'open'. FID_{adj} for incubators of covered nests was significantly shorter compared with those incubating in the open ($F_{1,44}$=7.174, *P*=0.010; Table 2). Importantly, SD was higher for open compared with covered nests (see Methods), indicating that open nests were associated with greater distances at which potential predators could be detected.

Egg survivorship and predator pressure

Of 38 nests on which cameras were deployed, cause of clutch loss could be determined in 86.8% (33) of cases. In every case where a predator was detected, Little Ravens were the predator. For the subset of false nests with cameras

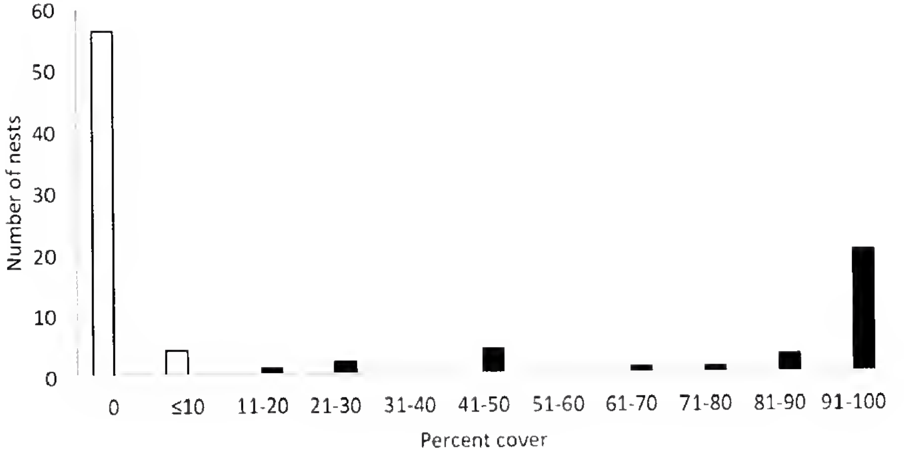

Fig. 1. The frequency distribution of nests found under varying degrees of cover as measured by the percentage of obscured grids of a circular quadrat placed in the nest scrape, when viewed from above. Open nests are shown as hollow bars and nests under cover are shown as solid bars.

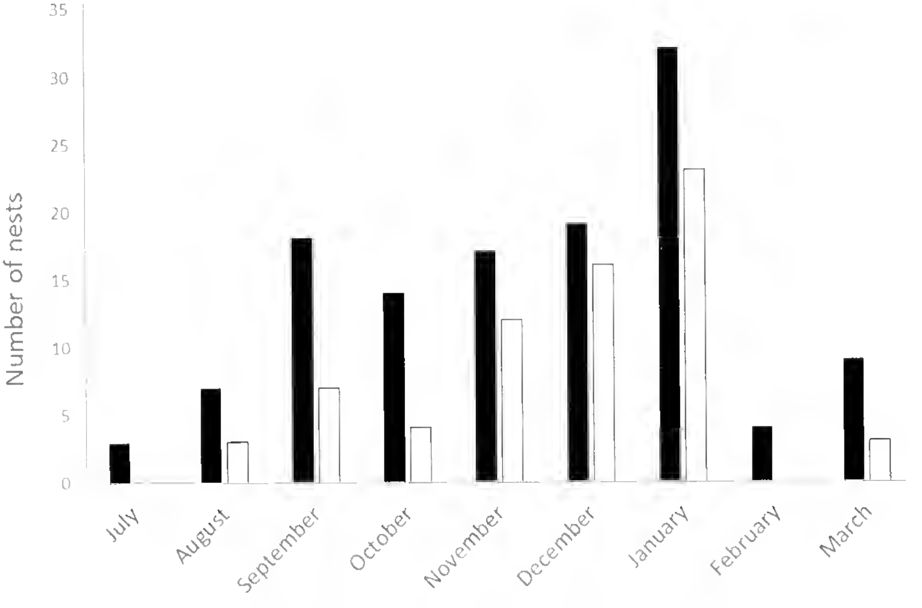

Fig. 2. The number of covered (solid bars) and open (hollow bars) nests in different months of the year, pooled across seasons, 2010 to 2012 (n = 191 nests).

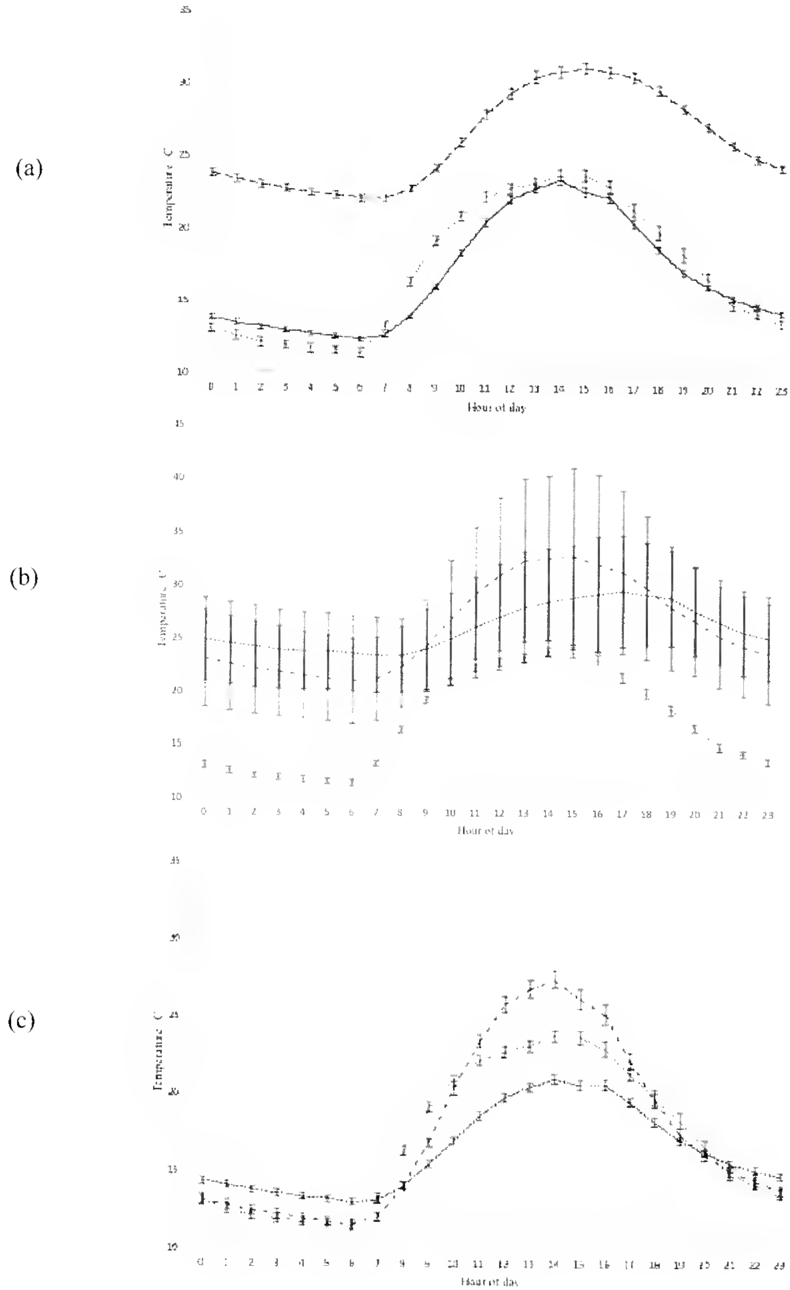

Fig. 3. Mean and standard error of temperature loggers in real and false nests, and at ambient temperature stations, recorded across a 24-hour period (midnight = 0). In each panel, ambient temperature is shown as a grey line. (a) Real nests (dashed line, n = 16) and false nests (solid line, n = 43); (b) real open nests (dashed-dot line, n = 9) and real covered nests (dashed line, n = 7); and, (c) false open nests (dashed-dot line, n = 17) and false covered nests (dashed line, n = 26).

Table 2. Mean and standard error of flight initiation distance (FID), starting distance (SD) and the log of FID adjusted to average SD ($logFID_{adj}$), for covered and open nests (in metres).

Variable	Covered	Open
FID	37.636 ± 9.163	67.154 ± 11.718
SD	62.909 ± 13.810	95.077 ± 13.932
logFIDadj	1.404 ± 0.192	1.869 ± 0.078

versus no cameras, overall take rates to the 10-day check were high (92.2%) which precluded statistical comparison of nests with and without cameras. Nests with cameras (n=40) did not survive beyond 10 days while six nests without cameras (n=37) survived. To remove any influence of cameras on already high take rates, the use of cameras was suspended for subsequent deployments (the time to depredation after camera deployment, in hours [logged], did not differ between covered and open nests; t=0.070, df=30, *P*=0.945).

Of 317 false nests (without cameras deployed), 12.9 % (41) of model clutches survived the first 10 days, 6.6 % (21) to 20 days and 4.7 % (15) to 30 days. The low survival of model clutches meant survival to 10 days was selected for further analysis. Moreover, the imbalance between model clutches preyed upon and those that survived precluded the meaningful application of a number of analytical approaches such as logistic regression. Survival of false nests with model clutches was low, reaching a trough in December before survival improved in January and February 2012 (Fig. 4). For false nests without cameras, covered nests (20.4%) exhibited better survival to 10 days compared with open nests (4.7%) (contingency table, χ^2=17.281, df=1, $P \leq 0.001$).

Of the 89 nests found across the study period, 82 were appropriate to use in analysis of daily survival rate (DSR). Mayfield's estimate of DSR revealed that nest cover did not influence the survival of real nests (t=0.030, df=80, $p \leq 0.976$). Although non-significant, nests under cover showed a slightly higher probability of surviving one day (DSR, 0.978 ± 0.009; 95% CI, 0.995 - 0.961) than those in the open (DSR, 0.950 ± 0.007; 95 % CI, 0.963 - 0.936).

Discussion

Findings from this study supported the predictions that (1) covered nests experienced a cooler and more stable thermal environment for incubation than nests in the open, and that (2) enhanced predator detection and response (as indexed by human approacbes) was associated with open nests.

Thermal environment

In an environment where nesting sites are not limiting and ground temperatures can be in excess of 50°C (this study), this study has demonstrated that incubating Red-capped Plovers gain a thermal advantage during incubation by nesting under vegetative cover, with cover protecting nests and their contents from the thermal extremes, presumably by buffering from wind and sun. This results in cooler nest temperatures being maintained in covered nests, while those in the open experience bigher nest temperatures (the magnitude of the difference in temperatures should not regarded as absolute; we indexed these temperatures). Studies of other ground-nesting birds have demonstrated a similar thermal advantage when nesting across a range of climatic zones and species (Amat and Masero 2004b; Kim and Monaghan 2005; D'alba *et al.* 2009). Methods of sampling nest temperature are diverse; for many studies temperature probes are deployed in eggs (Amat and Masero 2004b), away from the nest scrape (Kim and Monaghan 2005) or in the nest scrape itself (D'alba *et al.* 2009) to obtain thermal data. The diversity of sampling methods and the fact that similar results were obtained across all studies further highlights the influence of nest cover on nest temperature and validates our method of indexing temperature. The difference in nest temperature between open and covered nests suggest that Red-capped Plovers in the open may need to engage more egg and adult thermoregulatory bebaviours when nesting compared to those that nest under cover. Amat and Masero (2004b) showed that female Kentish Plovers *C. alexandrinus* that nest in the

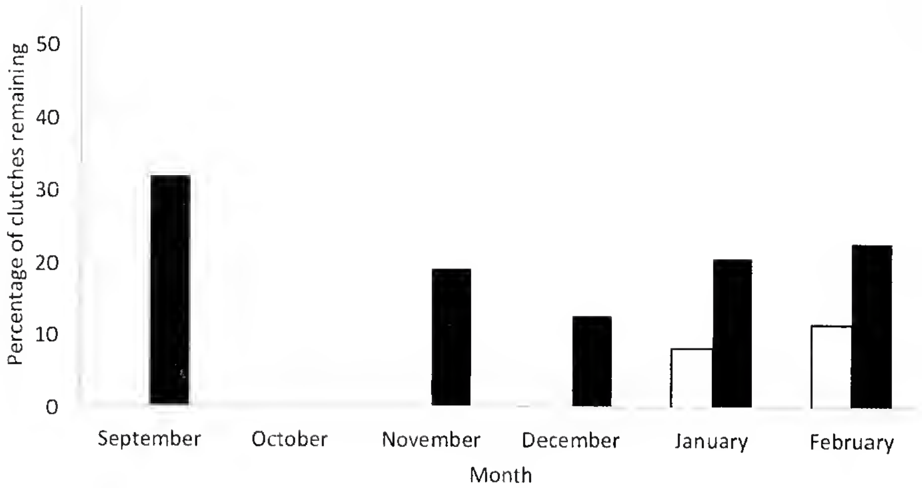

Fig. 4. Percentage of false clutches remaining after the first 10-day check (n = 317 nests) for each calendar month between September 2011 and February 2012. Open nests are represented by open bars and covered nests by solid bars. False nests were not deployed during October.

open and experience warmer temperatures, display behaviours indicative of heat stress such as panting, gaping and belly-soaking, while those that nest under cover in a cooler environment do not. The possibility of these behaviours being adopted by Red-capped Plovers is unknown; however, this would be worthy of further study. Additionally, the need for greater thermoregulation may compromise the crypsis of open nests (e.g. through more frequent change-overs of incubators), and this warrants further study.

