Understanding the Western Port Environment 2018

A summary of research findings from the Western Port Environment Research Program 2011-2017 and priorities for future research

VICTORIA State Government

Melbourne Water
Enhancing Life and Liveability

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1 Executive Summary

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Western Port Science Review

Western Port has experienced substantial changes in the past 200 years, including extensive clearing of catchment and coastal vegetation, draining of large areas of swampland and progressive agricultural, industrial and urban development. Despite these changes, the marine and coastal environment continues to support unique and important wildlife that have been internationally recognized (e.g. UNESCO Biosphere Reserve, Ramsar convention listing for migratory waterbirds) and contains three of Victoria's 13 marine national parks. Western Port's environment is characterised by a variety of important habitats including mudflats, seagrass meadows, mangroves, saltmarshes and rocky reefs. These habitats are home to a diverse range of aquatic animals such as waterbirds, fish, marine invertebrates and mammals.

In the coming decades, further changes within the catchment and marine environment are expected to place more pressure on the health of Western Port – most notably urban development along the southeastern growth corridor of Melbourne and projected changes in climate (e.g. rainfall patterns, water temperatures and sea level rise). In light of these future pressures, the report, *Better Bays and Waterways: a water quality improvement plan for Port Phillip and Western Port* (Melbourne Water and EPA 2009) recommended the consolidation of knowledge of the Western Port environment. This information was to guide strategic management to protect and improve Western Port's marine and coastal environment. Prior to 2009, there had been relatively few environmental studies within Western Port, especially when compared to Port Phillip, with the last major study in the early-mid 1970s (Shapiro et al. 1975).

In response to *Better Bays and Waterways*, a major review of scientific knowledge about Western Port was released by Melbourne Water in 2011 – *Understanding the Western Port Environment: a summary of current knowledge and priorities for future research* (Keough et al. 2011) (the 'Western Port review'). The review was led by Melbourne Water with co-funding from the Department of Environment and Sustainability (DSE, now Department of Environment, Land, Water and Planning - DELWP) and Port Phillip and Westernport CMA. The research team consisted of 11 experts from across Australia, coordinated by Prof. Michael Keough of the University of Melbourne.

The Western Port review was assisted by an inter-agency advisory group, represented by nine government organisations: Melbourne Water, DSE, Port Phillip and Westernport CMA, Central Coastal Board, EPA Victoria, Parks Victoria, the Department of Primary Industries and the Department of Transport (now Department of Economic Development, Jobs, Transport and Resources - DEDJTR) and South East Water. Confirmation of environmental values and key management issues was achieved through workshops with the Western Port Catchment Committee, and in a public seminar to over 200 participants.

The importance of science to underpin strategic management of a major embayment and its catchments was clearly demonstrated by the Port Phillip Environmental Study (Harris et al. 1996). The study led to significant policy, planning and management activities – most notably the 1,000 tonne annual nitrogen load reduction target captured in the *State Environment Protection Policy (Waters of Victoria) – Schedule F6 Waters of Port Phillip Bay (1997)* and subsequent *Port Phillip Bay Environmental Management Plan (2001)*. In the same way, the Western Port Environment Research Program presented in this document has been developed to produce high quality science to guide management of Western Port.

The Western Port review was intended to be a resource for the Victorian Government and other stakeholders involved in natural resource management within Western Port and its catchment. The review highlighted the importance of Western Port and outlined measures to protect and improve its health into the future based on the following broad questions:

- What's important about the Western Port environment?
- What are the major threats to the health of Western Port?
- Do we know enough to protect it?

Western Port Environment Research Program

The Western Port review also documented and prioritised strategic research projects where there were substantial gaps in our understanding of ecosystem processes, species and threats (Chapter 15, Keough et al. 2011). This formed the basis for an interconnected research program that initially focused on high priority research tasks, planned for medium priority tasks and sought opportunities for initiating low priority tasks.

The Western Port review recommended a total of 43 strategic research needs that were screened against three criteria: management benefit, immediacy, and likelihood of a successful outcome. These were assigned a priority ranking of either '1' (highest priority), '2' (medium priority) or '3' (lowest priority). High priority knowledge gaps were major impediments to scientific understanding and management, and for which gaining the information was expected to enhance management immediately. Low and medium-priority knowledge gaps were considered less urgent or to have a lower chance of a successful outcome.

Of the 43 strategic research needs, 13 high priority research projects were identified and grouped in five themes: physical processes, nutrients and sediments, seagrasses, toxicants, and iconic species. They were directed at answering fundamental questions such as:

- How important are nutrients for protecting the health of the bay?
- What water quality conditions do seagrasses need, and how can this knowledge guide management targets and investment?
- How important is coastal erosion in terms of sediment loads to the bay?
- Are toxicants (such as heavy metals and pesticides) an issue for Western Port?
- How important are certain habitat types for fish diversity?
- Why has there been a decline in the numbers of fish-eating birds over the past 20 years?
- What species of seagrass occur in Western Port?

Since the release of the Western Port review, a substantial body of new research has been completed, based on the original research priorities. All high priority projects have now been completed or are well underway, and many medium priority projects have been initiated (Table 1.1). While the majority of these projects have been commissioned and funded by Melbourne Water (MW), there has been substantial co-funding or in-kind resources provided by other organisations: DELWP, DEDJTR, EPA Victoria, Parks Victoria (PV), Port Phillip and Westernport CMA, Central Coastal Board (CCB) and The Nature Conservancy (TNC). Participating research organisations include: Arthur Rylah Institute (ARI), CSIRO, Deakin University, Eco Insights, eCoast, Federation University, Hydronumerics, Monash University, Museum of Victoria, Phillip Island Nature Parks, Riverbend Ecological Services, Southern Cross University, the University of Melbourne, University of Tasmania and Victoria University.

Table 1.1 Summary of research projects completed or underway that were identified as strategic needs in the Western Port review (Keough et al. 2011).

Project	No.	Priority	Funded by	Led by	Status	Publications
			Physical processes			
Detailed and up-to-date bathymetry for Western Port	R.1	1	DELWP	DELWP	Complete	DELWP 2017
Calibrate hydrodynamic models for more accurate water movement	R.2	1	DELWP/ CMA/EPA	Hydronumerics	Complete	Hydronumerics (in prep)
Atmospheric inputs into Western Port	R.8	2	MW/EPA/ CSIRO	EPA/CSIRO	Complete	EPA/CSIRO 2013
Identify contribution of waves to sea-level changes in Western Port	R.9	2	MW/DELWP	Water Technology	Complete	Water Technology 2014
Determine the contribution of storm tide sea levels to waterway flooding (when accompanied by high rainfall)	R.10	2	MW/DELWP	Water Technology	Complete	Water Technology 2014, 2015
Incorporate shoreline erosion into climate change predictions	R.11	2	MW/DELWP	Water Technology	Complete	Water Technology 2013

Project	No.	Priority	Funded by	Led by	Status	Publications
Nutrients and sediments						
Measure residence time of sediments entering the bay	R.4	2	MW/CSIRO	CSIRO	Complete	Wilkinson et al. 2016
Contribution of coastal erosion to nutrient and sediment budgets	R.6	1	MW	CSIRO	Complete	Tomkins et al. 2014 Wilkinson et al. 2016
Develop a preliminary nitrogen and phosphorus budget	R.12	1	MW	Monash University	Complete	Evrard et al. 2013 Wilkinson et al. 2016
Measure nutrient cycling in major habitats	R.13	1	MW	Monash University	Complete	Evrard et al. 2013; Russell et al. 2016
Build a process-based biogeochemical model	R.14	2	MW	MW/ Hydronumerics	Complete for hydrodynamics and sediments	Yeates and Okely 2016
Seagrasses, mangroves and saltmarshes						
Assess the degree of nutrient and light limitation of seagrass, benthic microalgae, macroalgae and phytoplankton	R.15	1-3	MW/EPA/PV	Monash University	Underway (seagrass only)	Russell et al. 2016 Manassa et al. 2017
Determine water quality targets for sediments and nutrients that support seagrasses, benthic microalgae, reef algae, saltmarshes and mangroves	R.16	1-3	MW/EPA/PV	Monash University	Underway (seagrass only)	Holland et al. 2013
Confirmation of seagrass species using genetic markers	R.19	1	MW/DELWP/ CMA	Deakin University	Complete	Keough and Sherman unpublished
Estimate extent of invasion of key habitats	R.22	2	MW	Victoria University	Tall wheat grass in saltmarsh only	Hurst and Boon 2016
Characterise importance of saltmarshes and mangroves for biodiversity	R.24	3	Deakin	Deakin University	Complete for invertebrates in mangroves	Monk 2012
Use historical aerial photographs and ground-truthing to quantify historical and current distribution of mangroves and saltmarsh vegetation	R.25	2	MW/TNC	Deakin University	Underway	
Capacity for Zostera to recover and colonise new areas	R.26	1	MW/EPA/PV	Monash University	Underway	
Identify determinants of saltmarsh and mangrove recovery and seedling establishment	R.27	2	MW	Deakin University	Underway (mangroves only)	Hurst 2013; Hurst et al. 2015; Hurst et al. (in press)
Relationships between sea levels, sedimentation/erosion rates and vascular plant communities	R.29	2	MW/TNC	Deakin University	Underway	
Iconic species						
Determine linkages between fish and habitats	R.28	1	MW/DELWP/ CMA	Melbourne University	Complete	Jenkins et al. 2013; Jenkins et al. 2015
Investigate marine and estuarine requirements of the listed Australian grayling	R.32	3	MW	ARI	Underway	
Determine relative significance of shorebird and waterbird intertidal feeding areas	R.34	2	CCB	ARI	Complete	Hansen et al. 2011
Examine the trends of fish-eating birds in Western Port and Corner Inlet	R.35	1	MW/DELWP/ CMA	ARI	Complete	Menkhorst et al. 2015
Determine the effects of recreational fishing on fish stocks	R.39	1	MW/DEDJTR	Melbourne University/ DEDJTR	Complete	Jenkins and Conron 2015
Effects of sea level rise on shore birds	R.42	2	CCB	ARI	Complete	Hansen et al. 2011
Toxicants						
Initial estimate of risk from toxicants	R.36	1	MW/DELWP/ CMA	Melbourne University	Complete	Sharp et al. 2013
Impacts of toxicants on vegetation	R.37	2-3	MW	Melbourne University	Underway (mangroves and seagrass only)	Myers et al. (2015)
Investigate climate change and toxicant effects on fish	R.38	2	MW	Melbourne University	Underway (toxicants only)	Hassell et al. (2016)

Purpose of this document

The concerted research effort following the release of the Western Port review has significantly increased our knowledge about the Western Port environment, major threats and opportunities for management. Ongoing communication of the Western Port Environment Research Program has largely been through the Melbourne Water website (www.melbournewater.com.au). This site contains information about the research program, the Western Port review, summaries of projects, and research reports and presentations as they become available. Findings of the research program have also been communicated at dedicated public research seminars in March 2013 and February 2016.

The purpose of this document is to provide an integrated summary of research findings since 2011. The document is structured according to the following themes: sediment dynamics, seagrass and nutrients, hydrodynamic modelling, toxicants, mangroves and saltmarsh, fish and waterbirds. The final chapter also provides an updated plan for future research priorities excluding projects that have been completed or are underway, while new priorities arising from the research program have been added.

It is hoped that this document is a useful complementary resource to the Western Port review, one that not only raises awareness of Western Port's important and unique environment, but also encourages careful consideration of the research findings to protect and improve the bay into the future.