This study revealed that covered and open nests were used throughout the breeding season. Under the hypothesis that nest selection is flexible within birds (most of the study birds renest within a season; unpubl. data), it might be tempting to predict a seasonal shift to cover during hotter months (or a similar shift in regard to any seasonal fluctuations in predator occurrence or activity). However, we caution against interpreting these data in this way, without correcting for relative detectability and survival, variation in the availability of covered and open areas, differences between seasons, and interactions between seasonal variation in climate and predators. Such a study would be a useful further endeavour but is beyond the scope of the present study.

Predators

Vegetative cover surrounding a nest obscures the view of an incubating adult of its surrounding environment, and could potentially lead to a reduced ability to detect the approach or proximity of potential predators (Gotmark *et al.* 1995; Javurkova *et al.* 2011). We showed that nest cover obscures the vision of the surrounding environment for incubators of covered nests to a greater degree than for incubators of nests in the open. We also showed that the distance at which detection of an approaching 'predator' (a human) occurs is considerably longer for open nests.

FID was shorter for Kentish Plover nests with more visual obstructions (covered) (Amat and Masero 2004a). This trend also occurs for many species of reptiles and amphibians that display longer FIDs in the open than under cover (Cooper 2006; Martin and Lopez 2000; Martin *et al.* 2006). Similarly, we report that cover is associated with shorter response distances for incubating Red-capped Plovers. Theoretically, the effect of cover on response distance could be reduced if incubators respond to alarm signals of their 'off-duty' partner (Beletsky 1989; Colombelli-Negrel *et al.* 2011; Leavesley and Magrath 2005). However, in our study this did not apparently occur.

Starting distance (SD) is commonly found to influence the response distance (FID) of birds to approaching predators (e.g. Blumstein 2003; McLeod *et al.* 2013; Symonds *et al.* 2014). Thus, the longer an incubating bird has to detect and assess an approaching risk or predator (i.e. a longer SD), the earlier it flees its nest. This study found higher SDs for open nests, again suggesting that potential predators are detected at longer distances at open nests.

Many generalist predators such as corvids are well adapted to urban areas because they use many anthropogenic resources available to them. Point Cook is a highly urbanised area and is predicted to grow in size in the future (Antos *et al.* 2007) possibly creating an ideal environment to support large corvid populations. Predator indexing in this study revealed that Little Ravens were the main predator acting on Red-capped Plover nest success, preying upon 100% of monitored false clutches. This result was not surprising because the surrounding area provides an ideal habitat to support large raven numbers and past studies on the nesting success of plovers in this area attributed clutch loss primarily to this predator (Cardilini *et al.* 2012); Whisson *et al.* unpubl. data).

Despite a slightly higher survival rate for covered nests, this study revealed no real advantage to clutch survival through nesting under cover or in the open, as daily survival rate (DSR) did not differ significantly between the two nesting habitats (though the false nest experiment revealed greater clutch survival under cover). Studies investigating an effect of nest cover on clutch survival vary in their conclusions with some showing no effect (Table 1). This is particularly true of real nests, suggesting the behaviour of an adult at the nest may have some effect on clutch depredation rate (Andersson and Wiklund 1978; Davison and Bollinger 2000). The real nests we studied apparently did not vary in survivorship between cover and open (see also Gotmark *et al.* 1995), though a greater sample size may have revealed a significant difference. Predators search for prey using different techniques and they often rely on visual cues (e.g., avian predators) or an acute sense of smell (e.g. many mammalian predators). Vegetative cover can influence the effectiveness of these search techniques differently and can vary depending on the habitat and predator involved (Ludwig *et al.* 2010; Santisteban *et al.* 2002). Many studies, including the current study, report that false nests under cover exhibit higher survival than those in the open (Brand and George 2000; Dwernychuk and Boag 1972; Gotmark *et al.* 1995; Santisteban *et al.* 2002).

Is nest site selection a trade-off?

Studies investigating the thermal and predator environments of birds during nesting suggest that species with a choice of nesting in the open and under cover employ a trade-off between thermally favourable nest sites, view from the nest and safety from predators when choosing nest sites (Gotmark *et al.* 1995; Tieleman *et al.* 2008; Wiebe and Martin 1998). This study reports a clear thermal advantage to cover, but found no egg survival advantage to nesting under cover (for real nests). Predators can apparently get closer to incubators in covered nests. The fact that Red-capped Plovers nest in both habitats suggest there is a balance of benefits and costs to both options, and these may lie in hitherto unstudied aspects such as incubator stress or survival (LXT, unpubl. data).

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Bird's eggs in a coastal nest. Photo by Michael A Weston

Overview of adaptive management for multiple biodiversity values at the Western Treatment Plant, Werribee, leading to a pilot nutrient addition study

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Abstract
Across south-eastern Australia the loss of natural wetlands since European settlement has been substantial such that even constructed waterbodies that provide a measure of habitat for waterbirds can assume importance for their conservation. Melbourne Water operates the Western Treatment Plant (WTP), occupying 10 500 ha near Werribee, primarily for the treatment of some 54% of Melbourne's industrial and domestic wastewater. During 1982 the site was included as a component of the Port Phillip Bay (Western Shoreline) and Bellarine Peninsula Ramsar site in recognition of its great importance for waterfowl (Anseriformes and Podicipediformes), shorebirds (Charadriiformes) and wading birds (Ciconiiformes and Gruiformes). The WTP supports other biodiversity values, with a significant population of the nationally Vulnerable Growling Grass Frog *Litoria raniformis*; a number of threatened species of plant, including the Spiny Rice-flower *Pimelea spinescens* subsp. *spinescens*; and two listed vegetation communities: *Natural Temperate Grassland of the Victorian Volcanic Plain* (Critically Endangered), and *Subtropical and Temperate Coastal Saltmarsh* (Vulnerable). This paper describes how site managers have endeavoured since 2002 to implement adaptive management to protect and promote the WTP's recognised biodiversity values during necessary sewage treatment upgrades. Results of management on waterfowl populations are obscured by the effects of the 1997–2009 drought across south eastern Australia and species' inherent variability in distribution across this vast area. A trial addition of straw to promote waterfowl food in one wetland showed no clear benefits. However, after 12 years of close monitoring of target populations our knowledge is much improved and we believe the site retains the biodiversity values that led to it being listed as a wetland of international importance. (*The Victorian Naturalist* **131** (4) 2014, 128–146)

Keywords: waterfowl, wetlands, zooplankton, phytoplankton, zoobenthos, management

Introduction

Across south-eastern Australia the loss of natural wetlands since European settlement has been substantial, primarily through drainage or other alterations to natural hydrology. In the state of Victoria it has been estimated that between 33% and 50% of natural wetlands have been lost or degraded since European settlement (Olston and Weston 2004). This loss of natural wetlands is of considerable significance to many species of Australian waterbirds, particularly waterfowl (Anseriformes and Podicipediformes). Permanent coastal wetlands in south-eastern Australia are important non-breeding refuges for many waterfowl species during summer, when inland wetlands dry out (Frith 1982). There has been a general decline in the abundance of Australian waterbird species, particularly over the past 50 years or so, and the loss of wetland habitat essential for breeding and foraging is considered to be the main factor contributing to this (Maher 1993; Briggs *et al.* 1994, 1997; Kingsford and Thomas 1995; Kingsford 1997, 2000; Kingsford and Johnson 1998; Leslie 1995, 2001). Thus now even constructed waterbodies that provide a measure of habitat for waterbirds can assume importance for their conservation.

The Western Treatment Plant

Melbourne Water operates the Western Treatment Plant (WTP), occupying 10 500 ha near Werribee, primarily for the treatment of some 54% of Melbourne's industrial and domestic wastewater (Fig. 1). The treatment methods used at the plant have included sewage treatment lagoons, grass filtration and land filtration. Established over 100 years ago (Penrose 2001), the WTP has discharged secondary-treated effluent into Port Phillip Bay through marine outfalls for much of that time. This effluent discharge has significantly altered the

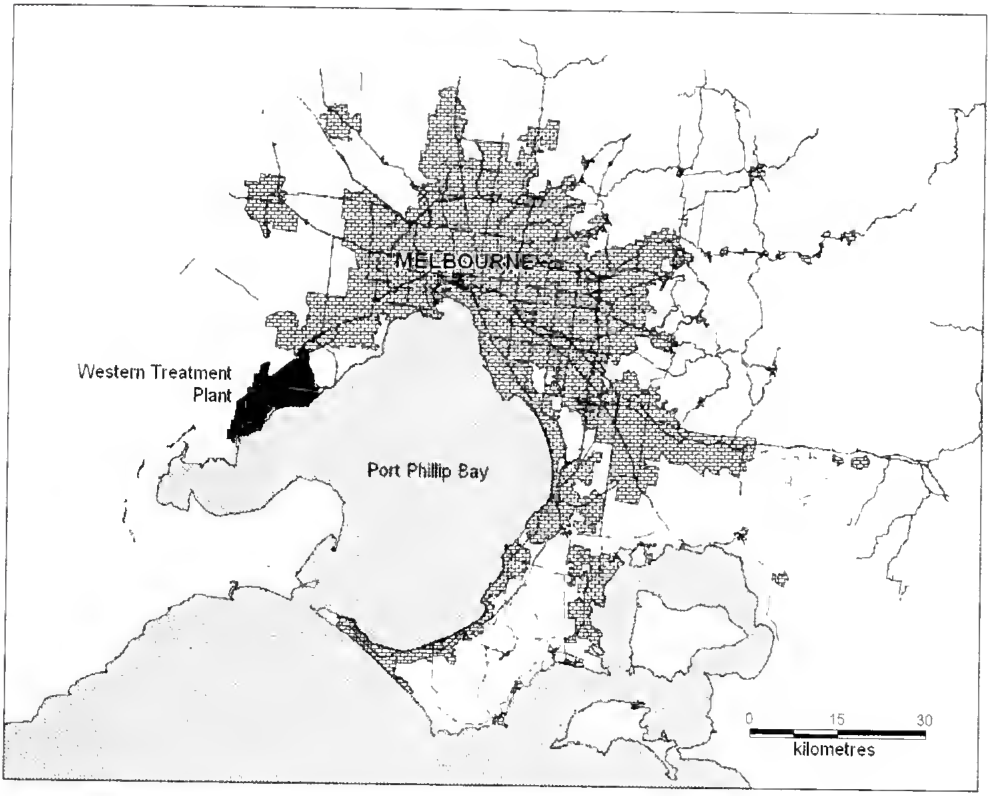

Fig. 1. The location of the Western Treatment Plant, Victoria.

ecology of intertidal and nearshore areas of the bay adjacent to the WTP: increasing macroalgae, microphytobenthos, phytoplankton and zoobenthos biomass (e.g. Axelrad *et al.* 1979; Brown *et al.* 1980; Dorsey 1982; Wood *et al.* 1991; Magro *et al.* 1995).

Almost certainly as a result of this nutrient enrichment, the WTP attracts large numbers of shorebirds (Charadriiformes), including migratory species that breed in northern Asia and which are subject to international agreements to protect migratory birds and their habitats. The area around the WTP supports internationally important populations (i.e. > 1% of flyway population) of seven shorebirds for part of the year: Double-banded Plover *Charadrius bicinctus*, Curlew Sandpiper *Calidris ferruginea*, Red-kneed Dotterel *Erythrogonys cinctus*, Red-necked Stint *Calidris ruficollis*, Sharp-tailed Sandpiper *Calidris acuminata*, Pied Oystercatcher *Haemotopus longirostris* and Banded Stilt *Cladorhynchus leucocephalus* (Watkins 1993; Dann 2007). The construction of numerous sewage treatment ponds has created a variety of permanent waterbodies that provide habitat for waterfowl and an important drought and hunting refuge for these birds. The plant has an extensive network of waterbodies, covering some 1600 ha and comprising 190 individual ponds (Fig. 2). There is a wide variety of waterbody type, with both constructed ponds (n = 181) and natural wetlands (n = 9). Constructed ponds include 122 operational sewage treatment ponds in eight lagoon systems of inter-connected ponds, and 59 decommissioned sewage treatment ponds now managed as wildlife habitat (Steele 2009). Natural wetlands include four waterbodies that retain something of their natural hydrological regime, with a further five now receiving direct loading of water in a managed regime.

During 1921 parts of the WTP were declared

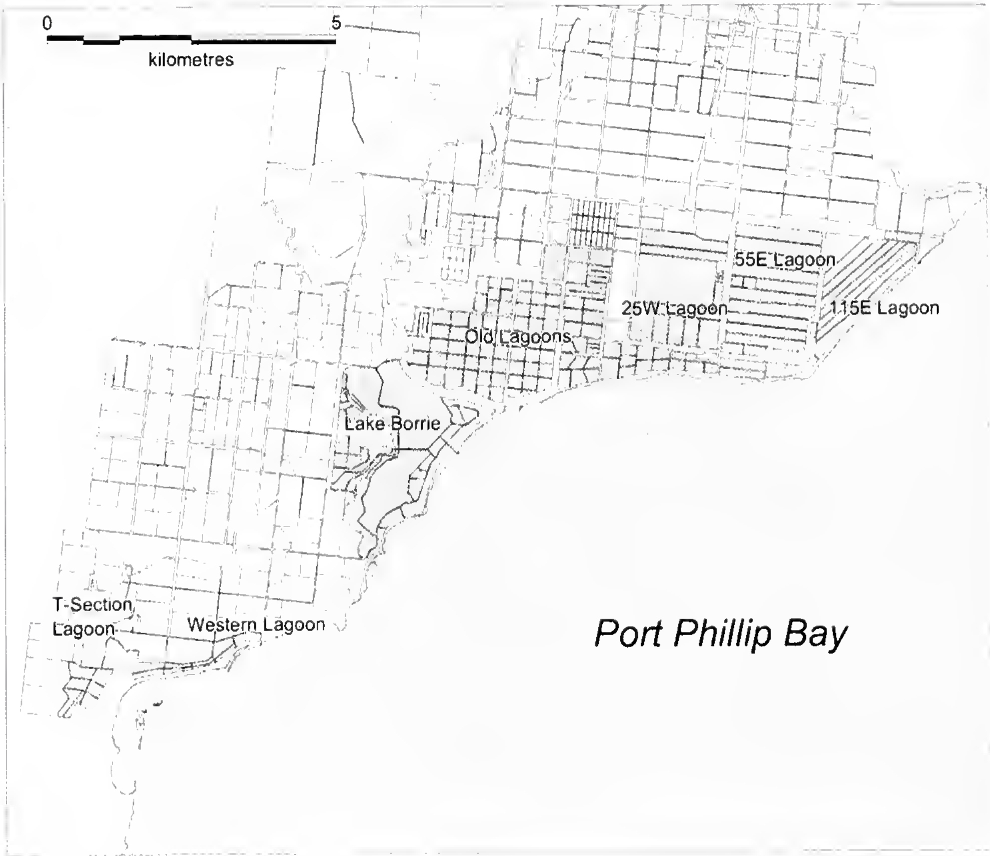

Fig. 2. The Western Treatment Plant, Werribee, with constructed wetlands shown as shaded areas.

a wildlife sanctuary in recognition of the site's avifauna values. Later, in 1982, the entire WTP was included as a component of the Port Phillip Bay (Western Shoreline) and Bellarine Peninsula Ramsar site, and formally recognised as a wetland of international importance. The waterbird populations supported at the WTP were the principal value that contributed to the site meeting the criteria for listing as a Ramsar wetland. However, the WTP also supports a significant population of the nationally Vulnerable Growling Grass Frog *Litoria raniformis*; a number of threatened species of plant, including the Spiny Rice-flower *Pimelea spinescens subsp. spinescens*; and two listed vegetation communities: *Natural Temperate Grassland of the Victorian Volcanic Plain* (Critically Endangered, equivalent to Victorian EVC 132 Plains Grassland; and EVC 654 Creekline Tussock Grassland), and *Subtropical and Temperate Coastal Saltmarsh* (Vulnerable, equivalent to EVC 9) (see Brett Lane and Associates 2002; Dann 2007; Ecology Australia 2010). The WTP coastal habitats provide the best known overwintering site in Victoria for the Critically Endangered Orange-bellied Parrot *Neophema chrysogaster* (Orange-bellied Parrot Recovery Team 2006).

As the site manager Melbourne Water is required under the terms of the Commonwealth *Environment Protection and Biodiversity Conservation Act* 1999 (hereafter referred to as the EPBC Act) to conserve habitat for significant wildlife at the WTP, while still meeting its core commitments to treat wastewater under regulation of the Environment Protection Authority Victoria.

The Environment Improvement Project at the WTP

During the late 1990s Melbourne Water initiated an Environment Improvement Project (EIP) to upgrade its wastewater treatment processes at the WTP as a direct response to the findings of the extensive Port Phillip Bay Environmental Study (Harris *et al.* 1996) and to meet the requirements of the Environment Protection Authority (EPA) Victoria waste discharge licence and the State Environment Protection Policy (Waters of Victoria) in the context of a rapidly growing human population in Melbourne. While the EIP as a whole pre-dated the EPBC Act, one of the final phases of the project, 'Post-effluent Reuse Stage 2', was referred to Environment Australia (now the Commonwealth Department of the Environment) for approval under the Act during June 2002.

The Department approved the Post-effluent Reuse Stage 2 phase of the EIP subject to certain conditions (Environment Australia 2002). Melbourne Water was, among other things, required to implement adaptive management of off-set habitat wetlands to mitigate possible adverse effects on waterbirds or the Growling Grass Frog from treatment upgrades in operational ponds. Our interpretation of the term adaptive management follows the definition: 'a management framework which has built into it the capacity to learn from management decisions and to change management strategies on the basis of improved knowledge' (DNRE 2002: 131). This paper describes how Melbourne Water has endeavoured since 2002 to implement adaptive management to protect the WTP's recognised biodiversity values during necessary sewage treatment upgrades.

Management Approach

Managing multiple wildlife populations at most large Australian sites entails working with large inherent variation and uncertainty. Therefore Melbourne Water seeks continual improvement of its management of biodiversity at the WTP using an adaptive management approach. We understand this to entail:

- precautionary and pro-active provision of 'compensatory' habitat to offset potential future negative effects;
- ongoing investigations to expand our knowledge base and allow sound scientific and data-based management decisions to be made;
- careful monitoring of target population/species' responses to management actions;
- sound management of habitat areas to ensure their value to the target population/species, and;
- frequent review and adjustment of management actions.

Compensatory habitat

Three large sewage treatment lagoons in the west of the WTP, scheduled for decommissioning during the EIP, were set aside to be managed for waterbirds – a key value contributing to the site's listing as a Ramsar Wetland. These lagoons, Lake Borrie, Western and T-Section, comprise 46 ponds and are protected by a *ca* 350 metre buffer zone within which cropping and other intensive land uses are not allowed (see Weston *et al.* 2009). In addition, a further 13 small ponds in the eastern part of the WTP are now managed as habitat ponds. Drains known to be used by Growling Grass Frogs are protected from disturbance or chemical spray drift, and watercourses are provided a 30-metre set-back, fenced to exclude stock. In addition, large-scale habitat improvement works were commenced at the decommissioned Western Lagoon during 2010, including the construction of a new ephemeral wetland intended to provide waterfowl and frog habitat.

Applied research

Since 2002 Melbourne Water has commissioned a large number of research and monitoring projects to improve the understanding of ecosystem structure and functioning at the WTP, and evaluate management actions. Too numerous and varied to detail here, these studies include investigations into waterfowl daily activity budgets (Mustoe and Waugh 2006; Mustoe 2009; Guay 2013), shorebird foraging ecology and movement patterns (Rogers *et al.* 2007, 2013), pond ecosystem structures (Mulder 2005), intertidal mixing zones and infaunal communities (Morris *et al.* 2010; Parry 2013; Parry *et al.* 2013), a trial of multiple effluent outlets (GHD 2013a), disturbance of birds by humans (Glover *et al.* 2011; Guay *et al.* 2013a, 2013b, in prep.; McLeod *et al.* 2013; Weston *et al.* 2012, in press), and satellite tracking of waterfowl (Guay and O'Shea 2012). These studies have not only informed management of the WTP but have contributed to publications that will assist other wetland managers.

One study of particular note, before the decommissioning of Lake Borrie, investigated the likely effects on waterfowl populations at the site of predicted changes in water chemistry following decommissioning. This early model using two years of bird count data and water chemistry sampling predicted that reduced nutrients in the waters of Lake Borrie would have the greatest effect on filter-feeding ducks, perhaps reducing their numbers there by up to 90% (Loyn *et al.* 2002).

Monitoring of significant populations

Commonwealth Department conditions require nine nationally significant wildlife populations at the WTP to be closely monitored to ensure these populations are being sustained and to evaluate management of the site. These populations are: Growling Grass Frog; migratory shorebirds; five waterfowl guilds (dabbling ducks, filter-feeding ducks, diving ducks, grazing waterfowl and grebes); Pied Cormorant *Phalacrocorax varius*; and Straw-necked Ibis *Threskiornis spinicollis*. Results of intensive monitoring of nationally significant wildlife populations are considered in terms of pre-designated 'trigger points' for further investigation or additional management action. These trigger points vary between target populations but typically include ≥10% declines in number over three successive years, or >25% declines over three years. Population declines must be site specific to the WTP to qualify as trigger points for new management at that site. Results are compiled annually, assessed and reported to the Commonwealth Department annually. Loyn *et al.* (2014b) describe the waterfowl monitoring program in more detail.

Management of habitat areas

Management is directed by a number of plans, from the high level WTP Ramsar Site Management Plan (Ecology Australia 2010) to more detailed plans targeting specific values, such as the Growling Grass Frog Management Plan (Renowden 2012) and Terrestrial Margins Management Plan (Wrigley-Dillon 2011). Table 1 lists management interventions undertaken at the WTP since 2002 to improve or maintain habitat. This table does not list ongoing maintenance (e.g. vermin control, weed control, revegetation and water level manipulation in habitat ponds), grassland management (e.g. ecological burn-offs, fencing and controlled grazing), or other actions (e.g. construction/maintenance of bird-hides, track maintenance, signage, roadside slashing, etc.).

Results of Monitoring

During 2005 two trigger points for increased or changed management intervention were activated: the three-year average number of Growling Grass Frog detected during surveys declined by more than 25%, and the average number of Straw-necked Ibis foraging over former grass filtration paddocks declined by more than 10% in the year after sewage irrigation ceased. Remedial management action was undertaken—with additional ponds set aside to be managed specifically for Growling Grass Frogs and increased paddock irrigation respectively—and monitoring after 2006 showed these populations returned to levels where triggers were not activated.

During 2006 one trigger point was reached when the three-year average number of filter-feeding ducks recorded at the site during 2004–06 was more than 25% lower than the average number recorded during 2001–03. This three-year average trigger for filter-feeding ducks was again reached during both 2007 and 2008. During 2008 a further two guilds of waterfowl reached trigger points when diving duck numbers declined by >10% for three consecutive years and the three-year average number of grebes and diving ducks declined by >25%. During 2009—after 12 years of severe drought and water shortages at the WTP—four trigger points were reached: the three-year average number of Growling Grass Frogs, diving ducks, filter-feeding ducks and grebes all declined by >25%. This was despite the fact that some populations saw increases during late 2009 as the so-called 'Millennium Drought' finally broke across south-eastern Australia.

Table 1. Commitments and implementation of on-ground works associated with habitat wetlands at the Western Treatment Plant.

	Commitment	Implementation
(1)	Experiment to attract filter-feeding ducks (Melbourne Water 2003).	115E Borrow Pit habitat creation works (April 2005).
(2)	Given lack of success of action 1, 85WC-9 earmarked for habitat creation (Melbourne Water 2005).	(a) Allocated as habitat during summer 2005/06 and first drawdown probably spring 2006. (b) Timber added, 2010 and 2011. (c) Major desilt and vegetation clearance (1–16 May 2012).
(3)	Trigger point for Growling Grass Frog reached in 2005. (a) From 2005/06 breeding period 115E Borrow Pit and 5W-9 to be managed for this species (Melbourne Water 2006) (b) 115E Borrow Pit to be drained of stagnant water before re-flooding in September 2010 in an attempt to attract frogs back to these ponds (Melbourne Water 2010).	(a) Done every breeding period from 2005/06. (b) Dewatered in 2007 and again during July 2009, 2010, 2011 and 2012.
(4)	Trigger point for filter-feeding ducks reached in 2006. Undertake habitat improvement works (e.g. positioning of fallen timber) in a large decommissioned sewage treatment pond to mimic conditions in Pond 9 of Lake Borrie—which is the most attractive pond for Pink-eared Ducks at the Western Treatment Plant. This will build upon experience gained during similar habitat creation works at 115E Borrow Pit (Melbourne Water 2007). A further trial of habitat modification, through placing fallen timber in a former sewage treatment pond, is to be conducted during 2007 (Melbourne Water 2007).	Wood added to Lake Borrie Pond 24 (March 2007).
(5)	Experimentally manipulate water flows to Lake Borrie, reducing inflows to extend cumulative retention time in the lake, to determine whether it is possible to promote phytoplankton stocks as food for filter-feeders (Melbourne Water 2007).	See Table 2.
(6)	Trigger point for filter-feeding ducks reached in 2007. Certain decommissioned ponds will be drained, some vegetation allowed to grow, and then flooded to promote nutrient cycling and the production of food for waterfowl (Melbourne Water 2007)	(a) Lake Borrie Ponds 28 and 29 first drawndown from February 2008, flooded again in March 2009; with drawdowns every subsequent spring (sometimes prolonged partial drawdowns due to rainfall and in seeping). (b) Lake Borrie Pond 12 drawdown from February 2013 (after several years of wet delaying response).
(7)	Trigger points for filter-feeding ducks, diving ducks and grebes reached in 2008. Test whether the addition of carbon (in the form of straw or seagrass wrack from the nearby coastline) and/or nutrients (in the form of fertiliser) promotes food resources for waterfowl, particularly filter-feeding ducks, in ponds at Lake Borrie (Melbourne Water 2009).	500 kg straw added to Lake Borrie Pond 22 during September to November 2011 as experiment to assess effect
(8)	Decommissioning of Western Lagoon provides opportunity to re-establish saltmarsh and otherwise improve this area from a habitat perspective. Large-scale habitat improvement works at Western Lagoon (Melbourne Water 2009).	(a) Ponds 4 and 5 drawndown (2008), sludge removed, and bunds removed (March/April 2010) to rehabilitate to saltmarsh. (b) Creation of Q4 Wetland (April 2010).
(9)	Trigger points for filter-feeding ducks, diving ducks and grebes reached in 2009. If required, sludge or biosolids could be added to the Lake Borrie system to ascertain whether this will promote pond productivity sufficiently to attract and support large numbers of waterfowl (Melbourne Water 2009).	This is the subject of advanced planning but is subject to approvals from various regulating authorities. Works are likely to proceed during 2015.