Summary of findings

Some of the notable findings arising from the Western Port Environment Research program include:

Sediments and Nutrients

- Resuspension of the sediment by tides and waves is the primary short-term driver of the light climate. Catchment sediment supply appears to have reduced in recent years, with an estimated mean-annual suspended solid delivery into Western Port of 23.8 kt y^{-1} (since 1980) (Chapter 2).
- Coastal banks near Lang Lang are eroding at around 30 cm each year, delivering approximately 4-8 kt y^{-1} of fine sediment into the bay. Erosion is predominantly occurring through the physical processes of abrasion and detachment of sediment from the bank surface during tidal cycles and wave attack (Chapter 2).
- There is a simulated net loss of fine sediments from the bay that exceeds the current estimated contribution from the shoreline erosion and catchment flow combined. Although there are significant deposition areas north of Corinella and the Rhyll Basin (Hancock et al. 2001), hydrodynamic modelling indicates there is a net flushing of fine sediments from the bay that is driven by residual clockwise currents (Chapter 4).
- Preliminary budget estimates suggest that catchment-derived nitrogen loads do not accumulate within the water column. This is likely to be largely associated with substantial exchange of water with Bass Strait during each tidal cycle (Chapter 4).

Toxicants

- Toxicants, such as heavy metals, hydrocarbons, pesticides and anti-foulants appear to be generally low across Western Port (Chapter 5).
- Some isolated areas, primarily estuaries to the north of the bay, were found to have elevated concentrations of herbicides and fungicides that pose a moderate ecological risk e.g. to seagrass and mangroves. Pesticides were primarily associated with agricultural areas and most frequently detected in freshwater and estuarine surface waters - tending to be restricted within 5 km of estuary mouths (Chapter 5).
- Storm events appear to increase the risk of exposure to pesticides, with increased rainfall being linked to increased pesticide occurrence and concentrations in the streams (Chapter 5).

Seagrass

- Genetic analysis of Western Port seagrass samples indicates there are only two *Zostera* species present: *Z. muelleri* in the intertidal-shallow subtidal areas, and *Z. nigricaulis* in shallow-deep subtidal areas. There is no molecular evidence for *Heterozostera tasmanica. Z. nigricaulis* occurs in deep and shallow sites and can show morphological differences at different depths that are likely to be environmentally driven (Chapter 3).

- Physical factors, in particular light availability, have a strong influence over seagrass cover and health. Resuspension of sediment by waves and currents is the most likely mechanism driving light limitation (Chapter 2).

- Seagrass (*Z. muelleri*) has some ability to cope with short-term increases in turbidity (up to five weeks) but persistent turbidity is likely to be detrimental to survival (Chapter 3).

- Improvement in the light climate of Western Port that would enable recolonisation and growth of seagrass across areas of the north and northeast (where seagrass was observed in the early 1970s) is likely to take at least 20 years. One way to improve the light climate may be by re-establishing seagrass coverage in less impacted areas, thereby stabilising the seabed and reducing resuspension (Chapter 4).

- While improved management of catchment loads and mitigating shoreline erosion are unlikely to have immediate benefits, they remain crucial elements to any long-term solution because they reduce further deposition and future mobilisation of fine material (Chapter 4).

- A key action to improving water quality to levels suitable for seagrass maintenance and restoration is to restrict sediment loads from the catchment and coastline to current levels of around 28 kt yr^{-1}. Suitable water quality for seagrass is then likely to occur once existing legacy sediments have been flushed out of the bay in the coming decades (Chapter 4).

Coastal Vegetation

- Historical aerial image analysis (over 58-70 years) of mangrove forests at three sites in Pioneer Bay - on the eastern side of Western Port - showed an overall increase in area and density. In this time, there has been little recolonisation of bare mudflats, potentially due to higher wave energy reducing the ability of propagules to recruit in those areas (Chapter 6).

- Mangrove planting along the Lang Lang coastline demonstrated that survival of seedlings in the first 12 months after planting could be substantially increased using a PVC guard, although additional protection measures are required as plants mature (Chapter 6).

- Larger mangrove seedlings grown in the nursery generally have higher survival rates when planted in the field. Seeds collected from the ground were found to germinate in the nursery in very high numbers, while seeds picked from trees had lower germination rates. Largest seedlings also grew from the largest seeds collected from the ground. It is recommended that seeds are collected in the middle of the season to optimise seedling growth and survival in time for pre-winter planting (Chapter 6).

- *Spartina anglica* (Spartina or common cord grass) has been recognised as a significant threat to intertidal habitats and recent mapping across Western Port has shown that the extent of Spartina has been significantly reduced following recent management efforts (Chapter 6).

- Field trials showed that a selective herbicide was not effective at controlling invasive tall wheat grass (*Lophopyrum ponticum*) while a broad-spectrum herbicide produced undesirable off-target effects. Alternative control options need to be explored such as manual removal, burning, biological control and grazing (Chapter 6).

Fish

- Some fish species known to use *Zostera* seagrass habitat can also use other habitats. However, seagrass is the most critical habitat for fish biodiversity in Western Port because of its extensive spatial cover and important role for larval settlement/development in shallow areas. *Zostera* also supports some unique species, in particular pipefish and seahorse species. Whilst *Zostera* in Western Port has declined since the 1970s, the cover of the oceanic seagrass, *Amphibolis antarctica*, has remained relatively stable in the Western Entrance over the same period (Chapter 7).
- The Rhyll Segment is an area of high catch rates for most fish species and is strongly influenced by water quality and sedimentation entering the northeast of the bay – primarily from the catchment and coastal erosion. Catchment management, aimed at maintaining water quality entering the bay, is therefore likely to be critical to maintaining fish biodiversity and sustaining recreational fishing in Western Port (Chapter 7).
- Fisheries data from 1998-2013 indicates that overall, King George Whiting stocks in Western Port appear to be improving, stocks of Snapper and Flathead are considered stable, and Gummy Shark stocks in good condition. On the other hand, it appears that stocks of Elephant Fish have declined significantly since 2004 (down by 75%), with catch rates falling from 0.21 fish per angler hour to 0.052 in 2013/14 (Chapter 7).
- An analysis of long term trends in Snapper, King George Whiting and Elephant Fish populations and environmental conditions, suggests that changes in population abundances are predominantly associated with El Niño and La Niña events (and associated changes in rainfall and air temperature) and, to a lesser extent, recruitment pulses and cessation of commercial netting (Chapter 7).
- On a local scale, nitrogen loads and planktonic algae concentrations affected fish abundance through the food web and via seagrass cover which provides essential habitat for juveniles. On a regional scale sea surface temperature in Bass Strait was important, especially in affecting catches of Snapper and King George Whiting (Chapter 7).

Waterbirds

- Population trends were determined for 39 of the 85 observed waterbird species (seabirds not included). Populations of 22 waterbird species in Western Port declined between 1973-2015, 15 species remained stable (despite fluctuations and some changes in distribution) and two of the 39 species have increased. A further 46 species were recorded in numbers that were too low or variable for useful analysis. The main declines were associated with trans-equatorial migratory shorebirds (nine species) and this may be due to habitat loss in the Yellow Sea, east Asia. Declines were also observed in some fish-eating birds (Chapter 8).
- Several species declined in the central to eastern part of the bay along with major loss of seagrass. That area has now been colonised by four waterbird species that were formerly rare in the bay, suggesting a local switch to a new type of habitat (Chapter 8).
- Fish-eating terns, cormorants and pelicans have decreased in Western Port and increased in West Corner Inlet. Little Pied Cormorant decreased in the late 1970s and early 1980s, in association with seagrass dieback in Western Port. Crested Tern decreased later in the 1980s and 1990s, to a greater extent than other fish-eating species despite establishing a large new breeding colony at the bay entrance on the Nobbies (Phillip Island). Numbers of two smaller and less numerous tern species (Fairy Tern and Little Tern) declined at the same time (Chapter 8).
- Crested Terns, Little Terns and Fairy terns have made less use of the bay since a decline in small fish and a larger predatory fish (i.e. Australian Salmon that drive small fish to surface waters where terns feed). The decline of Fairy Terns is of particular conservation concern (Chapter 8).
- Black Swans form 69% of the waterbird biomass in the survey area, and may be useful as highly visible indicators of seagrass abundance (Chapter 8).

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2 Sediment supply, seagrass interactions and remote sensing

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Research Priorities

This work addresses research priorities identified in the Western Port review (Keough et al. 2011) under the research theme, *Developing a Complete Sediment Transport Model*.

- Measure residence time of sediments entering the bay (research priority 4)
- Refine understanding of effects of seagrass on sediment transport (research priority 5)
- Estimate the contribution of coastal erosion to nutrient and sediment budgets (research priority 6)

These priorities are linked to research themes *Improving Hydrodynamic Models of Western Port* and identification of *Sediment and Nutrient Thresholds for Important Plants*.

Key Findings

- The primary sediment inputs into Western Port (measured as Total Suspended Solids, TSS) are rivers (stream bank and gully erosion) and coastal bank erosion. Some sediment is redistributed, mostly from the Upper North Arm, in a clockwise direction around French Island.
- Catchment sediment supply appears to be below a historical peak but is no longer declining.
- Coastal banks near Lang Lang are eroding at 30 cm y^{-1} (Tomkins et al. 2014, Wilkinson et al. 2016), delivering 4-8 kt y^{-1} of fine sediment to Western Port. It is likely that decline in seagrass area has increased the bank erosion rates.
- Mean-annual delivery into Western Port since 1980 (based on scaling of catchment areas and erosion patterns) is (TSS) 23.8 kt y^{-1}, (TN) 729 t y^{-1} and (TP) 69.8 t y^{-1}.
- Resuspension of sediment by tides and waves is the primary short-term driver of the light climate within Western Port.
- The light climate affects seagrass condition and extent by modifying growth rate and mortality. Sediment also impacts seagrass growth and mortality through smothering, and benthic aggradation that affects tidal exposure and temperature.
- Analysis of river sediment and nutrient concentrations suggests that changes in agricultural practices and urban development since the 1990s seem to be impacting catchment sources, which can inform further investigation and management responses (Wilkinson et al. 2016).
- The contributions of each catchment to the total river TSS load over this period were Cardinia 12%, Bunyip 31%, Lang Lang 41% and Bass 16%, and bay turbidity is affected by decadal variations in load.
- Remote sensing and river load monitoring can help to inform and evaluate management success and assess condition.
- Managing sediment supply at or below current levels may help improve water clarity in coming decades.

Links between seagrass and sediment inputs

Western Port experienced extensive loss of seagrass coverage between the 1970s and 2000 (Shepherd et al. 2009). Aerial photographs indicate that the greatest losses occurred in the northern section of the bay in the period up to 1979 (Marsden et al. 1979). Today much of the area north and east of French Island (the Upper North Arm and Corinella segments) are chronically turbid and the substrate is non-vegetated. The causes of seagrass loss have not been conclusively identified, although the sensitivity of seagrass growth to light limitation and sediment smothering is well known (Collier et al. 2012). Recent modelling also indicates that the concentration of suspended sediment within Western Port is likely to be a key driver of seagrass distribution (Holland et al. 2013). Sediments mobilised by the channelisation of rivers through the Koo Wee Rup swamp prior to the 1950s have been suggested as a possible primary cause (Wilk et al. 1979; Roberts, 1985), and sediment source tracing confirms that most of the fine sediment delivered from river catchments is derived from river channel banks and gully erosion (Wallbrink et al. 2003). River channelization would have also increased the delivery of fresh water.

The primary linkages between seagrass in Western Port, the water quality, sediment inputs and their management can be described in a conceptual model, to help identify the important processes and their time-scales (Figure 2.1). The major sediment inputs are rivers and coastal bank erosion. River inputs are mainly derived from stream bank and gully erosion (Wallbrink et al. 2003), but also surface runoff. River inputs are mostly controlled by stream bank vegetation and runoff controls such as stormwater treatment. Most of the sediment input enters Western Port north of French Island. In that area, resuspension by tides and waves, of sediments previously supplied, is the primary driver of turbidity on a daily basis (Hancock et al. 2001). Some re-suspended sediment is redistributed, mostly in a clockwise direction around French Island to the Corinella and Rhyll segments of Western Port. River plumes can occasionally also have a direct impact on the light climate in some areas, and nutrient inputs can impact on the light climate experienced by seagrass by growth of algae on their stems (epiphytes). The light climate affects seagrass condition and extent by modifying growth rate and mortality. Sediment also impacts seagrass growth and mortality through smothering, and benthic aggradation (build-up of sediments) that affects tidal exposure and temperature.

Recent research into the processes, magnitudes and dynamics linking sediment inputs and seagrass has focused on: (i) estimating time-series of terrestrial sediment and nutrient inputs from river loads and coastal bank erosion in recent decades, (ii) assessing historical spatial and temporal water clarity and seagrass extent using remote sensing imagery, and (iii) modelling the relative effects of water clarity and other conditions on seagrass.

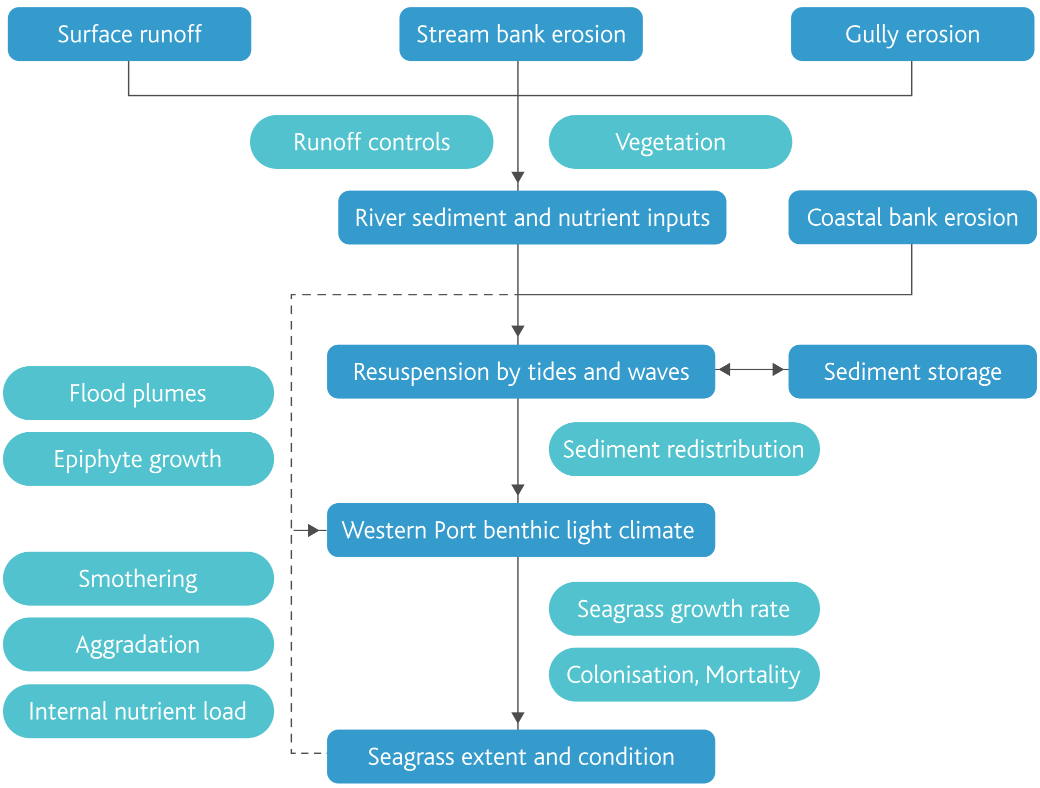

Figure 2.1 Conceptual model of the primary linkages between sediment and nutrient inputs to Western Port and the extent and condition of seagrass.

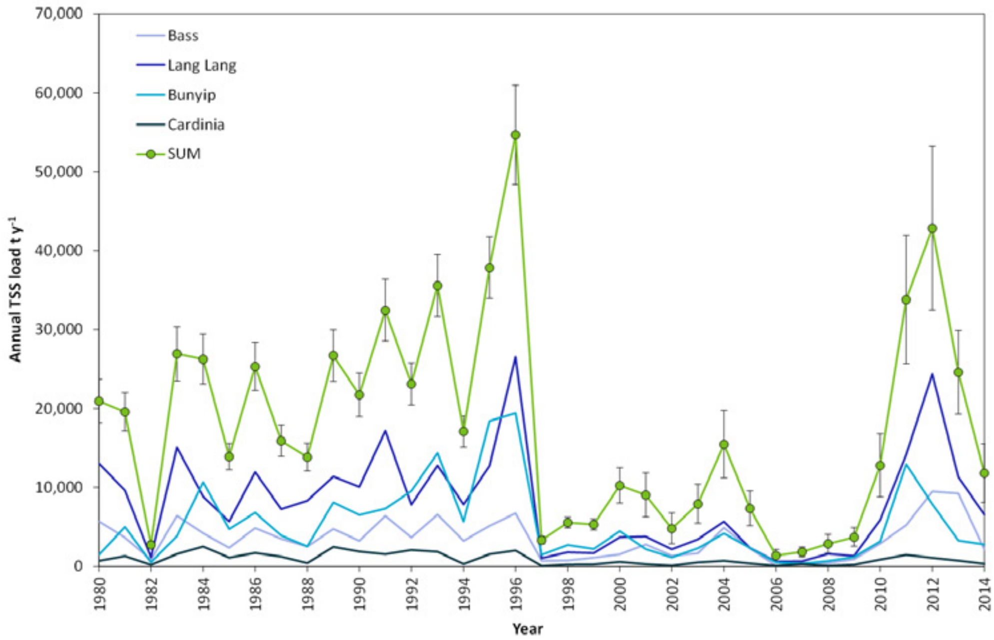

Figure 2.2 Annual river station TSS loads since 1980. Instantaneous concentration was estimated by turbidity regressions during the period of turbidity monitoring (2001–2014), and by discharge regressions in earlier years (Wilkinson et al. 2016).

The recent research is summarised below, in the context of the conceptual model and previous studies. We also outline opportunities for more focused studies on individual components of the conceptual model, which can help to define objectives and pathways to benefit seagrass, and to better inform the management of Western Port water quality.

Sediment inputs from rivers

Four contributing catchments (from west to east: Cardinia Creek, Bunyip River, Lang Lang River and Bass River) cover more than two thirds of the total area draining to Western Port, and discharge to the areas most affected by seagrass loss. Daily time-series of historical total suspended solids (TSS), total nitrogen (TN) and total phosphorus (TP) river loads in these rivers are now available to inform analysis of impacts on bay turbidity and as a baseline to evaluate catchment management outcomes. These new load estimates were based on regressions fitted between sampled concentration and continuous turbidity monitoring at river gauges during the Melbourne Water load monitoring program (2001-2014; Wilkinson et al. 2016). To estimate loads prior to 2001 we also used concentration regression curves against discharge based on monthly water sampling from the 1990s. The mean-annual TSS load summed across the four gauges was estimated at 17.7 kt yr^{-1} over the period 1980-2014, or 12.9 kt yr^{-1} over the period 2001-2014 (Wilkinson et al. 2016). Annual loads during the millennium drought (1997-2009) were generally much smaller than those in prior or subsequent years, with 1982 being the only other year since 1980 with an annual load as small as those in the millennium drought years (Figure 2.2). In contrast, the loads in 2011 and 2012 were some of the largest since 1980. Scaling of these gauged loads, based on catchment areas and erosion patterns downstream of the gauges, indicates the mean-annual delivery into Western Port at the mouths of these streams (as opposed to where the flow gauges are located) since 1980 at (TSS) 23.8 kt y^{-1}, (TN) 729 t y^{-1} and (TP) 69.8 t y^{-1}. The contributions of each catchment to the total river TSS load over this period were Cardinia 12%, Bunyip 31%, Lang Lang 41% and Bass 16%.

The above mean-annual TSS load we estimated (from 1980) contrasts with higher estimates from fine sediment aggradation rates in Western Port prior to 1950 of 70-100 kt y^{-1} (Hancock et al. 2001), and with the earliest estimated river inputs for the period 1973 -1976 of 65 kt yr^{-1} (Sargeant et al. 1977, Dale and Pooley 1979, Wilkinson et al. 2016). Sediment aggradation is today occurring primarily in the Corinella segment of the bay, but the sediment is mostly derived from the Upper North Arm of the bay (north of French Island) from where it is redistributed by tidal currents (Hancock et al. 2001).

A decline in catchment loads since the period 1950-1970 and earlier is consistent with observed stabilisation of the river channels since the 1970s. The rivers draining to the north of Western Port were progressively channelized from the 1850s, concluding with major works after very large floods in the 1930s (Roberts 1985). Substantial down-cutting of the river beds was initiated by this channelization due to the increased stream gradients. Catchment clearing and urbanisation also increased event runoff and channel instability. A range of stabilisation works including drop structures, sheet piling weirs and rock lining, were carried out from the 1960s-1990s (Sargeant 1977, King and Kay 1980, Brizga et al. 2001).

These works remain evident today, so despite their large depth the river channels are apparently more stable today than they were in the period prior to 1970. However, the relationship between sediment inputs, coastal water quality and seagrass loss are not direct, and considerable lags and feedbacks can confound understanding causal mechanisms.

Stream bank and gully erosion are understood to remain the largest sediment sources. Analysis of river sediment and nutrient concentrations also suggests that changes in agricultural practices and urban development since the 1990s seem to be impacting catchment sources. This information can inform further investigation and appropriate management (Wilkinson et al. 2016). For example, low-flow fine sediment concentrations have increased in Bunyip River catchment since the 1990s, possibly associated with new urban development. Changes in nutrient concentrations suggest increased livestock numbers or fertiliser application in some catchments, and improved agricultural runoff controls in other catchments.

Sediment inputs from coastal erosion

Another significant and ongoing source of fine sediment to Western Port is erosion of clay-rich coastal banks, which extend for about 10 km near Lang Lang and are exposed to the predominant westerly winds. Today the coastal banks are eroding at 30 cm y^{-1} (Tomkins et al. 2014; Wilkinson et al. 2016), delivering 4-8 kt y^{-1} of fine sediment to Western Port. This represents approximately 30% of the terrestrial fine sediment input to the Upper North Arm and Corinella segments of the bay, an estimate supported by sediment source tracing (Wallbrink et al. 2003).

We now know that coastal bank erosion is reasonably consistent at monthly timescales, tending to increase slightly in warmer months apparently due to enhanced inter-tidal drying of the bank face, rather than being episodic in storms as was earlier thought (Tomkins et al. 2014). It is possible that the magnitude of bank wetting and drying may have been enhanced by removal of natural vegetation and constructing drains. On steeper headland sections of the coastal banks the bank toe is scoured by waves, leading to mass failure of the upper bank face (Wilkinson et al. 2016). Seagrass can moderate wave energy and height (Pinsky et al. 2013), and so it is likely that decline in seagrass area has increased the bank erosion rates. Mangroves did not occur historically on this area of coastline. On some high tides, waves overtop the bank for tens of metres inland (Figure 2.3), constrained by the constructed embankments. Engineered erosion control solutions can be successful at controlling coastal erosion but are expensive (e.g., wave dissipation or hard walling). Sea level is likely to rise 44-74 cm by 2100 depending on emissions scenarios (Church et al. 2013), or more if Antarctic ice sheets destabilise. It is expected that sea level rise will exacerbate coastal erosion as coastal vegetation becomes degraded. Bank erosion control measures will need to be adaptable to continued sea level rise.