Table 1. Continued.

	Commitment	Implementation
(10)	Trigger point for Growling Grass Frog reached in 2009. T-Section Pond 4 will be managed as an additional Growling Grass Frog pond (Melbourne Water 2010).	Drawdown of T-Section Pond 4 to promote vegetation growth before flooding during Growling Grass Frog breeding period (2011 and 2012).
(11)	Opportunity for creation of new Growling Grass Frog pond arose during desilting works.	Growling Grass Frog pond created at 35E Pond 9 (November 2006).
(12)	Installation of cormorant nesting platforms.	Twenty cormorant nesting poles with 100 platforms installed (August 2004).
(13)	Maintain water levels in T-Section drain over summer (Environment Australia 2002).	Pipe to T-Section drain from T-Section Pond 3 installed (2010).
(14)	Improve water level management at T-Section ponds (VWSG request).	Cross pipes installed between T-Section Pond 1 and Ponds 6 and 7 (October 2006).
(15)	Recommendation of WTP Biodiversity Conservation Advisory Committee.	35E Pond 8 vegetation clearance (April/May 2010).
(16)	Trial multiple outlets for effluent to enrich mudflats (Steele 1996).	Multiple outlets installed 2005/06 (December 2007; GHD 2013a).
(17)	Recommendation of WTP Biodiversity Conservation Advisory Committee.	Drawdown T-Sections Ponds 5 to 7 (done for many years, since at least the 1990s).
(18)	Recommendation of WTP Biodiversity Conservation Advisory Committee.	Weir reset at outlet to Ryans Swamp (February 2005).
(19)	Recommendation of WTP Biodiversity Conservation Advisory Committee.	35E Pond 9 desilt (2008/09).
(20)	Growling Grass Frog management plan (Organ 2003; Renowden 2006, 2009, 2012).	Overwintering harbour (timber and rock) introduced to Growling Grass Frog ponds 5W-9, 5W-10 (2011).
(21)	Request of Clive Minton, to provide shorebird roost (and trapping) site.	Clearing of vegetation from the spit at 5W Pond 9 (April 2005).
(22)	Identified maintenance requirement.	Desilt of outlet to 270S Borrow Pit (January 2006).
(23)	Identified maintenance requirement.	Desilt of outlet to 35E Pond 9 (January 2006).
(24)	Identified maintenance requirement.	Desilt of inlet pipe at Western Lagoon Pond 3 (15 January 2013)
(25)	Staff initiative.	Wood added to Western Lagoon Ponds 1, 2 and 3 (2011).
(26)	Request of the Orange-bellied Parrot Recovery Team.	Supplementary feeding of Orange-bellied Parrots at 55E Lagoon (two winters, one was June 2011).
(27)	Identified maintenance requirement.	Couch Grass removed from upper ponds of Lake Borrie South.
(28)	Recommendation of WTP Biodiversity Conservation Advisory Committee.	35E conservation ponds put onto a two-year drawdown cycle from late 2012.
(29)	Identified need to improve record keeping of water depths in habitat ponds (during drought).	Depth gauges installed.
(30)	Identified need to improve record keeping by community-based birdwatchers.	Pond name boards installed.
(31)	Identified need to improve, community-based monitoring of works sites, beach seaweed deposition, and *Phragmites/Typha* patches.	Fluker posts installed (Nov/Dec 2011).
(32)	2009 Growling Grass Frog trigger point reached.	T-Section Pond 7 and Q4 wetland allocated to Growling Grass Frog habitat.

Altogether four population 'trigger points' were reached during 2012. The rolling three-year average number of (1) diving ducks, (2) filter-feeding ducks, and (3) grebes declined by >25% over the previous three-year period. In addition, (4) the number of Straw-necked Ibis counted foraging across paddocks at the WTP declined by >10% between 2011 and 2012. Thus a number of trigger points have been reached but remedial management has apparently led to a recovery of affected populations, with the exception of filter-feeding ducks and diving ducks and, more recently, Straw-necked Ibis.

Thus post-decommissioning surveys of waterfowl have shown a decline which has been most marked among the filter-feeding ducks, as forecast (Loyn *et al.* 2002). Declines have been most marked at Lake Borrie, which formerly supported the highest numbers of waterfowl of any lagoon at the WTP. The causal factors behind this observed result are unclear. Waterfowl populations will likely have declined as a result of reduced breeding success during the long drought of 1997–2009, and then through dispersal to inland sites to breed following the breaking of the drought (e.g. Loyn *et al.* 2010; Loyn and Swindley 2012; Loyn *et al.* 2014a; Loyn *et al.* 2014b). But the close agreement between modelled response and observation, coupled with an observed reduction in foraging activity at Lake Borrie following its decommissioning (Hamilton *et al.* 2002; Mustoe and Waugh 2006; Mustoe 2009; Guay 2013) suggest that Lake Borrie in 2013 no longer supported very high numbers of filter-feeding ducks.

Management Response at Lake Borrie

In response to these repeated trigger points for filter-feeding and diving ducks at Lake Borrie our adaptive management approach proposes an escalating management intervention (Melbourne Water 2007, 2009) involving the following approaches:

(1) Experimentally manipulating water flows to Lake Borrie, reducing inflows to extend cumulative retention time in the lake, to determine whether it is possible to promote plankton stocks as food for filter-feeders.

(2) Undertaking habitat improvement works (e.g. positioning of fallen timber) in a large decommissioned sewage treatment pond to mimic conditions in Pond 9 of Lake Borrie, which was the most attractive pond for Pink-eared Ducks *Malacorhynchus membranaceus* at the WTP.

(3) Temporarily draining some decommissioned ponds over summer, allowing vegetation to grow—fixing carbon and recycling nutrients from sludge deposits—and then flooding during late winter/spring to promote the production of food for waterfowl the following summer. Given the generally nutrient-rich water and sediments of most ponds at the WTP, this management technique requires careful timing and observation to avoid avian botulism outbreaks.

(4) Supplying nutrients, in some form, to selected decommissioned ponds to promote waterbird food resources in these ponds.

(5) Finally, if required, sludge or biosolids will be added to the Lake Borrie system to promote pond productivity and attract waterfowl. Trucking in concentrated biosolids was considered likely to be inadequate for the purpose and so construction of a new sewage transfer pipeline to replace the decommissioned Main Western Carrier to Lake Borrie will be required.

a. Water inflows manipulation

The first action was carried out from Lake Borrie's decommissioning in December 2004 to winter 2008 (Table 2), with no noticeable success in attracting increased numbers of filter-feeding ducks. Thereafter, from spring 2008, moderate to high inflows were tried, with spring flushes mimicked during some years. While there was no observed increase in filter-feeding ducks at Lake Borrie following this change in management approach, the data are confounded by the severe drought prevailing at that time. Apart from the drought effects on waterfowl populations, environmental water requests could not always be met due to water shortages, blue-green algae contamination of source lagoons, and operational or maintenance constraints.

Simple daily activity budgets have been determined for waterfowl guilds at Lake Borrie Pond since 2000. This provides an indication of how birds are using the pond and was suggested as a possible early warning indicator of changes

Table 2. Planned environmental flows to Lake Borrie post-decommissioning as a sewage treatment lagoon.

Season	Request (ML/wk)	Intent	Comment (where required)
Summer 2004	N/A	Low inflows to allow time for uptake of nutrients from sludge, and development of phytoplankton communities.	25W Lagoon offline and no water supply available.
Autumn 2005	210	Low inflows	25W Lagoon offline and no water supply available.
Winter 2005	210	Low inflows	Drought affected water availability and meant loading to Lake Borrie did not always meet our requested volumes.
Spring 2005	210	Low inflows	"
Summer 2005	210	Low inflows	"
Autumn 2006	210	Low inflows	"
Winter 2006	210	Low inflows	"
Spring 2006	210	Low inflows	"
Summer 2006	175	Low inflows	"
Autumn 2007	210	Low inflows	"
Winter 2007	280	Moderate inflows, given lack of response to low inflow strategy	"
Spring 2007	280	Moderate inflows	"
Summer 2007	200	Low inflows	Extreme drought and water shortages necessitated reduction in planned inflows.
Autumn 2008	230	Moderate inflows	Drought affected water availability and meant loading to Lake Borrie did not always meet our requested volumes.
Winter 2008	280	Moderate inflows	"
Spring 2008	350	Change to high inflows	"
Summer 2008	350	High inflows	"
Autumn 2009	~316	High inflows, with weekly fluctuations	"
Winter 2009	~316	High inflows, with weekly fluctuations	"
Spring 2009	200, with 420 one week	Moderate inflows, with spring flush mimicked	Pulsed loading.
Summer 2009	290	Moderate inflows	
Autumn 2010	267	Moderate inflows	
Winter 2010	250	Moderate inflows	
Spring 2010	280, with 600 one week	Moderate inflows, with spring flush mimicked	Pulsed loading.
Summer 2010	300	High inflows, with weekly fluctuations.	
Autumn 2011	270	Moderate inflows, with weekly fluctuations.	
Winter 2011	280	Moderate inflows.	
Spring 2011	280, with 600 one week	Moderate inflows, with spring flush mimicked	Pulsed loading.
Summer 2011	300	High inflows, with weekly fluctuations	
Autumn 2012	270	Moderate inflows	
Winter 2012	280	Moderate inflows	
Spring 2012	280, with 600 one week	Moderate inflows, with spring flush mimicked	Pulsed loading.
Summer 2012	300	High inflows.	

in habitat quality (Hamilton *et al.* 2002). Subsequent studies found a decline between 2000 and 2006 in the proportion of time birds of some guilds spent foraging, as opposed to time spent resting, moving or preening, at Pond 9 (Hamilton *et al.* 2002; Mustoe and Waugh 2006; Mustoe 2009). However, the most recent observations show some increase between 2007 and 2012 in time spent foraging at Pond 9 by filter-feeding waterfowl (Guay 2013). The only management intervention applied to Lake Borrie Pond 9 since 2007 has been the increase in inflows (Tables 1 and 2). The possibility that pulsed through flow rates might benefit filter-feeding ducks needs to be investigated further.

b. Timber addition

The large Pond 9 of Lake Borrie was established at a natural depression, where many River Red Gum *Eucalyptus camaldulensis* trees became permanently waterlogged. These trees provide valuable habitat complexity and a source of carbon to this pond, which supports many more waterbirds than other ponds in the Lake Borrie system. However, these trees have now been waterlogged for around 80 years and are rapidly becoming lost as habitat, although not as a carbon source. During March 2007 Melbourne Water added felled trees to Pond 24 of Lake Borrie to replicate the carbon source in Pond 9. The volume and distribution of added timber was limited by the reach of excavator arm, but the timber that was added was successful in attracting waterfowl (Melbourne Water 2008). Again, these did not include significant numbers of filter-feeding ducks.

c. Seasonal drawdowns

Two ponds of Lake Borrie, Ponds 28 and 29, were drawn down from February 2008 to March 2009 to allow access to place more timber, as well as to allow vegetation growth to fix carbon and sediment exposure and oxidation (Melbourne Water 2009). It was planned to leave these two ponds shallow or dry over winter and then to flood them in late summer to coincide with the peak demand for wetland habitat for waterfowl after inland waters dry up. However, Black-winged Stilts *Himantopus himantopus* attempted to breed in the lowered ponds. Consequently the ponds could only be flooded in March/April. In subsequent years these two ponds have proved highly attractive to a range of waterbird species during the summer drawdown. However, the ponds have not attracted large numbers of filter-feeding ducks, which are our target, when flooded. This emphasises the difficulty in managing natural systems, and reinforces the need for ongoing, research-based and flexible management.

d. Nutrient addition

'Lake Borrie' comprises 30 ponds in two discrete wetland systems: Lake Borrie North (Ponds 1 to 14, less Pond 10) and Lake Borrie South (Ponds 10 and 15 to 30) (Fig. 3). A simple experimental design was decided, with Lake Borrie North as the 'control' chain of ponds and Lake Borrie South as the treatment chain. Within these two wetland systems Pond 9 supports the greatest diversity of waterfowl of any of the Lake Borrie ponds, whereas the equally large Pond 24 generally supports few waterfowl. To influence Pond 24 we added nutrients to Pond 22, upstream in the chain of ponds. The equivalent pond in the Lake Borrie North chain of ponds is Pond 6 (Fig. 3).