Figure 2.3 Coastal bank erosion monitoring site during high tide, 9 October 2012 (Tomkins et al., 2014), and during low tide on February 8, 2016 (Wilkinson et al. 2016).

Remote sensing of bay water quality and vegetation

Remote sensing of non-algal particulate concentrations (an estimate of suspended sediments) since the 1980s indicates that the bed height complexity and shallow depth of the bay makes it sensitive to resuspension by tides and waves at sub-daily to annual time-scales. Concentrations are highly variable across Western Port at points in time and at individual sites over time (Wilkinson et al., 2016). Remote sensing accounts for the spatial and temporal patterns much more comprehensively than can be achieved by field monitoring, and can be highly complementary to monitoring. This analysis shows that tidal and wave resuspension are the main drivers of the turbidity north of French Island under most conditions, and that resuspension can affect water quality all around French Island to varying degrees. The remote sensing analysis also indicates that particulate concentrations in the bay were low more frequently during the prolonged Millennium drought when river sediment and nutrient inputs were small. This suggests that the turbidity of Western Port is elevated by fresh sediment and nutrient inputs over subsequent months to years, and so it can be assumed that turbidity might decline over several years in response to sustained reductions in sediment inputs.

Figure 2.4 CA Landsat 8 image acquired on 25 April 2016 at very low tide (reproduced from Wilkinson et al. 2016). Significant macrophyte coverage is exposed on the intertidal flats and substrate visibility is possible in most the bay.

Such behaviour is consistent with the redistribution of fine sediment from the Upper North Arm to the Corinella segment - indicated by aggradation rates and substrate particle size (Hancock et al. 2001). The sediment redistribution within Western Port is partly a natural process, which has built the substantial inter-tidal and sub-tidal mudflats and seagrass beds to the south and west of French Island. However, the magnitudes of resuspension and redistribution have been elevated by historical and ongoing human activities.

Seagrass and macroalgae extent have varied historically across Western Port due to the complex bathymetry. Several field surveys have documented change at different resolutions. As for water quality, remote sensing offers a way to more frequently and consistently map the substrate over time, adding value to field data. Recent analysis of ten Landsat images from 1973 to 2014 indicates that the combined extent of seagrass and macro-algae declined from 1973 to 1979, then increased in the period to 1998 and then declined since, albeit remaining above the mid-1970s levels (Wilkinson et al. 2016). This is generally consistent with mapping and field surveys (Blake and Ball, 2001). However, only submerged macrophyte areas were estimated and differences in tide stage between images may have affected the estimated changes in area. Seagrass could not be distinguished from macroalgae in models of the historical Landsat imagery due to the radiometric and spectral resolution of the sensors, and seagrass density was not assessed. However, the current Landsat and Sentinel sensors do enable seagrass to be separated from macroalgae, and can better predict density as well as cover. Remote sensing enables a deeper understanding of the temporal dynamics of sediment transport and seagrass extent within Western Port, and modern sensors show great promise as a monitoring tool and to evaluate management outcomes (Figure 2.4).

Modelling the controls on seagrass

An earlier Melbourne Water hydrodynamic model of Western Port contained a simple seagrass model but did not incorporate several important aspects of seagrass dynamics including the interplay between above ground and root biomass. A more detailed stand-alone seagrass model has now been developed which simulates the impact of water quality on seagrass growth and density under long-term scenarios (Wilkinson et al. 2016). Variables in the model included light, temperature, salinity and nutrient limitation. The new model is based on bed height information for Western Port and is driven by meteorological data and includes light absorption coefficients (to discriminate between different types of vegetation and bare areas) on a spatial grid across Western Port (derived from satellite imagery).

The simulated extent of seagrass beds was found to be similar to that measured. Scenario modelling confirmed that seagrass extent is strongly correlated with light availability in Western Port. Thus, turbidity caused by sediment loads and variation in water depth plays a major role in seagrass decline or growth.

Further, scenario modelling over 100 years indicates that one metre of sea level rise and/or an increase in water temperature may cause a substantial reduction in seagrass extent within existing seagrass beds. This and other changes in seagrass distribution will need to be taken into consideration in future seagrass restoration strategies (Wilkinson et al. 2016).

Future directions and opportunities

The recent research has developed our understanding of the links between sediment inputs to the bay and seagrass (Figure 2.1). However, the effect of specific catchment management actions on Western Port water quality and seagrass extent are less-well understood. Initial modelling indicates that, once the legacy fine sediments have been flushed from the bay over the next two decades, maintaining current catchment loads of around 28 kt/yr in the face of future urban growth is likely to support comparable seagrass cover to the early 1970s, but this requires further validation (Chapter 4). The likely directions, magnitudes and time-scales in which water quality and seagrass extent may change in future decades are also uncertain. Opportunities and priorities for further investigation are identified based on the components of the conceptual model (Figure 2.1) that are less-well known. Research priorities are listed below. They are grouped into receiving waters, sediment inputs, and source management.

2.1 Receiving water quality and seagrass monitoring, modelling and targets in Western Port

i. The Landsat based predictions of particulate concentrations, water clarity and seagrass/macro-algae extent can be used to further improve the accuracy of the hydrodynamic models of Western Port and to monitor future changes in seagrass cover.

ii. Field measurements of the spectral characteristics of Western Port would improve remote sensing analysis of seagrass extent and particulate concentrations, and digital data from additional historical seagrass surveys will improve validation of remote sensing.

iii. Developing a more detailed historical archive of water quality from remote sensing would assist further investigation of the effect on particulate concentrations of wind and tidal resuspension relative to river inputs.

iv. Analysis of data from new satellite sensors, including Landsat 8 and the Sentinel series, can be investigated for improved future monitoring of seagrass and macro-algae extent and density.

v. Particle size variations in turbid parts of Western Port could be modelled from remote sensing imagery to help distinguish between new sediment inputs and sediment resuspension by tidal currents and wind-induced waves.

vi. The improved seagrass model developed by CSIRO can be integrated with the Melbourne Water ELCOM/CAEDYM hydrodynamic-biogeochemical coupled model.

vii. Modelling the colonization of new areas by seagrass, especially under sea level rise, can give new insights into adaptation capability of seagrass under future climate change scenarios.

viii. Assimilating remote sensing data into the seagrass model would improve predictions.

ix. The effect of river loading on seagrass shading events can be tested by simulating river plume development.

x. Sediment redistribution can be simulated to assess the timescales over which sediment stores in the Upper North Arm of Western Port may be depleted under sediment input scenarios.

2.2 River load monitoring, modelling and targets for fine sediment and nutrients

i. Priorities for erosion management, and evaluating the effect of changes in management, would be informed by implementing a catchment model such as Dynamic SedNet that represents the primary land use sources of sediment and nutrients.

ii. Renewing the monitoring of river sediment and nutrient concentrations would help to inform management priorities, to evaluate their effects, and to constrain modelling of catchment sources; turbidity sensors have been demonstrated to improve load estimates.

iii. Further analysis of historical river fine sediment and nutrient concentration data could be undertaken to better define and attribute the timing and magnitudes of change.

2.3 Identifying priorities and goals for targeted management of sediment (and nutrient) sources through actions such as streamside revegetation and livestock control, stream bed stabilisation, urban stormwater treatment and rural runoff management, including:

i. Mapping the extent and severity of stream bank and gully erosion throughout catchments (e.g., using LiDAR imagery), and assessing the local effectiveness of stream bank vegetation at mitigating erosion.

ii. Investigations and trials to identify suitable options for control of coastal bank erosion.

iii. Quantifying the contributions of urban development relative to runoff from existing urban and agricultural areas.

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3 Seagrass – nutrients, light and genetics

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Tim Smith and Craig Sherman, Deakin University

Photo: Parks Victoria

Research Priorities

This project addresses the following research priorities identified in the Western Port review (Keough et al. 2011):

A nutrient budget for Western Port

- Measure nutrient transformation in major habitats (research priority 13)

Sediment and nutrient thresholds for important plants

- Assess the degree of nutrient and light limitation of major primary producers (research priority 15)
- Determine water quality targets for sediments and nutrients that support seagrasses, microphytobenthos, reef algae, saltmarshes and mangroves (research priority 16)

Resolve seagrass identities

- Determine which *Zostera* seagrasses are present in Western Port (Research priority 19)

Key Findings

- Current nutrient loads to Western Port are not posing a significant risk to seagrass cover on a bay-wide scale, although there may be some localised problem areas.

- Physical factors, especially light availability, exert a strong influence over seagrass cover and health.

- At some sites, seagrass is growing in (suboptimal) light-limited conditions for most of the year.

- Although we have demonstrated that seagrass has some ability to cope with short-term (i.e. up to five weeks) increases in turbidity, persistent turbidity is likely to be detrimental to survival e.g. exhausted carbohydrate stores.

- Genetic analysis shows there are two common species of seagrass in Western Port – *Zostera muelleri* (intertidal -shallow subtidal) and *Z. nigricaulis* (shallow-deep subtidal).

- Analysis of genotypic diversity in *Z. muelleri* shows there is a high level of gene flow and connectivity between sites with the exception of the northeast of the bay, with possible implications for seagrass resilience and population persistence in this region.

- Seagrass beds provide an important source of nitrogen through nitrogen fixation in their root zone.

- Overall, seagrass meadows were found to contribute 320 t of nitrogen each year to the Western Port ecosystem compared to approximately 110 t a year in unvegetated soft sediments. Nitrogen fixation by seagrass contributed ~40% of the nitrogen inputs compared to the catchments (small rivers and streams (~50-60%) and atmospheric deposition (rainfall (<10%), confirming the importance of seagrass habitats as a key component to nutrient cycling in Western Port.

- Seagrasses throughout Victorian estuaries become overgrown with epiphytes when nitrogen inputs exceed 10 t km^{-2} of estuarine area (Woodland et al 2015). By comparison, the rates of nitrogen input to Western Port are less than 1 t km^{-2}, suggesting that, with the possible exception of some sites adjacent to agricultural drains, nitrogen inputs are not a major issue in Western Port.

Background

Seagrass meadows are a critical habitat for a broad range of aquatic life within Western Port and play a vital role in coastal and estuarine ecosystem functions including: regulation of nutrients, stabilisation of sediments, nursery grounds for recreationally/commercially important fisheries, and as an essential food source for a range of marine animals (Collier et al. 2012, Connolly 2009). Like many coastal habitats throughout the world, seagrasses are subject to multiple stressors (e.g. environmental, biological and climatological) and understanding their survival capabilities in a range of environments can assist with their conservation and management.

In Western Port, the dominant intertidal species is *Zostera muelleri* (previously referred to as *Z. capricornii*) (Kuo 2005). In the mid-1970s to early 1980s extensive loss (up to 75%) of intertidal seagrasses was observed, with increased sediment inputs from human activities and associated changes in water quality a likely, but unconfirmed, cause. Seagrass recovery has been limited in many areas of the bay, with sites of poor water quality still showing minimal signs of recovery. As such, the need for research-based approaches was recognized to not only prevent further declines, but also assist in recovery (Keough et al. 2011). Seagrasses rely on a stable environment and fluctuations in nutrients, light, substrate suitability, temperature and dissolved CO_2 can severely impact primary production and cause decline in coverage (Connolly 2009). As such, studies which examine the relationship between environmental conditions in Western Port and the cover of seagrasses are essential.