The form of nutrient to be added was strictly limited because of operational and safety considerations. No nutrient addition was permitted to cause effluent discharges to Port Phillip Bay (from Ponds 14 and 30) to exceed Melbourne Water's waste discharge licence issued by EPA Victoria. Nitrogen addition, in the form of commercial fertilizer, was vetoed because of safety concerns in purchasing, transporting and storing this potentially explosive compound. Seagrass wrack was considered but excluded because of concerns that this might increase the salinity of ponds and the possibility that wrack would add excessive potassium to ponds (D Rogers, Arthur Rylah Institute for Environmental Research, pers. comm.).

Eventually a simple experiment adding carbon, in the form of straw, was decided upon. Straw provides a large surface area for microbial slime to adhere to and it was expected that this microbial slime could act as the basis of a food chain and promote zooplankton populations in the treatment and downstream. Over time decomposing straw should add particulate organic carbon (POC) to the water column.

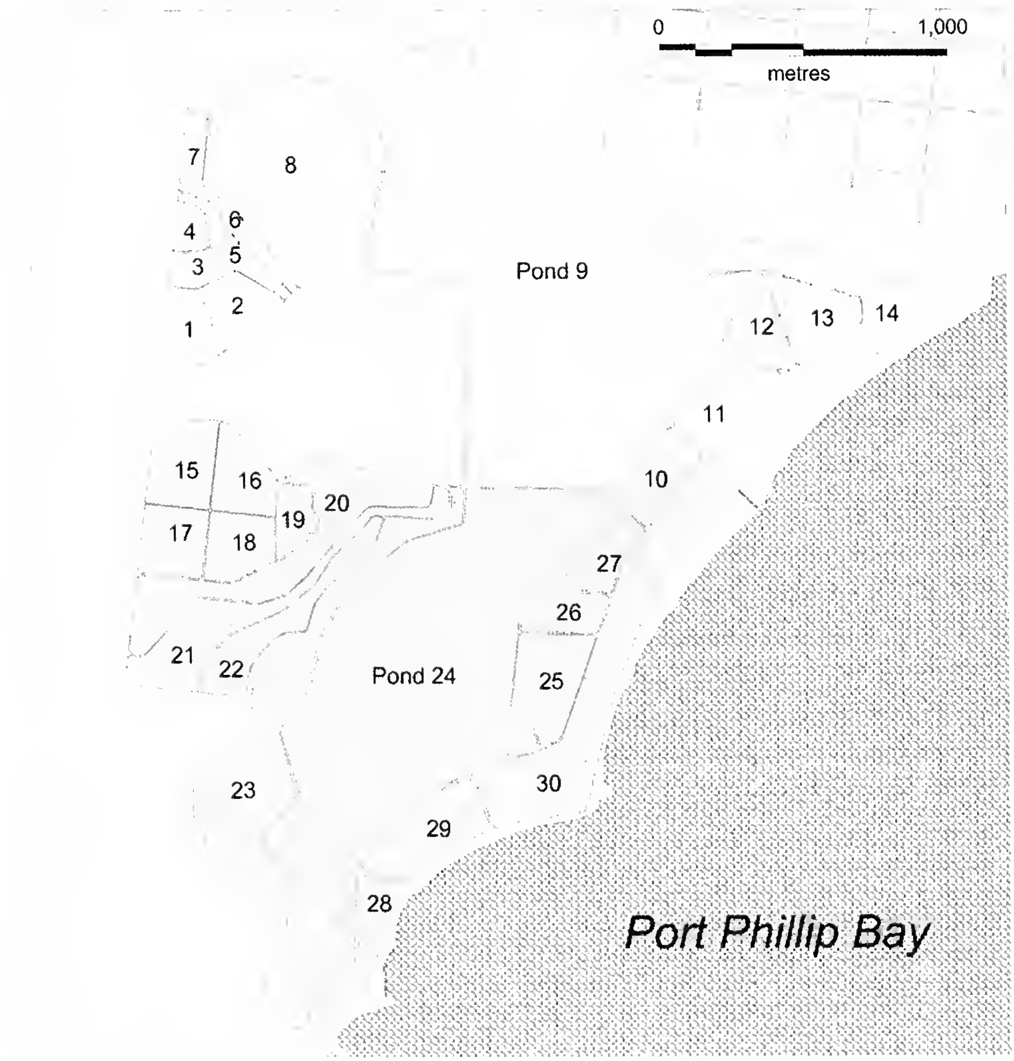

Fig. 3. The Lake Borrie system of 30 ponds in two treatment streams: Lake Borrie North (Ponds 1 to 14, except Pond 10) and Lake Borrie South (Ponds 10 and 15 to 30).

POC is considered to be an important source of food for invertebrates of intertidal mudflats (prey of migratory shorebirds) nourished by effluent discharges from Lake Borrie. There is also the possibility that straw provides cover for zooplankton to hide from predators, promoting zooplankton populations in ponds (e.g. Street 1982; Butler *et al.* 2005). A 500 kg bale of straw (estimated composition *ca* 40% carbon) was sliced into *ca* 10-cm thick 'biscuits'. These were added to Pond 22 over three weeks between September and November 2011.

Water chemistry, zooplankton, phytoplankton and zoobenthos (macroinvertebrate) sampling was initiated in the four Lake Borrie ponds from November 2009, following a scoping study (Bryant and Papas 2008). This sampling takes place four times per year and is planned to provide ongoing data on potential prey for waterfowl in these ponds before, during and after nutrient addition (GHD 2013b).

Benthic macroinvertebrates

Airlift sampling was utilised to sample macroinvertebrates in the pond sediments. The apparatus consisted of a 12 litre SCUBA tank attached to a base unit that was in turn attached to a 250 μm mesh bag and collecting vessel. The base net was placed firmly on the pond bottom and when the pressurised air tank was opened the benthic macroinvertebrates, without excess sediment, were collected. At each sampling point three sub-samples were taken, with each consisting of a 6 second burst. Sub-samples were combined and treated as one sample. These samples were preserved in 70% ethanol and retained for identification in the ALS Aquatic Ecology laboratory (now GHD Aquatic Ecology laboratory). All macroinvertebrate samples were processed using stereo or compound microscopy, with organisms identified to species (where possible) using published keys listed in Hawking (2000). All macroinvertebrates were picked and enumerated with the exception of samples with superabundant Oligochaeta (worms) where approximately 10% of all worms were identified and total worm abundance was estimated based on this sub-sample.

Zooplankton

Zooplankton was collected using a modified 11 L perspex Schindler-Patalas trap. At each sampling point five sub-samples were collected from at least 10 cm below the water surface. Sub-samples were consolidated into one sample, filtered through a 60 μm mesh net and preserved in 70% ethanol. Samples were then dispatched to the Australian Waterlife laboratory for processing and identification. Samples were processed using stereo or compound microscopy, with organisms identified (where possible) to species level. Zooplankton and rotifer abundances are reported as the total number of organisms per litre.

Phytoplankton

Whole water samples per sampling point consisted of a single sub-surface 1 L water sample. These samples were stored at approximately 4°C, protected from light, until they were transferred to the ALS Botany laboratory. Samples were then preserved using an appropriate volume of Lugol's Solution and identified to species level where possible, using the Sedgwick-Rafter method, under compound microscopy. Phytoplankton abundances are reported as cells/units per millilitre.

Results for all three taxonomic groups were summarised by abundance and richness (number of taxa recorded) for two periods: before (four sampling runs per year, November 2009 to July 2011, or 8 replicates) and after the addition of straw (8 replicates). The resulting 2009-11 and 2012-13 means were compared using a two-tailed *t*-test, assuming unequal variances.

Results of the Lake Borrie Nutrient Addition

The results of zooplankton, zoobenthos and phytoplankton sampling are summarised in Tables 3–5 and Figs 4–6. Within the limits imposed by our simplistic experimental design there is no evidence that the addition of straw was associated with any increase in waterfowl food resources in treatment ponds. All ponds showed increases in zoobenthos (macroinvertebrates) abundance in the periods 2009–11 and 2012–13 and this increase was not larger in the treatment ponds 22 and 24 (Fig. 4a, Table 3a). Interestingly, the pond that had straw added, Pond 22, displayed significant declines in macroinvertebrate taxonomic richness ($t = 3.46$, $P < 0.01$) in contrast to the other three ponds (Fig. 4b, Table 3b). All four ponds showed significant declines in abundance of zooplankton (including rotifers) between the two periods (Fig. 6a, Table 5a). Patterns of change in zooplankton taxonomic richness were similar in both the treatment and control ponds. Zooplankton richness decreased in higher ponds while increasing in both downstream ponds (Fig. 6b, Table 5b). There was an apparent increase in phytoplankton abundance in Pond 22 ($t = 2.44$, $P < 0.05$), where the straw was added, in contrast to the other three ponds (Fig. 5a, Table 4a).

Discussion

The value of long-term monitoring at multiple sites

As we obtain further monitoring data it is apparent that significant wildlife populations at the WTP vary widely between years, making interpretation and evaluation of management

Table 3a. Abundance of zoobenthos (macroinvertebrates) per sample from four ponds before and after addition of straw to Pond 22.

Pond	2009-11	2012-13	*t*-test
LB Pond 6 Control	Mean = 102.61 SE = 23.20 *N* = 28	Mean = 255.54 SE = 30.82 *N* = 28	$t = 3.9647$ $P < 0.01$
LB Pond 9 Control - downstream	Mean = 110.00 SE = 16.75 *N* = 44	Mean = 142.54 SE = 22.33 *N* = 26	$t = 1.1657$ n.s.
LB Pond 22 Treatment	Mean = 197.69 SE = 34.84 *N* = 32	Mean = 230.10 SE = 28.69 N = 28	$t = 0.7175$ n.s.
LB Pond 24 Treatment - downstream	Mean = 123.52 SE = 21.25 *N* = 44	Mean = 198.64 SE = 52.20 *N* = 28	$t = 1.3327$ n.s.

Table 3b. Diversity (taxa recorded) of zoobenthos (macroinvertebrates) from four ponds before and after addition of straw to Pond 22.

Pond	2009-11	2012-13	*t*-test
LB Pond 6 Control	Mean = 9.75 SE = 0.75 *N* = 28	Mean = 9.96 SE = 0.41 *N* = 28	$t = 0.2503$ n.s.
LB Pond 9 Control - downstream	Mean = 8.57 SE = 0.48 *N* = 44	Mean = 9.96 SE = 0.58 *N* = 26	$t = 1.8424$ n.s.
LB Pond 22 Treatment	Mean = 14.09 SE = 0.76 *N* = 32	Mean = 10.86 SE = 0.54 *N* = 28	$t = 3.4628$ $P < 0.01$
LB Pond 24 Treatment - downstream	Mean = 9.07 SE = 0.44 *N* = 44	Mean = 10.00 SE = 0.76 *N* = 28	$t = 1.0666$ n.s.

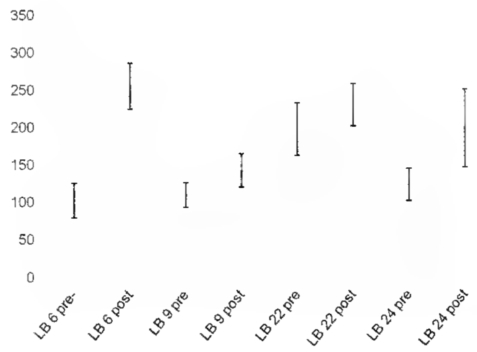

Fig. 4a. Zoobenthos (macroinvertebrate) abundances, per sample, in four ponds, 2009-11 and 2012-13. Standard error bars are shown.

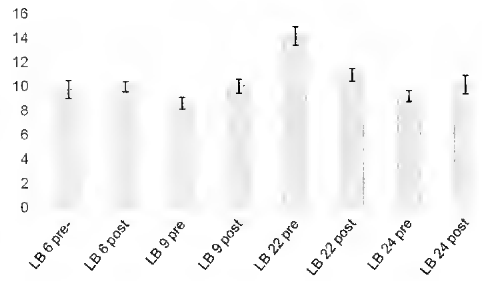

Fig. 4b. Zoobenthos (macroinvertebrate) diversity in four ponds, 2009-11 and 2012-13. Standard error bars are shown.

efforts problematic. It is difficult to assess to what degree downward trends in population are site-specific, and due to treatment process changes at the WTP, or more widespread and resulting from regional effects, such as long-term drought. Results from long-term surveys of wetlands across south-eastern Australia show that numbers and breeding of waterbirds declined significantly during the drought (Kingsford *et al.* 2005; DEC 2006; Chambers and Loyn 2006; Kingsford and Porter 2007; Norman and Chambers 2010). Thus, observed declines in waterbird populations at the WTP in recent years were likely to be heavily influenced by regional, mostly ex-situ, population trends and reduced breeding success (Loyn *et al.* 2014a). Indeed, an assessment of waterfowl numbers in Victoria shows that the WTP supported a significantly greater proportion of the State's waterfowl during the 1997–2009 drought than in previous years, with around 70% of the waterfowl counted during the Summer Waterfowl Count recorded at this one site.

Total numbers of waterfowl counted during the Victorian Summer Waterfowl Count increased dramatically in 2012 (almost six-fold) over the very low count for 2011. However, the EPBC 2002/688 trigger point (>25% decline in a three year period) was still exceeded for diving ducks, filter-feeding ducks and grebes for the three-year period 2009/10 to 2011/12. In each case this was due to the impact of the extremely low counts of 2010/11 on the three year mean. Numbers and distributions of Australian waterbirds at coastal sites are linked to patterns of rainfall and filling of ephemeral inland wetlands in our large, dry continent with its variable rainfall. The first decade of the 21st century saw a major drought affect much of southern Australia, and this inevitably affected the number of waterbirds using the WTP as national populations declined. However, the site also played a major role in providing a drought refuge for native waterbirds over this period. Sudden declines in waterbird numbers at the WTP during 2008 and 2010 were clearly related to rains and flooding in inland northern Australia. The count increases of 2012 are likely to reflect a return of individuals that had moved north to breed, plus immature birds recruited during those breeding events.

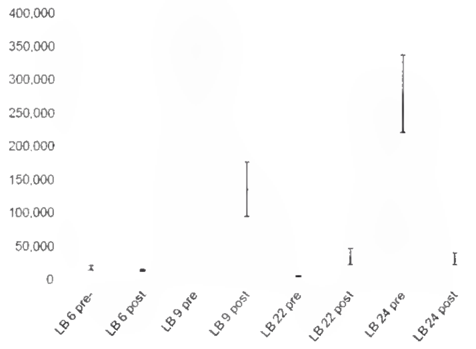

Fig. 5a. Phytoplankton abundances, per sample, in four ponds, 2009-11 and 2012-13. Standard error bars are shown.

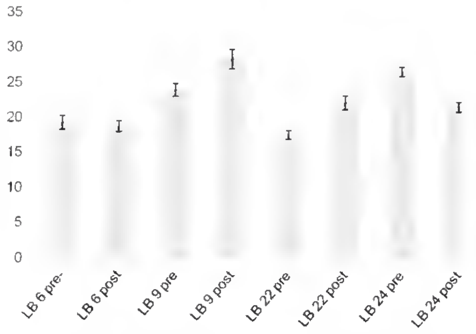

Fig. 5b. Phytoplankton diversity in four ponds, 2009-11 and 2012-13. Standard error bars are shown.

Thus it would seem that the WTP is becoming more important as a waterfowl refuge in recent years, not less so, and that population trigger points for extra management effort may be set unnecessarily low. Nonetheless, extensive works are being planned to return additional nutrients to Lake Borrie to meet Melbourne Water's commitments to respond to observed declines in waterfowl numbers following the EIP.

Pilot nutrient addition study

The lack of any result through the experimental addition of straw is disappointing. Straw was not the preferred additive but safety and other considerations meant the experiment was limited to this form and quantity of carbon. Further, the pond layout precluded multiple 'experimental' (nutrients added) and 'control'

Table 4a. Phytoplankton abundance per sample from four ponds before and after addition of straw to Pond 22.

Pond	2009-11	2012-13	*t*-test
LB Pond 6 Control	Mean = 17 172.86 SE = 3 411.26 *N* = 28	Mean = 13 383.54 SE = 1 267.21 *N* = 28	$t = 1.0413$ n.s.
LB Pond 9 Control - downstream	Mean = 1 074,052.57 SE = 368 026.45 *N* = 44	Mean = 135 814.79 SE = 40 706.98 *N* = 28	$t = 2.5339$ $P < 0.05$
LB Pond 22 Treatment	Mean = 5 107.47 SE =417.82 *N* = 32	Mean = 34 996.79 SE = 12,252.54 *N* = 28	$t = 2.4380$ $P < 0.05$
LB Pond 24 Treatment - downstream	Mean = 279 384.23 SE = 58 050.59 *N* = 44	Mean = 31 459.46 SE = 8783.84 *N* = 28	$t = 4.2228$ $P < 0.01$

Table 4b. Diversity (taxa recorded) of phytoplankton from four ponds before and after addition of straw to Pond 22.

Pond	2009-11	2012-13	*t*-test
LB Pond 6 Control	Mean = 19.18 SE = 0.90 *N* = 28	Mean = 18.75 SE = 0.78 *N* = 28	$t = 0.3593$ n.s.
LB Pond 9 Control - downstream	Mean = 23.95 SE = 0.89 *N* = 44	Mean = 28.32 SE = 1.40 *N* = 28	$t = 2.6364$ $P < 0.05$
LB Pond 22 Treatment	Mean = 17.53 SE = 0.63 *N* = 32	Mean = 22.11 SE = 0.96 *N* = 28	$t = 3.9797$ $P < 0.01$
LB Pond 24 Treatment - downstream	Mean = 26.5 SE = 0.65 *N* = 44	Mean = 21.5 SE = 0.67 *N* = 28	$t = 2.2762$ $P < 0.05$

ponds. Monitoring of pond biota in the experimental ponds will continue to ascertain if long-term effects become evident. This monitoring will be of value when Lake Borrie is re-engaged with the treatment process and receives partially treated sewage in the near future.

Conclusion

This paper described adaptive management from the perspective of the land manager of a large property. While targeted adaptive management occurs for a small number of individual species (e.g. Dowling and Weston 1999; Maguire *et al.* 2011), adaptive management of sites is more complex because of multiple and sometimes conflicting values and responses. Lessons from 12 years of attempting adaptive management of multiple biodiversity values at the WTP include the importance of long-term standardised monitoring of values, despite its cost. Population levels of target species can fluctuate widely between seasons, years, and longer term periods following climatic 'cycles'. To evaluate management effectiveness one needs long-term monitoring data, ideally including sites external to the managed property, to ascertain site-specific population responses.

Acknowledgements

A critically important element of Melbourne Water's adaptive management approach to biodiversity conservation at the site is the WTP Biodiversity Conservation Advisory Committee. This Committee was first formed in 1986, when it was called the WTP Wildlife Advisory Committee, and provides specialist advice to Melbourne Water on the conservation and management of the WTP's unique native biodiversity values. The Committee consists of representatives from

Table 5a. Zooplankton abundance (including rotifers) per sample from ponds before and after addition of straw to Pond 22.

Pond	2009-11	2012-13	*t*-test
LB Pond 6 Control	Mean = 3 550.25 SE = 1 286.11 $N = 28$	Mean = 189.28 SE = 56.26 $N = 28$	$t = 2.6108$ $P < 0.05$
LB Pond 9 Control - downstream	Mean = 23,448.72 SE = 5 690.78 $N = 44$	Mean = 387.62 SE = 53.04 $N = 26$	$t = 4.0522$ $P < 0.01$
LB Pond 22 Treatment	Mean = 6 151.03 SE = 1 489.47 $N = 32$	Mean = 140.11 SE = 23.24 $N = 28$	$t = 4.0351$ $P < 0.01$
LB Pond 24 Treatment - downstream	Mean = 19,900.95 SE = 4 488.65 $N = 44$	Mean = 509.15 SE = 95.82 $N = 28$	$t = 4.3092$ $P < 0.01$

Table 5b. Diversity (taxa recorded) of zooplankton (including rotifers) from four ponds before and after addition of straw to Pond 22.

Pond	2009-11	2012-13	*t*-test
LB Pond 6 Control	Mean = 12.11 SE = 0.67 $N = 28$	Mean = 9.43 SE = 0.51 $N = 28$	$t = 3.1825$ $P < 0.01$
LB Pond 9 Control - downstream	Mean = 8.20 SE = 0.43 $N = 44$	Mean = 10.29 SE = 0.49 $N = 26$	$t = 3.2007$ $P < 0.01$
LB Pond 22 Treatment	Mean = 11.25 SE = 0.52 $N = 32$	Mean = 9.57 SE = 0.56 $N = 28$	$t = 2.1914$ $P < 0.01$
LB Pond 24 Treatment - downstream	Mean = 7.61 SE = 0.37 $N = 44$	Mean = 9.36 SE = 0.58 $N = 28$	$t = 2.5347$ $P < 0.05$

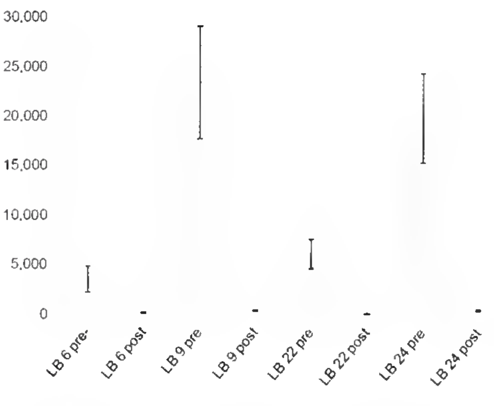

Fig. 6a. Zooplankton abundances, per sample, in four ponds, 2009-11 and 2012-13. Standard error bars are shown.

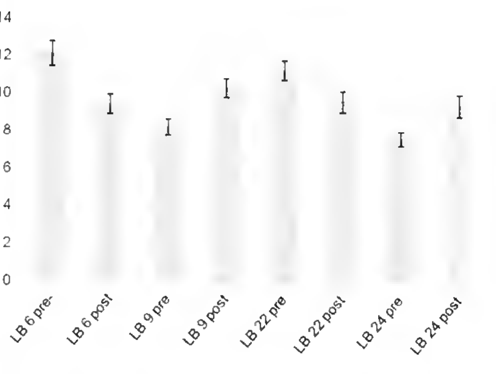

Fig. 6b. Zooplankton diversity in four ponds, 2009-11 and 2012-13. Standard error bars are shown.

relevant Government agencies, a number of environmental non-governmental organisations, and experts in a number of biological fields. The experience and knowledge of this Committee is of enormous value to Melbourne Water, which relies on its expert advice to guide biodiversity conservation, land management and operations at the WTP.

Numerous State government agencies, universities, consultancies and NGOs have been involved in research, monitoring and/or management planning at the WTP since 2002. Too numerous to mention 'we wish to express our gratitude to these bodies and individuals for the volume and quality of data now available to inform and evaluate management at the WTP.

We wish to thank specifically Dr Mike Weston (Deakin University) and a reviewer for their most helpful comments on drafts of this paper, and Ben James (Melbourne Water's former Werribee Agriculture Group) for his assistance with the practical implementation of the nutrient addition study.

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One Hundred and Three Years Ago

Report

A report of the excursion to the Metropolitan Farm at Werribee on Saturday, 21st October, was given by the leader, Mr. G. A Keartland, who said that there had been a good attendance of members, who, besides studying the natural history of the farm, had the opportunity of seeing how the sewage of Melbourne was disposed of. The bird-life, though numerous, was not very varied, and he had been somewhat disappointed in the results of the afternoon. It was interesting to find the little Grass-bird, Megalurus gramineus, nesting quite close to the shore-line. Regarding the botanical aspects of the excursion, Dr. Sutton reported that the flora was very similar to that on the eastern side of Port Phillip. The principal plants noted were:—*Atriplex cinereum* (tree form), *Salicornia arbuscula, S. Australis, Suceda maritima, Apium prostratum, Mesembryanthemum australe, Frankenia lavis, Samolus repens, Wilsonia rotundifolia, W. humilis,* and *Cotula filifolia.*

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Waste water not wasted: the Western Treatment Plant as a habitat for waterfowl

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Abstract
The Western Treatment Plant (WTP) is an outstanding example of a case where the waste water from a large city (Melbourne) is used to provide habitat for waterfowl and other birds. This paper provides a brief summary of the results of monitoring waterfowl numbers over 12 years, documenting the higb numbers of many waterfowl species that the WTP supports (often >100 000 birds in total). Active and adaptive management by Melbourne Water based on ongoing monitoring strives to maintain WTP's value for waterfowl. (*TheVictorian Naturalist* **131** (4) 2014, 147-149)

Keywords:

Introduction

Melbourne is a city of >4 million people and uses a lot of water (360 billion litres in 2011-12, http://www.melbournewater.com.au/waterdata). After use, the water is treated to remove harmful contaminants, and re-used for specific purposes or released to the sea. More than half this waste water is treated at the Western Treatment Plant (WTP) near Werribee where more than 40 billion litres of recycled water is produced per year (http://www. melbournewater.com.au/whatwedo/treatsewage/wtp/Pages/western-treatment-plant.aspx - accessed 14/3/2014). The WTP provides valuable habitat for waterfowl, and is a centrepiece of a Ramsar-listed wetland of international importance: Port Phillip Bay (western shoreline) and the Bellarine Peninsula. Hence Melbourne Water needs to manage the WTP to conserve waterfowl as well as to treat waste water. Since 2000, this has involved a program to monitor waterbirds. This was initiated by Melbourne Water as part of an Environment Improvement Program (EIP, 2003-05), designed to reduce nutrient inputs to Port Phillip Bay and meet requirements set by the Environment Protection Authority. The EIP involved phasing out land-based treatment processes in favour of ponding; intensifying treatment on two modernised lagoon systems and ceasing to use certain lagoons including Lake Borrie for sewage treatment. As this could affect Ramsar values (positively or negatively), the EIP became a controlled action under the Australian Government's *Environment Protection and Biodiversity Conservation Act* 1999 (EPBC Act). It was approved subject to continued monitoring and adaptive management. The adaptive management has many aspects, including creation and management of conservation ponds and a major capital works program to return nutrient-rich sewage to Lake Borrie through a new pipeline (Steele and Harrow 2014).