The Western Port review (Keough et al. 2011) identified a number of research priorities relating to seagrass habitats. This chapter will summarise research undertaken over the last five years on these priority areas, highlighting key management actions and mitigation strategies, along with future research priorities.

Measuring nutrient transformation in major habitats

Productivity of coastal waters is typically limited by a few key nutrients, most commonly nitrogen. Excess influx of nitrogen from land activities can lead to toxic algal blooms or expansive algal growth that can smother seagrasses (Larkum et al 2006). Subtidal and intertidal sediments are important sites for nutrient recycling in coastal systems. In Port Phillip, the sediments are known to be important sites for nitrogen cycling, in particular nitrogen removal through the process of denitrification. The intertidal areas of Western Port represent a major habitat type, yet there is limited data on nutrient transformation in this system. To address this research priority two approaches were used. The first was to measure the exchange of nutrients between the sediment and the water column directly using sediment cores (Figure 3.1) and the second was to take boat-based water samples from a channel over a 24-hour tidal cycle, which were then used to calculate nutrient exchange rates (Evrard et al. 2013). Both approaches gave broadly consistent results and indicated that nutrient exchange between the tidal flats and the water column in Western Port was very low compared to catchment inputs. For example, the maximum measured input of nitrogen scaled to the area of the tidal flats in the northern section of Western Port gave an input of around 1 t y^{-1}, compared to riverine inputs of ~650 t y^{-1}. The most likely reason for this low rate of nutrient release is uptake by microphytobenthos (microalgae that live on the sediment surface) and seagrass.

Such low-nutrient-ecosystems remain productive because of their ability to tightly recycle and produce their own nutrients. In the case of nitrogen, the atmosphere provides a huge reservoir that is not bioavailable and must be converted to a form that can be used by organisms via a process known as nitrogen fixation. Nitrogen fixation adds nitrogen to the system in a form that is taken up by seagrass and fish through fixation in the root zone followed by uptake into the seagrass, which is then consumed. This contrasts with catchment inputs which lead to higher concentrations of bioavailable nitrogen in the water column that feed algal growth.

Seagrasses are known to be important factories for nitrogen fixation, yet we know little about how important the beds in Western Port are, and how they compare to areas where seagrass has been lost. To answer this, nitrogen fixation – and the opposite process of denitrification (the removal of bioavailable nitrogen) – was measured in unvegetated soft sediments and seagrass habitats within Western Port using core incubations (Figure 3.1). Overall, seagrass meadows were found to contribute 320 t of nitrogen each year to the Western Port ecosystem compared to approximately 110 t a year in unvegetated soft sediments. Nitrogen fixation by seagrass contributed ~40% of the nitrogen inputs compared to the catchments (small rivers and streams (~50-60%) and atmospheric deposition (rainfall (<10%), confirming the importance of seagrass habitats as a key component to nutrient cycling in Western Port.

To investigate how nitrogen fixation and denitrification in sediments respond to nutrient inputs (nitrogen), two sites within Western Port were examined - one site close to inputs (Corinella) and one further away (Rhyll). Interestingly, at the site closer to the inputs, nitrogen fixation rates were at times reduced, often becoming a net sink through denitrification. In contrast, at times of high nitrogen limitation, sediments were able to increase their rates of nitrogen fixation, highlighting the functional importance of seagrass habitats within Western Port.

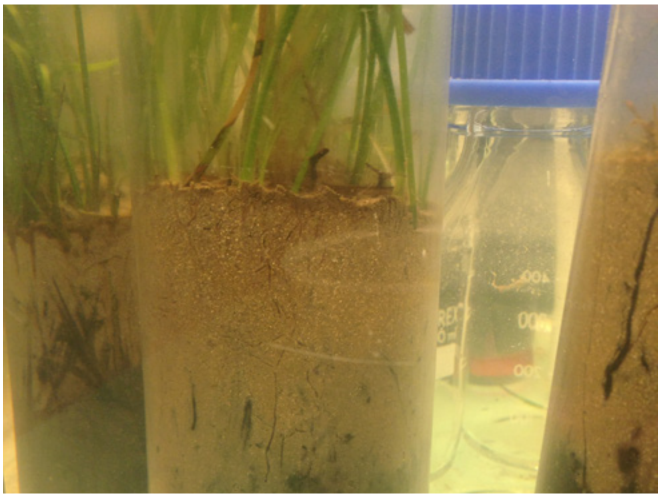

Figure 3.1 Sediment core set up used to measure rates of nutrient exchange, denitrification and nitrogen fixation in Western Port.

Assessing the degree of nutrient and light limitation of the major primary producers

Light availability is critical for seagrass health, with loss of water clarity likely to cause a decline in health and cover (Bjork et al. 1999, Ralph et al. 2007). Minimum light requirements for most seagrass species is within the range of 2-37% of surface irradiance (Lee et al. 2007). Seagrasses that grow in the intertidal zone are often subjected to oversaturating irradiances - which can also cause decline through thermal stress, desiccation and photo-inhibition (Petrou et al. 2013). As such, seagrass survival in the intertidal zone depends on a species' ability to acclimate to site-specific conditions (Silva and Santos 2003). By understanding light thresholds, management decisions directed towards maintaining or restoring these critical light levels are likely to be more effective (York et al. 2013).

The impact of light availability on intertidal *Z. muelleri* physiology and morphology was investigated at two sites - Coronet Bay and Crib Point. Daily average light regimes (over a 12-month period) at Coronet Bay were both lower and higher (3 mol m^{-2} d^{-1} to 38.7 mol m^{-2} d^{-1}) than Crib Point (6 mol m^{-2} d^{-1} to 24.8 mol m^{-2} d^{-1}). When compared to low light thresholds developed for tropical and sub-tropical *Z. muelleri* (2-6 mol m^{-2} d^{-1} and 1.7-7.2 mol m^{-2} d^{-1} respectively), the lower limits fall within these thresholds. When light was examined against depth (Figure 3.2), light penetration during inundation was significantly lower at Coronet Bay, suggesting that it may be more turbid (caused by either re-suspension of bed sediments or sediment plumes from adjacent freshwater discharges). Whilst the low light thresholds have been developed for tropical/sub-tropical species and may not be relevant to temperate species, these results suggest that seagrasses at both sites may be growing in light limited conditions for a large proportion of the year.

To examine the effect of varying light regimes and exposure on seagrass physiology and morphology, the relationship between light during inundation (high tide) and exposure (low tide) was examined along a vertical gradient from the high to low intertidal. Results were consistent with seagrasses having optimised their photosynthetic capacity, with physiological acclimations being site specific. Longer-term morphological changes were also noted (e.g. increase in leaf length within the lower intertidal and a decrease in leaf width in the higher intertidal zone), suggesting a dissimilar light history between sites for an extensive period. The impact of light and tidal exposure on seagrasses at Coronet Bay suggests that they may already be living at their minimum/maximum light thresholds, with any further changes in light likely to cause significant decline as seen in the northeast of the bay.

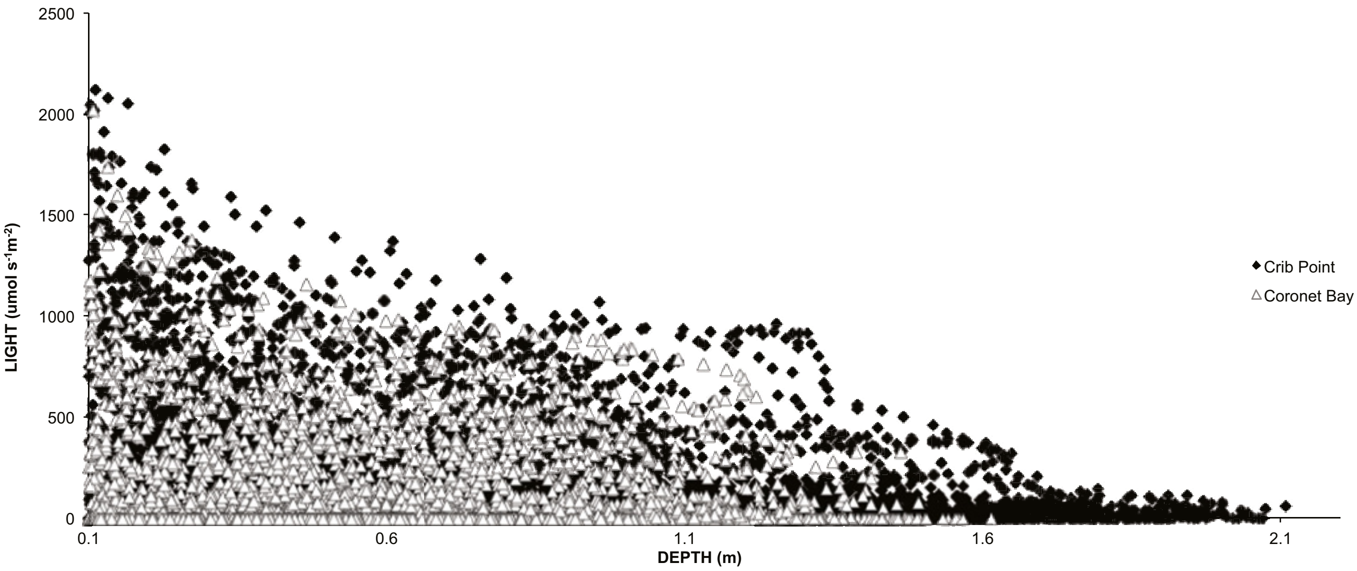

Figure 3.2 Light as a function of depth at Crib Point versus Coronet Bay – February to May 2015.

Similarly, if light levels increase at Crib Point during exposure (i.e. low tide), photosynthetic efficiency may decrease, and/or if light levels decrease during inundation, current exposure levels may cause a reduction in photosynthetic performance. This study highlights the need for site-specific studies to inform management strategies within Western Port, and demonstrates the capacity of temperate *Z. muelleri* to tolerate a range of light conditions.

As previously noted, seagrasses have evolved to grow in relatively nutrient-poor waters and are therefore adapted to efficiently take up nutrients from diverse sources including through nitrogen fixation. Nitrogen is therefore unlikely to be limiting to seagrass growth in Western Port. Ratios of carbon (C), nitrogen (N) and phosphorus (P) are commonly used as indicators of nutrient limitation in seagrass. In general, it is thought that a C:N ratio greater than 18 in seagrass leaves indicates nitrogen limitation. Consistent with this, we observed higher rates of nitrogen fixation when seagrass leaves had a C:N greater than 18 (Figure 3.3), suggesting seagrass can stimulate nitrogen fixation in the root zone as required (Russell et al. 2016). Phosphorus is another nutrient that could limit seagrass productivity, and it has been shown that seagrass N:P ratios can exceed 90 in phosphorus limited systems (Fraser et al. 2012). In Western Port the average seagrass N:P ratio was found to be 29 in August and 24 in February, and rarely exceeded 30 (Holland et al. 2013). This strongly suggests the system is not strongly P limited. From this it can be concluded that physical factors, especially light, are likely to exert the overwhelming control over seagrass productivity in Western Port.