Waterfowl numbers have been counted across the whole WTP at two-monthly intervals as part of the monitoring program (Loyn *et al.* 2014). This paper uses that dataset as a case study to illustrate the value of using waste water in this way to provide habitat for waterfowl. The data are being analysed further to reveal how chemical, physical and climatic variables interact to influence the use of habitat by waterfowl, to inform future management. Here we define waterfowl as ducks, geese and swans (Anatidae) along with other birds that typically feed while swimming (grebes and coot). These species are also monitored more widely in Victoria through a Summer Waterbird Count (Loyn 1991; Murray *et al.* 2012; Purdey and Loyn 2012). This paper focuses on describing the numbers of waterfowl that use the WTP, making comparisons to numbers elsewhere in Victoria to show how the waste water is not wasted, but re-used to provide important habitat for these birds.

We note that the WTP also provides very valuable habitat for waders (shorebirds), cormorants, ibis and other birds, which are subject to parallel studies (Loyn *et al.* 2014).

Methods

Waterfowl were counted (by species) across the entire WTP six times per year from 2000 to the present. A single observer (RJS) conducted these counts after initial tests for observer variation. Notes were made on breeding activity when observed. Data were recorded separately for every discrete wetland at the WTP, including individual treatment ponds. However, this short paper just focuses on the total counts, presenting mean and maximum numbers of each species observed across the whole WTP from 2000 to 2012. We also show how we classified waterfowl into feeding guilds, which will be used for subsequent analyses.

Results

Waterfowl species are shown in Table 1, along with the guilds to which they have been assigned and their breeding status at the WTP. Mean and maximum counts of each species at the WTP are shown in Table 2. Maximum counts greatly exceed the means, reflecting marked variation between seasons and years (Loyn *et al.* 2014). Counts of waterfowl across all species exceeded 100 000 in many years.

Breeding was recorded frequently for Black Swan *Cygnus atratus*, Cape Barren Goose *Cereopsis novaehollandiae* and Chestnut Teal *Anas castanea*, less often for Pacific Black Duck *Anas superciliosa* and rarely for other species. Large numbers of Chestnut Teal bred successfully every year in nest boxes provided on one of the treatment ponds, Lake Borrie pond 9 (E Walker pers. comm.).

Discussion

Total counts of waterfowl on the WTP often constituted a large proportion (40-80%) of the totals recorded across Victoria on the Summer Waterbird Count for the same years (DEPI unpublished data), and ~70% of the total recorded during aerial surveys of Victoria in one year (2008) when an attempt was made to make a comprehensive aerial count (R Kingsford pers. comm.).

Several species were frequently present in higher numbers at the WTP than at other wetlands counted in the annual Summer Waterbird Count (DEPI unpublished data; e.g. Purdey and Loyn 2012). These included two of the filter-feeding ducks (Pink-eared Duck *Malacorhynchus membranaceus* and Australasian Shoveler *Anas rhynchotis*), two of the diving ducks (Blue-billed Duck *Oxyura australis* and Musk Duck *Biziura lobata*) and a grebe (Hoary-headed Grebe). Blue-billed Duck exceeded 12000 on one occasion, equivalent to what was then believed to be the global population (Garnett and Crowley 2000). Counts of Australian Shelduck *Tadorna tadornoides* often exceeded those at other wetlands, except in early years when even larger numbers congregated in the large saline wetlands of south-western Victoria.

One species (Australian Wood Duck *Chenonetta jubata*) that is very common on farm dams and freshwater wetlands was remarkably scarce at the WTP, probably because it is sensitive to salinity (Loyn *et al.* 2006). Waterfowl that occur commonly on a wide range of Victorian wetlands (e.g. Pacific Black Duck, Grey Teal and Black Swan) (Kingsford *et al.* 1999; Marchant and Higgins 1990; Murray *et al.* 2013) were well represented at the WTP, but numbers did not typically exceed numbers on certain wetlands elsewhere.

Clearly the WTP provides important habitat for very large numbers of waterfowl of many species. Its special contribution is as non-breeding habitat and drought refuge for filter-feeding ducks, diving ducks, Australian Shelduck and Hoary-headed Grebe, but it is used (to varying degrees) by all species. Waste water treatment plants elsewhere can also be valuable (Murray and Hamilton 2010; Murray *et al.* 2013) but to a much lesser extent because they are usually small and not actively managed for waterfowl. Sympathetic management of the WTP by Melbourne Water, including the provision of partially treated water in redundant treatment ponds with the aim of maintaining or enhancing waterfowl habitat quality, has made an important contribution to conserving waterfowl, especially at times of drought when there are limited amounts of habitat for them in this corner of Australia. Our waste water has not been wasted at all.

Acknowledgements

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Table 1. Waterfowl recorded at the Western treatment Plant 2000-2012, along with the feeding guilds to which they have been assigned and their breeding status at the WTP. # B=breeds regularly at WTP; NB=non-breeding visitor; RB=rarely breeds at WTP; V=vagrant

Species	Scientific name	Guild	Breeding status #
Magpie Goose	*Anseranas semipalmata*	Goose	V, RB
Musk Duck	*Biziura lobata*	Diving duck	RB
Freckled Duck	*Stictonetta naevosa*	Filter-feeding duck	NB
Cape Barren Goose	*Cereopsis novaehollandiae*	Goose	B
Domestic Goose	*Anser sp.*	Goose	V
Black Swan	*Cygnus atratus*	Swan	B
Australian Shelduck	*Tadorna tadornoides*	Grazing duck	RB
Australian Wood Duck	*Chenonetta jubata*	Grazing duck	NB
Pink-eared Duck	*Malacorhynchus membranaceus*	Filter-feeding duck	RB
Australasian Shoveler	*Anas rhynchotis*	Filter-feeding duck	RB
Northern Shoveler	*Anas clypeata*	Filter-feeding duck	V
Grey Teal	*Anas gracilis*	Dabbling duck	RB
Chestnut Teal	*Anas castanea*	Dabbling duck	B
Mallard	*Anas platyrhynchos*	Dabbling duck	V
Pacific Black Duck	*Anas superciliosa*	Dabbling duck	B
Hardhead	*Aythya australis*	Diving duck	NB
Blue-billed Duck	*Oxyura australis*	Diving duck	RB

Table 2. Mean, standard error and maximum counts of waterfowl species recorded at the Western Treatment Plant 2000-2012 (n=73).

Species	Mean	SE	Max
Magpie Goose	<1	<1	11
Musk Duck	1005	68	2103
Freckled Duck	65	14	554
Cape Barren Goose	14	2.2	65
Domestic Goose	<1	<1	2
Black Swan	2977	195	6879
Australian Shelduck	5623	1046	34922
Australian Wood Duck	8	1.8	109
Pink-eared Duck	12419	1517	50991
Australasian Shoveler	3759	449	17433
Northern Shoveler	<1	<1	1
Grey Teal	3651	279	12466
Chestnut Teal	3578	295	10914
Mallard or Domestic Duck	<1	<1	2
Pacific Black Duck	1001	80	3148
Hardhead	3429	402	15518
Blue-billed Duck	4078	402	12178

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Are vehicles 'mobile bird hides'? A test of the hypothesis that 'cars cause less disturbance'

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Abstract
We tested the 'cars cause less disturbance' hypothesis by comparing the flight-initiation distance (FID) evoked by a car versus a single walker for 38 species of waterbird ($n = 657$ standardised approaches). For the 15 species where we had sample size adequate for statistical testing ($n \geq 5$), we found that cars elicited shorter responses after controlling for starting distance. Within-species analyses revealed that this difference was significant in 8 of 15 species. Although mean FIDs for car approaches were always shorter than FIDs toward single walkers in the remaining species (7), the tests in those species lacked sufficient power to draw meaningful conclusions. Our results provide support for the hypothesis that birds respond to cars at shorter distances. The wide taxonomic breadth of species investigated suggests that this principle may be broadly applicable, at least in waterbirds. The results of this study and the FID estimates we present will allow development of meaningful stimulus-specific buffer zones to protect waterbirds from disturbance. (*The Victorian Naturalist* **131** (4) 2014, 150-155)

Keywords: Flight-initiation distance, disturbance, wetlands, vehicles, motorised transport

Introduction
'Disturbance' is the disruption of the normal activity or physiology of wildlife, such as birds, in the proximity of an agent such as a person or vehicle (i.e. a stimulus; Weston *et al.* 2012). One broadly accepted metric used to describe disturbance is flight-initiation distance (FID), the distance between a stimulus and a bird when an escape response is initiated (Blumstein 2003). While a range of internal and external factors influence FID (Guay *et al.* 2013a, Guay *et al.* 2013c), the type of stimulus is a little studied but important one (Mcleod *et al.* 2013). For example, birds alter aspects of their responses including their FIDs when presented with different stimuli (Miller *et al.* 2001, Glover *et al.* 2011; Schlacher *et al.* 2013b; McLeod *et al.* 2013). The type of stimulus which is permitted in a given area is often under the influence of land managers (e.g. Antos *et al.* 2007), and given that disturbance is regarded as a conservation problem in some circumstances (e.g. Schlacher *et al.* 2013a), understanding which stimuli are associated with which responses will aid the management of disturbance (Weston and Elgar 2005, 2007, Weston *et al.* 2012). Theoretically, managers could permit only certain stimuli, or prescribe stimulus-specific buffer zones to minimise disturbance (Weston *et al.* 2009; Weston *et al.* 2012; Mcleod *et al.* 2013). Currently, the vast majority of avian FIDs available worldwide are elicited by single walkers, thus there is a dearth of available information on other, common, stimuli (Mcleod *et al.* 2013).

One commonly held but little tested belief is the somewhat counter-intuitive idea that birds can be approached more closely in vehicles (henceforth 'cars') than on foot i.e. the 'cars cause less disturbance' hypothesis. Many birdwatchers and photographers use cars to approach birds because they believe this allows them to approach the birds more closely than would otherwise be possible on foot (authors, pers. obs.). However, this hypothesis has only rarely been tested, and the available results vary between species, with cars evoking shorter, similar, and longer FIDs compared with single walkers (reviewed in Mcleod *et al.* 2013). This study aims to test whether FIDs evoked by vehicles are shorter than those evoked by a single walker on foot by examining a greater taxonomic breadth of comparisons, and by carefully conducting experimental 'approaches' to birds.

Methods

Fieldwork was conducted at the Western Treatment Plant (WTP), Werribee, near Melbourne, Victoria (38°01'S, 144°34'E). Access to the plant is restricted; visitors are required to obtain a permit and register each visit. The common birdwatching areas of the WTP comprise various ponds and lagoons and the coastline, all of which are easily accessible via car or foot from the roads and paths that run throughout the plant, usually between every pond. The waterbirds at the WTP are thus exposed to some human activity, by cars and humans on foot, which is less than that evident in unrestricted areas such as urban parks (Glover *et al.* 2011).

Measuring Flight-Initiation Distances

Fieldwork took place between January 2011 and January 2012. All fieldwork was conducted between 0730 and 2100 hours, and as is customary and practical, only when it was not raining and in no stronger than moderate winds. We presented two types of stimuli to waterbirds within the WTP: single walker (1.4 m/s) and car (2.8 m/s). A stimulus type was randomly selected for each fieldwork day. For each stimulus type, we recorded FID rather than Alert Distance (AD) as it is a more reliable measure of response when multiple observers collect data (Guay *et al.* 2013b). FID was assessed by moving towards the focal bird at a constant pace. While approach speeds can influence FIDs (Glover *et al.* 2011) we used approach speeds which were typical of the stimuli being tested; our aim was to mimic realistic behaviour of each stimulus type. During the approach the observer/s were silent and made no sudden body movements. The distance at which we started an approach was recorded as the Starting Distance, and was maximised i.e. we used the longest Starting Distance possible (Blumstein 2003). The distance at which the bird walked, swam, dived, or flew away in response to the approach was recorded as the FID. Approaches were included only if the bird's response was determined to occur as a result of the approach. When a flock was approached, the FID was taken from the point at which the first individual showed a response to the approach. An approach was abandoned if it was unclear whether the bird was responding to the observer or to another potential stimulus, such as a bird of prey. Depending on the target bird's original location, we approached either directly or tangentially. All distances were measured using a laser rangefinder.