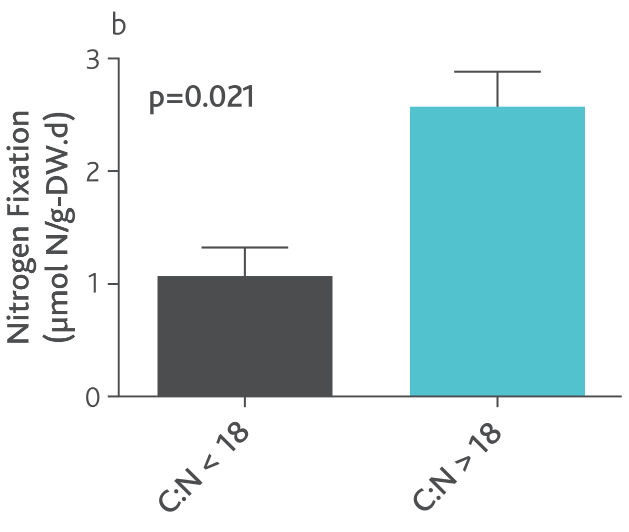

Figure 3.3 Rates of nitrogen fixation shown in seagrass beds in Western Port with a C:N ratio less than 18 and greater than 18.

Determining water quality targets - sediments and nutrients - for seagrasses

Eutrophication caused by excess nutrient inputs is known to be a major driver of seagrass loss globally and it has been postulated that this may be the case in Western Port. Our research suggests that nutrients are not a major threat to seagrass in Western Port based on observations of seagrass density in relation to known point sources of nutrient inputs (Holland et al. 2013). Furthermore, we have shown that seagrasses throughout Victorian estuaries become overgrown with epiphytes when nitrogen inputs exceed 10 t km^{-2} of estuarine area (Woodland et al 2015). By comparison, the rates of nitrogen input to Western Port are less than 1 t km^{-2}, suggesting that, with the possible exception of some sites adjacent to agricultural drains, nitrogen inputs are not a major issue in Western Port.

Reductions in light through increases in particulate matter from re-suspension of bed sediments, or incoming fine sediments from sources such as dredging and catchment runoff, are thought to be a primary cause of seagrass degradation and loss (Erftemijer and Lewis 2006). Previous studies have shown that Total Suspended Solids (TSS) (and hence light) controls the current distribution of seagrass in Western Port (Holland et al 2013). As such, determining water quality targets for light/turbidity thresholds is necessary for effective management. To date, the effects of turbidity on the health of seagrasses has been studied indirectly using shading experiments (See Erftemeijer and Lewis 2006 for review) with the majority showing a reduction in photosynthetic capacity and productivity along with increased mortality (Cabaco et al. 2008, Erftemeijer and Lewis 2006). However, limited studies have examined the direct effects of acute turbidity on intertidal seagrasses.

To investigate this, seagrass (*Z. muelleri*) was collected from Coronet Bay and Crib Point and placed in a series of aquarium experiments to evaluate the tolerance of *Z. muelleri* to acute increases in turbidity (measured as NTU – Nephelometric Turbidity Units). Plants were placed into one of four turbidity treatments; control (average 0.7 +/- 4 NTU, average 31.6 mol m^{-2}d^{-1}), low (average 7 +/- 4 NTU, average 28.1 mol m^{-2}d^{-1}), medium (average 15 +/- 4 NTU, average 21.2 mol m^{-2}d^{-1}) and high (average 29 +/- 4 NTU, average 16.8 mol m^{-2}d^{-1}) for a 5-week exposure period. Following this, tanks were cleaned free of sediment and conditions were returned to control seawater for a 2-week recovery period. During both exposure and recovery, the health of seagrasses was examined through chlorophyll fluorescence measurements, along with samples collected for carbohydrates, chlorophylls, metabolomics and tissue elemental ratios/isotope composition.

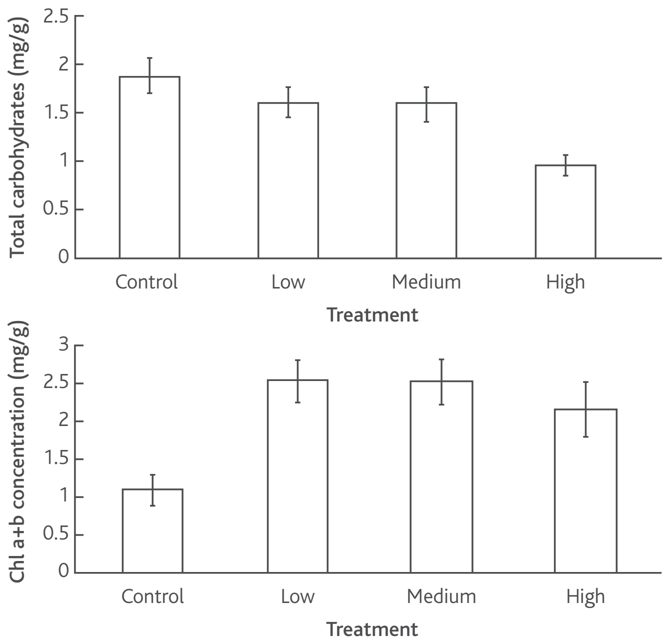

Figure 3.4 Seagrass rhizome total carbohydrate content and leaf chlorophyll content in an experiment with control, low, medium and high turbidity.

Overall, results showed that photosynthetic parameters decreased with increasing turbidity, but returned to control levels after turbidity was subsequently reduced. Increased turbidity caused a decrease in below ground carbohydrates due to reallocation of reserves to supplement growth and respiration, along with increases in chlorophyll concentrations for more efficient light capture (Figure 3.4). Changes in some metabolites were also observed. The levels of turbidity used in this study were consistent with site-specific conditions, indicating that *Z. muelleri* can withstand short-term exposure to turbidity through physiological plasticity. Nevertheless, the longer-term decline of seagrass in the north east of Western Port shows that the reductions in light levels have exceeded the capacity of *Z. muelleri* to adapt. We suggest that further field studies be undertaken to map how the key physiological parameters identified above vary in Western Port to identify areas at risk of seagrass loss.

Species determination and population genetics of *Zostera* in Western Port

The taxonomic classification of *Zostera* seagrass in Western Port has been somewhat confused in the past, with a lack of clear morphological differentiation making species identification difficult. Proper identification of Western Port seagrass species was identified as an immediate research need in the Western Port review, and essential to understanding the physiological barriers to recolonisation and the efficacy of potential restoration actions. Historically, *Z. muelleri* was considered the dominant intertidal species in the bay and morphologically distinct from the main subtidal species, *Heterozostera tasmanica. H. tasmanica* has since been reclassified into four *Zostera* species, including *Z. tasmanica* and *Z. nigricaulis* (Les et al. 2002, Kuo 2005, Jacobs and Les 2009). Based on recommendations in the Western Port review, Keough and Sherman (unpublished) used molecular markers to test the hypothesis that three different *Zostera* species were present in the bay – *Z. muelleri, Z. tasmanica* and *Z. nigricaulis*. Genetic analysis confirmed *Z. muelleri* in the intertidal-shallow subtidal, with *Z. nigricaulis* in the shallow to deep subtidal. There was no molecular evidence for the species formally classified as *Heterozostera tasmanica*.

Genetic diversity can tell us about connectivity of seagrass populations around Western Port which is relevant when thinking about recolonisation. It has also been suggested that levels of genetic and genotypic diversity play an important role in the resilience of a species to a range of environmental stressors (Connolly 2009, Hughes and Stachowicz 2011). Seagrass meadows often consist of a single or small number of species, and populations are often maintained by a combination of sexual and asexual reproduction. Levels of genotypic diversity in seagrass ecosystems are thought to play an equivalent role to species diversity in other ecosystems (Hughes and Stachowicz 2011, Massa et al. 2013). In theory, more genetically diverse populations may have greater adaptive potential, are more resilient to future environmental change and are less likely to suffer from problems of inbreeding (Frankham 2005, Willi et al. 2006, Jenkins et al. 2015).

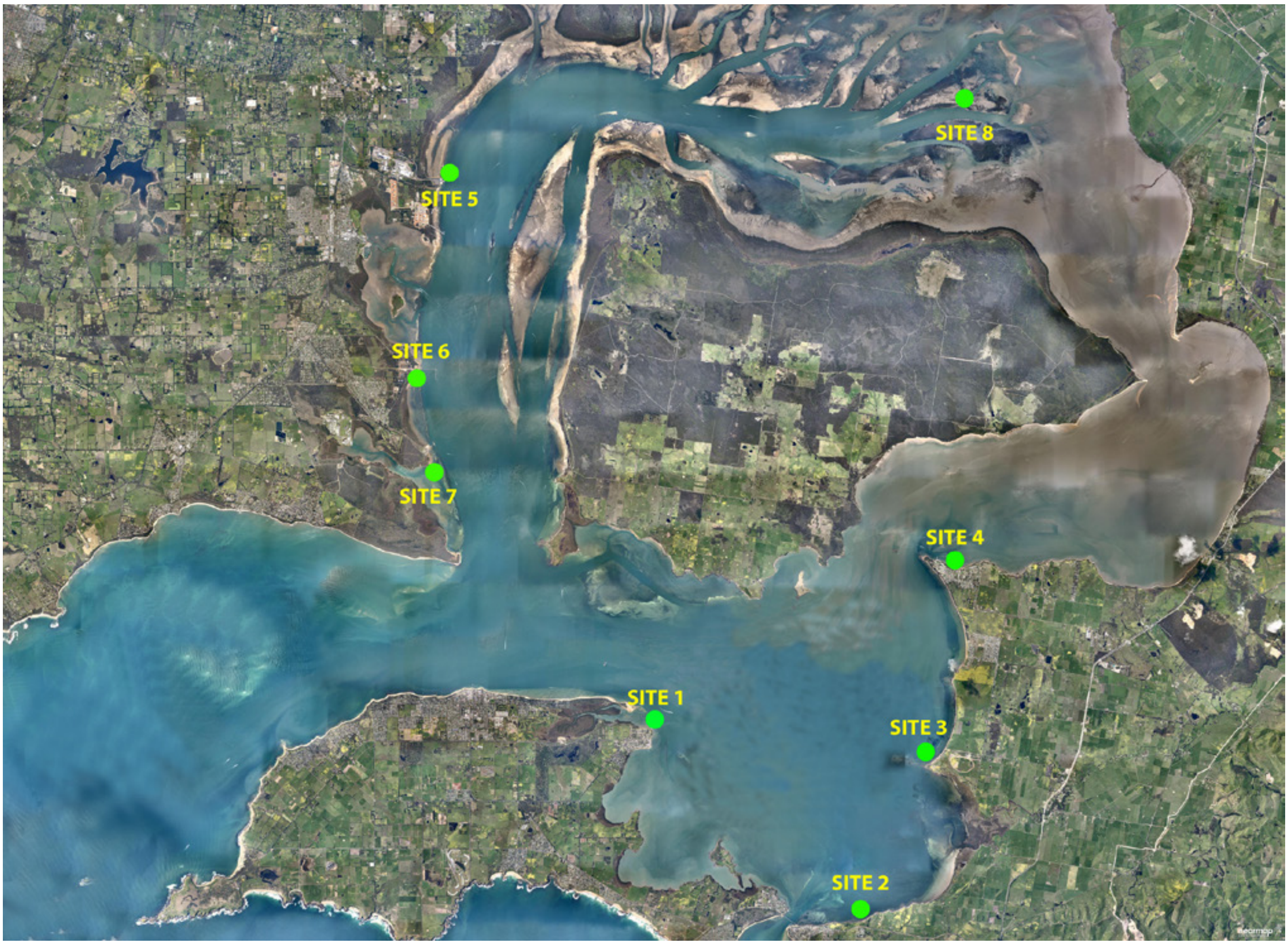

Figure 3.5 Sites where seagrass, *Zostera* muelleri, was collected around Western Port for genetic analyses.

In addition to determining which species of *Zostera* are present in the bay, a further study was done to examine levels of genetic diversity and patterns of population connectivity. A total of 360 *Z. muelleri* samples were collected from the intertidal zone at eight sites around Western Port (Figure 3.5). Moderate levels of genetic diversity were detected suggesting that sexual reproduction is important for population maintenance in the bay. Genetic patterns also showed that Rhyll had the most clonal population – indicating that asexual reproduction is important for maintaining seagrass cover at that site. Alternately, low genetic diversity may indicate that the population was established by only a small number of individuals (i.e. founder effect). The highest numbers of genetically unique individuals were found at Corinella and Crib Point, suggesting that sexual reproduction plays an important role in maintaining seagrass populations at these sites. In the Upper North Arm, genetic diversity was much lower than expected, suggesting inbreeding may be occurring within the site.

The sites that shared clones were: Wooleys Beach and Coronet Bay; Stony Point and Corinella; Hastings North and Crib Point; San Remo and Rhyll. This suggests that asexual propagules disperse between sites and act as an important mechanism for dispersal and recruitment (the dispersal of floating seagrass fragments). High levels of gene flow and connectivity between sites were also noted, except for Upper North Arm (Figure 3.6). Given the apparent differences in light adaptation between Coronet Bay and Crib Point, and the genetic uniqueness of *Z. muelleri* in the northeast area of Western Port – where most of the seagrass loss has occurred – further examination of the connectivity of the seagrasses within the top end is required. This would enable the potential for seagrass colonisation and recovery to be determined, as well as the tolerances of these intertidal populations to environmental changes – especially high turbidity levels.

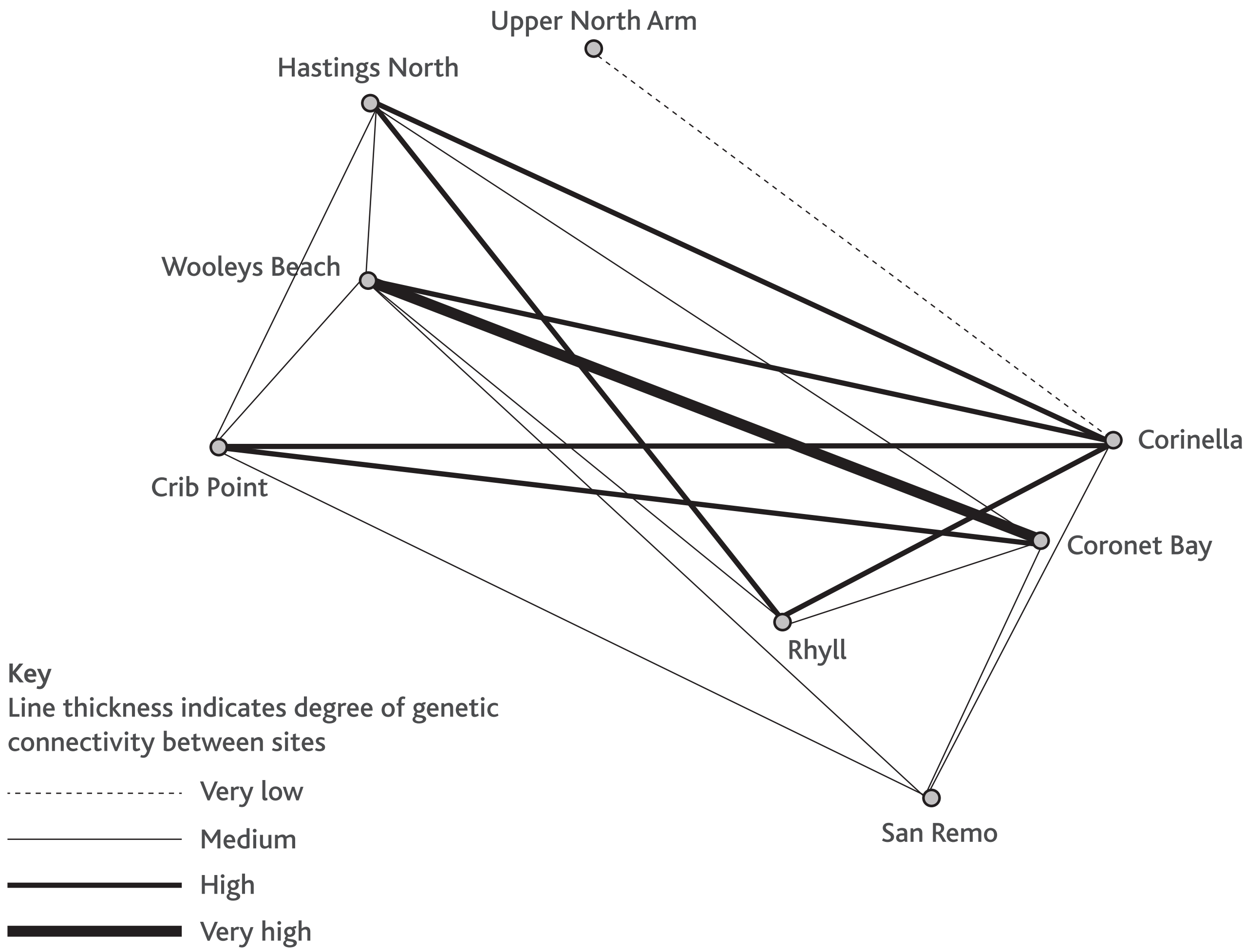

Figure 3.6 Network analysis showing connectivity of seagrass populations, *Zostera muelleri*, between sites in Western Port.

Future directions and opportunities

3.1. **Further examination of the patterns in seagrass metabolite concentrations is needed before this technique can be applied as a robust indicator of stress.**

3.2. **Undertake field studies that link light climate with physiological indicators including carbohydrates, chlorophyll a and morphology to find indicator thresholds for light stress in comparison to experiments described here.**

3.3. **Nitrogen fixation by seagrass has been shown to be an important process. Further studies are required to determine the importance of this to food webs within Western Port. This would involve a combination of meta-analysis of existing data followed by sampling key species with missing data.**

3.4. **We have observed that seagrass are able to stimulate nitrogen fixation under more nutrient limited conditions. As this is carried out by bacteria in the root zone, there is a need to better understand how seagrass and their microbiome interact, and the implications of this for seagrass health more broadly.**

3.5. **Undertake further assessments of seagrass tolerance to elevated turbidity using specimens collected from the northern section of the bay.**

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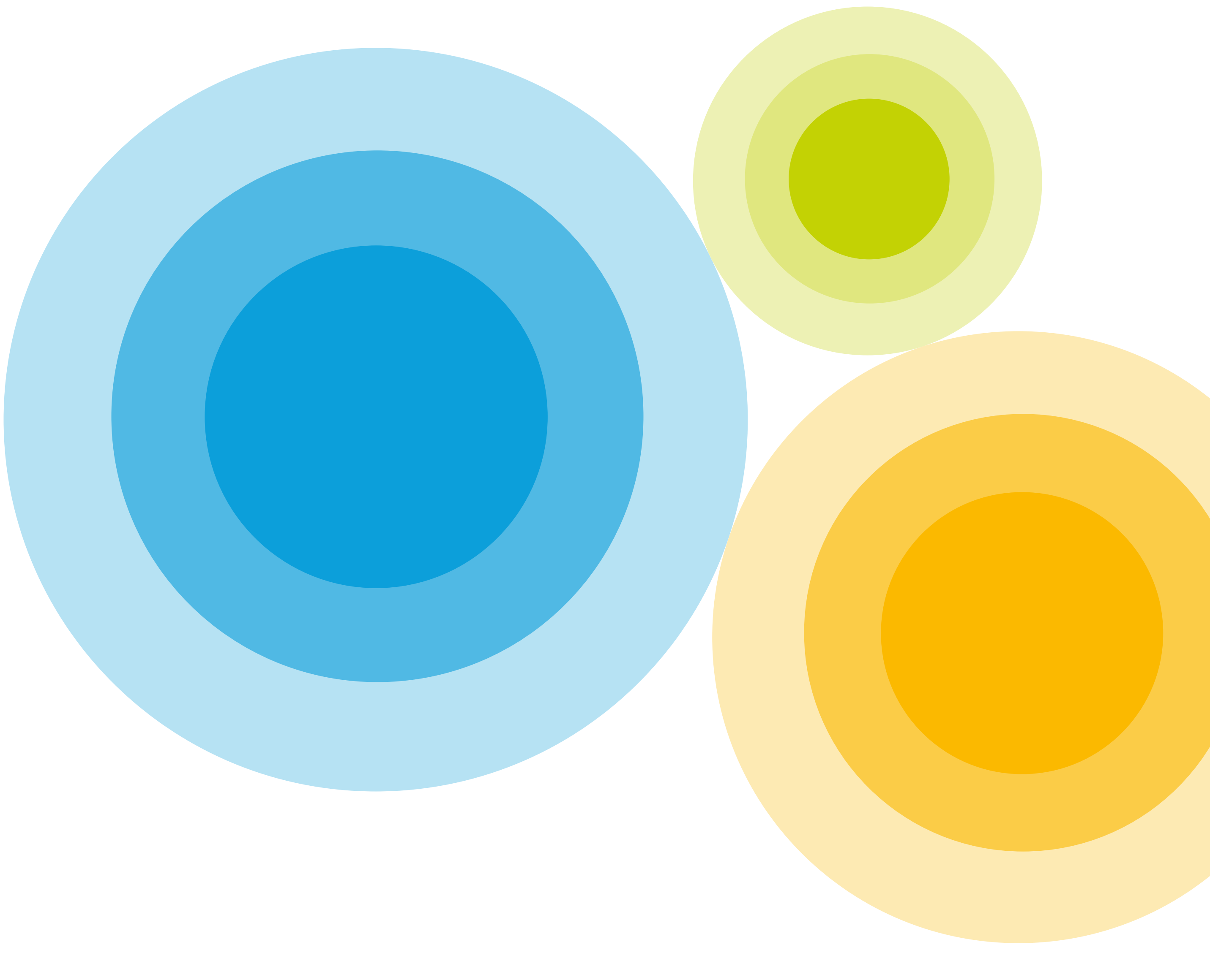

4 Hydrodynamic and sediment modelling to forecast seagrass coverage and recovery in Western Port

Kathy Cinque, Melbourne Water
Patricia Okely and Peter Yeates, Hydronumerics

Research Priorities

This project addresses research priorities identified in the Western Port review (Keough et al. 2011):

Develop a complete sediment transport model

- Refine understanding of effects of seagrass on sediment transport (research priority 5).

Sediment and nutrient thresholds for important plants

- Determine water quality targets for sediments and nutrients that support seagrasses, microphytobenthos, reef algae, saltmarshes and mangroves (research priority 16).

Key Findings

- There is a simulated net loss of fine sediments from the bay that exceeds the current estimated contribution from the shoreline erosion and catchment flow combined. Although there are significant deposition areas north of Corinella and the Rhyll Basin (Hancock et al. 2001), hydrodynamic modelling indicates there is a net flushing of fine sediments from the bay that is driven by residual clockwise currents.

- The model simulation results indicate that current sediment loads from the catchments and from the shoreline erosion along the Lang Lang coast make up only a small contribution to suspended and deposited sediments in the bay compared to the daily tidally driven cycles of resuspension and deposition of existing bed sediments (although these existing bed sediments would have historically been derived from catchment and shoreline erosion sources).