For all walking approaches the observers wore standard clothes (dark pants and a dark long-sleeved top). Different vehicles (from small hatchback to 4WD twin cab) were used for car approaches. All approaches were conducted on non-breeding adult waterbirds and only single-species flocks were approached. We attempted to avoid resampling individuals by closely monitoring where birds flushed to after an approach, before moving on to the next site.

Statistical analysis

For tangential approaches, FID was calculated as the Euclidian distance between the observer and the subject at the time escape behaviour was initiated by taking into account the bypass distance, the minimum distance between the focal bird and the path of the observer (Cooper 1997). Data for both approach types were pooled for further analysis.

We restricted our statistical analyses to 15 species for which we obtained at least five FID estimates per stimulus. We used a General Linear Model (GLM) to investigate the effect of species, stimulus type and their interaction. Starting Distance, which influences FID in birds (Blumstein 2003), varied between species ($F_{38,595} = 5.13$, $P <$ 0.001) but not between stimuli ($F_{1,595} = 2.42$, $P =$ 0.120). We controlled for the difference in Starting Distance between species by including it in our models. We further used GLMs to compare responses between stimuli for all 15 species individually. All distances were Log_{10} transformed prior to analyses. Summary statistics are presented as mean ± standard errors.

Results

We collected data for 657 approaches from 38 species (car, n=269; walker, n=388; Appendix 1). Results of the GLM for the 15 species for which we had five or more approaches for each stimulus (11 to 85 FIDs per species; car, 66.3 ±

2.6 m, n = 246; walker, 74.9 ± 2.3 m, n = 311) (adjusted $R^2 = 0.57$) revealed significant effects of Starting Distance (logged; $F_{1,526} = 333.68$, $P < 0.001$), stimulus (car vs. walker; $F_{1,526} = 53.36$, $p < 0.001$), and species ($F_{14,526} = 6.27$, $p < 0.001$); the interaction between species and stimulus was not significant ($F_{14,526} = 1.57$, $P = 0.084$) but was associated with high power (0.87). Within-species GLMs (Table 1) revealed that in every case cars had shorter FIDs compared with walkers. Eight of these fifteen comparisons were significantly different with the remaining seven having low statistical power.

Discussion

Few general principles are available to help explain FID in regard to environmental or internal factors (Weston *et al.* 2012), and here we have shown that the 'cars cause less disturbance' hypothesis has at least broad, and possibly universal, relevance across species. From a conservation management perspective, in no case were cars associated with longer FIDs, suggesting that at the WTP cars are effective mobile hides for observing many waterbirds. Additionally, cars can carry multiple people, thus arguably reduce the number of stimuli in an area (Mcleod *et al.* 2013). Birds at the WTP are exposed to many cars and perhaps fewer people on foot (though workers and birdwatchers are not uncommon on foot as they move around the vicinity of their cars; authors, pers. obs.). As for any behavioural study, confirmation of these results at different sites, with different prevailing regimes of cars and walkers, would be useful. Such a study could disentangle local learning on the part of the birds from perception and innate risk judgement of birds. It is important to note that many of the species involved in this study are migratory or nomadic and move in and out of the WTP every year (Hamilton and Taylor 2004; Hamilton *et al.* 2004). In particular, Australian Shelducks *Tadorna tadornoides* come to the WTP only during summer, thus limiting the opportunity for local adaptation.

Several caveats exist regarding the implications of the finding that cars reduce FIDs. Firstly, shorter FIDs in response to cars may not be adaptive in all circumstances. Cars cause direct bird mortality throughout the world and in Australia (Taylor and Mooney 1991; Schlacher *et al.* 2013a), presumably because responses are inadequate, absent or initiated too late. Such mortality can influence roadside bird populations (Bujoczek *et al.* 2011). The vehicle we used moved at slow speeds to mimic the prevailing speed of cars at the WTP; however, high vehicle speeds require earlier flight responses for successful evasion, and faster stimuli are associated with longer FIDs (Glover *et al.* 2011). At least some European birds apparently adjust their FIDs in regard to prevailing speed limits for traffic, but not to car speed *per se* (Legagneux and Ducatez 2013). Thus, the average speed of vehicles may influence FID and there may be a speed above which FIDs exceed those associated with walkers.

Table 1. Results of within-species GLMs for each species where at least five approaches were recorded for each stimulus. We report degrees of freedom (d.f.), F-value, P-value and observed power (Power). Species are presented alphabetically, by common name (BirdLife 2012).

Species	d.f.	F-value	P-value	Power
Australian Shelduck *Tadorna tadornoides*	1, 82	12.11	0.001	0.930
Australian White Ibis *Threskiornis molucca*	1, 26	0.53	0.475	0.108
Black Swan *Cygnus atratus*	1, 55	10.39	0.002	0.886
Chestnut Teal *Anas castanea*	1, 76	14.80	<0.001	0.967
Eastern Great Egret *Ardea modesta*	1, 18	17.35	0.001	0.976
Eurasian Coot *Fulica atra*	1, 16	2.22	0.155	0.289
Hardhead *Aythya australis*	1, 28	3.42	0.075	0.431
Little Black Cormorant *Phalacrocorax sulcirostris*	1, 8	0.06	0.816	0.055
Little Pied Cormorant *Microcarbo melanoleucos*	1, 60	6.56	0.013	0.712
Masked Lapwing *Vanellus miles*	1, 9	10.84	0.009	0.833
Pacific Black Duck *Anas superciliosa*	1, 34	2.33	0.136	0.317
Pied Cormorant *Phalacrocorax varius*	1, 15	5.14	0.039	0.564
Purple Swamphen *Porphyrio porphyrio*	1, 34	8.03	0.008	0.786
Straw-necked Ibis *Threskiornis spinicollis*	1, 31	0.83	0.369	0.143
White-faced Heron *Egretta novaehollandiae*	1, 20	0.00	0.986	0.050

Secondly, while cars may decrease FIDs among many species, they still have profound ecological effects on birds and their habitats (e.g. Reijnen and Foppen 1994) and can cause substantial levels of disturbance to birds especially when they are driving at speed and are common (e.g. Schlacher *et al.* 2013a; Schlacher *et al.* 2013b). Roads and tracks can cause a range of negative ecological effects (Forman and Alexander 1998), and the high mobility of cars means that the 'human footprint' is more expansive than for walkers alone, at least in many areas (McLeod *et al.* 2013). Clearly all impacts need to be considered by managers before the decision to promote a 'disturbance mediation by stimulus' strategy occurs.

The underlying mechanisms involved in birds discriminating between cars and walkers in terms of response remain unknown (see Weston *et al.* 2012). Each stimulus is associated with different visual and auditory cues, with cars being relatively novel evolutionarily. If size, colour and noise are used by birds to judge risk, then responses may vary with stimulus types (e.g. hybrid versus internal combustion cars), and this would be a useful subject of future study.

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Appendix 1. Raw flight-initiation distance (FID) data for all 38 species studied. We report sample size (n), mean start distance (SD) (± one standard deviation) and mean FID (± one standard deviation) for each stimulus separately. Blanks indicate no data were collected. Taxa are presented in alphabetical order by common name, and scientific names follow BirdLife (2012).

Species	Car			Walker		
	n	SD (m)	FID (m)	n	SD (m)	FID (m)
Australasian Darter *Anhinga novaehollandiae*				2	108.5 ± 19.0	77.4 ± 0.6
Australasian Grebe *Tachybaptus novaehollandiae*	1	30.5	17.7	3	70.6 ± 10.9	53.5 ± 2.2
Australian Pelican *Pelecanus conspicillatus*	4	208.1 ± 122.3	114.6 ± 51.7	9	212.5 ± 123.7	123.9 ± 104.9
Australian Shelduck *Tadorna tadornoides*	43	277.5 ± 154.5	106.9 ± 48.4	42	219.2 ± 131.2	122.3 ± 59.7
Australian White Ibis *Threskiornis molucca*	17	142.8 ± 78.7	56.2 ± 20.3	12	93.9 ± 41.2	48.6 ± 24.8
Black Swan *Cygnus atratus*	18	147.6 ± 89.4	66.4 ± 59.4	40	124.1 ± 97.0	78.3 ± 51.1
Black-tailed Native-hen *Gallinula ventralis*				6	85.0 ± 40.2	52.7 ± 16.8
Blue Billed Duck *Oxyura australis*				3	85.1 ± 54.7	68.3 ± 36.1
Cape Barren Goose *Cereopsis novaehollandiae*				5	119.6 ± 58.5	82.6 ± 40.3
Cattle Egret *Bubulcus ibis*				1	26.9	23.4
Chestnut Teal *Anas castanea*	33	148.6 ± 71.3	65.1 ± 29.4	46	149.7 ± 71.1	80.1 ± 19.9
Dusky Moorhen *Gallinula tenebrosa*	1	16	14			
Eastern Great Egret *Ardea modesta*	5	121.2 ± 119.4	32.8 ± 18.5	16	86.1 ± 51.1	57.0 ± 29.4
Eurasian Coot *Fulica atra*	14	142.0 ± 74.8	74.3 ± 47.6	5	92.0 ± 29.2	72.8 ± 30.5
Glossy Ibis *Plegadis falcinellus*	1	114.4	22.9	1	68	45
Great Cormorant *Phalacrocorax carbo*	2	100.8 ± 0.7	23.5 ± 8.2	6	92.2 ± 26.0	74.0 ± 20.7
Grey Teal *Anas gracilis*	2	191.5 ± 153.4	61.6 ± 9.4	6	145.0 ± 97.4	82.8 ± 30.8
Hardhead *Aythya australis*	13	129.6 ± 62.0	64.6 ± 24.0	18	160.3 ± 93.1	87.2 ± 44.8
Hoary-headed Grebe *Poliocephalus poliocephalus*	1	47.3	29.9			
Intermediate Egret *Mesophoyx intermedia*	1	210	20	1	27	13
Little Black Cormorant *Phalacrocorax sulcirostris*	6	102.4 ± 66.5	38.8 ± 22.3	5	119.1 ± 106.7	57.3 ± 69.5
Little Egret *Egretta garzetta*				1	39	35
Little Pied Cormorant *Microcarbo melanoleucos*	19	106.6 ± 52.3	33.9 ± 14.9	44	97.7 ± 57.4	46.1 ± 28.8
Masked Lapwing *Vanellus miles*	6	189.8 ± 96.8	40.8 ± 24.2	6	142.0 ± 97.8	79.7 ± 18.0
Musk Duck *Biziura lobata*	3	101.5 ± 77.3	34.1 ± 19.1	7	107.1 ± 42.7	69.9 ± 28.8
Pacific Black Duck *Anas superciliosa*	20	159.2 ± 80.9	72.0 ± 40.2	17	189.6 ± 89.7	89.1 ± 31.4
Pied Cormorant *Phalacrocorax varius*	5	202.4 ± 150.8	50.0 ± 14.0	13	144.8 ± 113.7	77.9 ± 57.9
Pink-eared Duck *Malacorhynchus membranaceus*				12	96.2 ± 60.4	67.1 ± 27.2

Appendix 1 continued.

Species	Car			Walker		
	n	SD (m)	FID (m)	n	SD (m)	FID (m)
Plumed Whistling Duck *Dendrocygna eytoni*				1	178	130
Purple Swamphen *Porphyrio porphyria*	15	89.2 ± 29.8	43.2 ± 32.2	22	90.2 ± 47.1	57.9 ± 26.6
Red-necked Avocet *Recurvirostra novaehollandiae*				1	104.6	32.5
Red-necked Stint *Calidris ruficollis*	1	29.3	26.3			
Royal Spoonbill *Platalea regia*	2	56.0 ± 26.4	46.0 ± 34.9	9	73.7 ± 42.9	48.8 ± 27.9
Silver Gull *Larus novaehollandiae*	3	87.4 ± 19.0	17.4 ± 3.2			
Straw-necked Ibis *Threskiornis spinicollis*	22	222.6 ± 146.3	81.8 ± 38.3	12	164.4 ± 96.1	84.9 ± 40.0
White-faced Heron *Egretta novaehollandiae*	10	143.5 ± 84.5	63.0 ± 31.6	13	77.4 ± 31.4	46.4 ± 19.6
White-necked Heron *Egretta novaehollandiae*	1	116.7	26.4	2	71.2 ± 2.5	63.4 ± 7.6
Yellow-billed Spoonbill *Platalea flavipes*				1	38.4	24.7