- Improvement in the light climate of Western Port to a level that would enable recolonisation and growth of seagrass across areas of the north and northeast (where seagrass was observed in the early 1970s) is likely to take at least 20 years.

- A key action to improving water quality to levels suitable for seagrass maintenance and restoration is to restrict sediment loads from the catchment and coastline to current levels of around 28 kt yr^{-1}. Suitable water quality for seagrass is then likely to occur once existing legacy sediments have been flushed out of the bay in the coming decades.

- One way to accelerate an improvement in the light climate may be by re-establishing seagrass coverage in areas where success is likely, thereby stabilising the seabed and reducing resuspension. Seagrass replanting is, however, very challenging and identifying areas where success is likely needs to be studied further to ensure the benefits from rehabilitation efforts are realised.

- While improved management of catchment loads and mitigating shoreline erosion are unlikely to have immediate benefits, they remain crucial elements to any long-term solution because they reduce further deposition and future mobilisation of fine material.

Introduction

Western Port is a semi-enclosed embayment on the Bass Coast of southern Victoria and is characterised by a complex series of deep channels, intertidal flats and two large islands (French and Phillip Islands). Seagrass meadows have long been recognised as a key habitat that supports the ecology of the bay and maintains water quality (Shapiro 1975, EPA 1996). Loss of seagrass habitat in Western Port follows a global trend with a world-wide decline of approximately 30% since the late nineteenth century - largely attributed to human activity (Waycott et al. 2009). This is of broad ecological concern because seagrasses are critical, highly productive 'ecosystem engineers', that play a role in sediment movements, nutrient and energy transfer, and provide habitat for a diverse range of animals (Gutierrez et al. 2011). Concern regarding the health of seagrass meadows in Western Port is reflected in state legislation with the protection of seagrass habitats identified as an overarching ecosystem health objective in the SEPP (Waters of Victoria). Actions to address this include reductions in nitrogen and sediment loads into the bay.

The need to protect and restore seagrass has resulted in an increased effort to characterise the sediment-seagrass-light feedback process - which is a potential mechanism for seagrass loss (Wallbrink et al, 2003, Keough et al. 2011). Significant portions of the tidal flats in Western Port appear to be in an unfavourable 'mode' for seagrass ecosystems that is characterised by high turbidity, low light and low seagrass coverage. Shifting the system to conditions favourable for seagrass is likely to be extremely slow and challenging (Adams et al. 2016). A self-sustained recovery of seagrass meadows is potentially achievable by improving environmental conditions - along with improved understanding of recolonization - but will require careful management of future inputs into the bay. Increased fine sediment loads from the draining of Koo Wee Rup swamp for agriculture are thought to be the initial trigger for seagrass loss (EPA 1996; Wallbrink et al. 2003) that shifted the northern and eastern regions of Western Port into a turbid, low light environment unable to sustain seagrass growth. Wide-spread seagrass recovery is a long-term goal and by identifying regions of the bay that are most threatened by additional loss and/or those where colonisation is more likely to occur, it may be possible to stabilize the sediments and improve seagrass coverage.

Models provide us with the predicative capability to identify how factors such as climate change and sea-level rise might modify the basic hydrodynamics and sediment transport and the ecological response that follows. Additionally, they can be applied as an ongoing management tool to investigate how different sources of sediment, now and in the future, may impact bay ecology, as indicated by seagrass cover. This project involved the development of a hydrodynamic, and sediment transport model for Western Port. A model that can forecast seagrass decline and recovery in Western Port may potentially be applied to other systems to assist in global efforts to reverse seagrass loss and avert the collapse of coastal ecosystems (Orth, 2006).

This chapter focuses on modelling the sediment-light-seagrass response by describing:

- The type of model that is being developed;
- How it integrates with concurrent research; and
- How it can be used to assist with the management of Western Port.

Developing the model remains a work-in-progress, so at the conclusion of this chapter key areas of research and development have been identified that will improve the ability of the model to provide reliable decision support.

Addressing knowledge gaps

The Western Port review (Keough et al. 2011) documented field research and numerical modelling to describe the physical environment, aquatic chemistry and flora and fauna in Western Port. It also identified the major threats to the health of Western Port and highlighted the key gaps in current knowledge. The focus here is on modelling (a) the fate and transport of catchment and shoreline sediment loads; (b) the role of sediment resuspension and deposition by currents and waves; and (c) the impact on seagrass coverage. By developing a model that incorporates these processes other knowledge gaps regarding the future impacts of climate and land use change can be explored.

Previous research has provided a sound understanding of the bay-wide hydrodynamic behaviour and the patterns of sediment transport and deposition (Harris et al. 1979, Hancock et al. 2001, Lee et al. 2011, see graphical summary in Figure 4.1). Currents and waves resuspend sediments, and deposit them on the north and northeastern tidal flats during calm conditions. Residual clockwise flow around French Island controls water clarity and sediment distribution through much of the eastern side of the bay. Large amounts of the sediments from the catchments that flow into the northeast of the bay are transported clockwise into the eastern arm (near Corinella), where there is substantial deposition. In addition to clockwise circulation of suspended sediment, local increases from activities in the catchment are likely, particularly where drains and streams enter the bay. Recent research (see Chapter 2) has provided more detailed information on the terrestrial sediment loads from catchments and shoreline erosion.

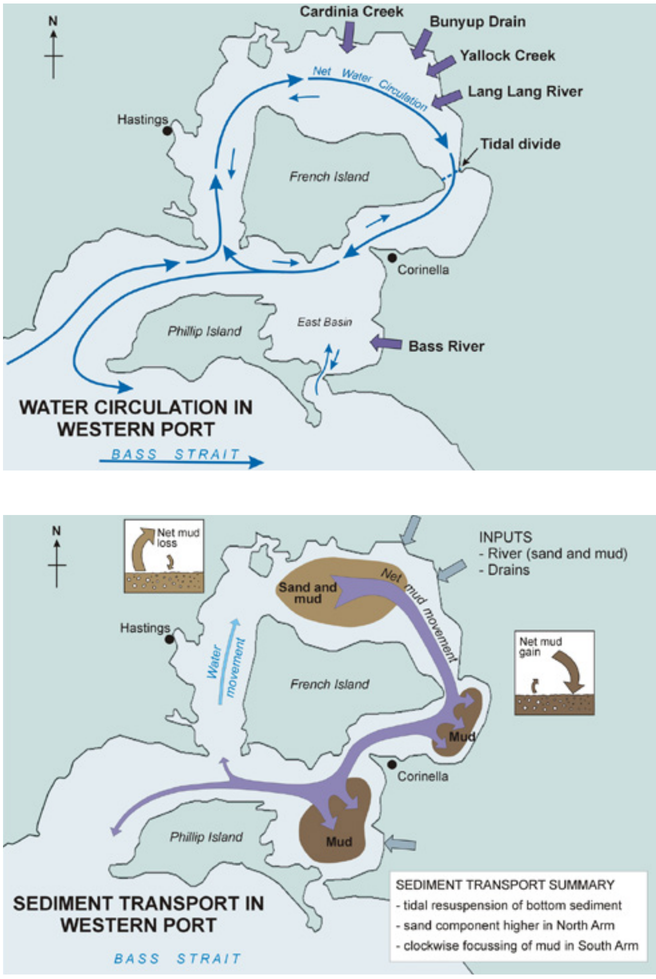

Figure 4.1 Schematic summaries of (a) bay-wide circulation and (b) sediment transport in Western Port (Source: Hancock et al. 2001)

Seagrass can stabilise sediments, meaning that a loss of seagrass has the potential to alter the cycles of resuspension and deposition, and therefore amplify the impacts of bed changes (Suykerbuyk et al. 2016). The long-term recovery of the light climate in Western Port is largely dependent on a decrease in the resuspension and transport of large reserves of fine sediment on the northern tidal flats. The extent of these reserves of fine material and the rate of flushing is, however, not fully understood.

There are gaps in our understanding about the mechanisms that lead to seagrass loss and recovery. One theory suggests that increases in sea bed height due to sedimentation may be exposing seagrasses to more extreme heat resulting in desiccation during low tides (Parry pers. comm.). However, turbidity and its effect on light availability at the seabed is accepted as the main process limiting the growth of seagrasses although it has been important to consider the potential (but previously unknown) impact of catchment-derived nutrients (Keough et al. 2011). The processes of sediment deposition, resuspension and transport in Western Port are complex and require careful attention in the calibration of this model. Understanding the water quality improvements that are required to shift the system towards seagrass recolonization and recovery remains the focus of ongoing research.

Need for a bay model

Building a model that incorporates the key processes that impact on the health of Western Port is a technical challenge, and includes tidal dynamics, surface waves, wind-driven currents, catchment plumes and sediment resuspension and deposition. The complexity of incorporating biological factors such as seagrass growth into the model means that the focus has been on sediment transport as the controlling mechanism for light availability, with the assumption that poor light is a key inhibitor of current seagrass coverage and future recovery.

The integration of catchment models with bay models can provide a holistic picture of regional processes and can be used to forecast the future condition of the bay. This study aims to address this gap, but is only a first step in a process to improve knowledge and management of Western Port's unique environment. Future research efforts, directed towards the development of more comprehensive knowledge, will result in more accurate predictive tools.

Model development

A three-dimensional hydrodynamic model has been developed for simulating water velocity, temperature and salinity throughout the bay (Hydronumerics 2016). The model includes external environmental forcing such as wind stress, surface fluxes, catchment flows and ocean boundary conditions.

In the model, sediments are treated as a concentration of inert particles and grouped in size classes based on early sediment surveys (Hancock et al. 2001): clay (4 µm), silt (30 µm), sand (60µm) and coarse sand (250 µm). Sediment particles are introduced into the bay via terrestrial flow from the four main inflows (Bass, Bunyip, Lang Lang and Cardinia), from cliff erosion along the Lang Lang shore line and from resuspension of internal bed load. Resuspension rates are determined for each particle size based on the particle density, bottom shear stress (above a critical shear stress), particle availability in the bed and user defined erosion rates. Bottom shear is determined from tidal and wind driven currents in addition to spatial time-series maps of surface wind-wave induced bottom stress that was derived using the SWAN model (Booij et al. 1999). The light extinction throughout the water column is determined based on the aggregated effect of the suspended particles which means that the amount of available light for seagrass can be determined at the sea bed and mapped across the bay.

Once the particles are in suspension they are transported around the bay before settling. The resuspension and deposition of the sediments changes the bottom morphology (with the inclusion of packing in the sediments after settling); however, the model does not currently account for flocculating sediments or shielding at the bed.

The model has previously been applied to simulate nutrient cycles, dissolved oxygen, phytoplankton, zooplankton and explicit seagrass growth (Hydronumerics 2016b). The model currently includes light and temperature limited seagrass growth, which will be further developed based on the outcomes of parallel research undertaken by CSIRO (see Chapter 2). The effects of phytoplankton and dissolved organic matter on the light availability are included when water quality is modelled.

Model Application and Calibration

The bathymetry for Western Port was gridded by the *CRC for Spatial Information* based on available LIDAR and boat survey data. Gaps in the data were filled using standard gridding techniques, but the bathymetry remains a source of uncertainty that should be addressed in future research. Inputs into the model include:

- Open ocean conditions of tide, temperature and salinity at the southern boundary (taken from Lorne jetty tidal gauge);
- Meteorological conditions at the bay surface including air temperature, humidity, wind speed and direction, atmospheric pressure, shortwave and longwave radiation (Bureau of Meteorology, Rhyll); and
- Initial sediment maps (Hancock et al. 2001) and terrestrial inflows from the major rivers (see Chapter 2).

A suite of algorithm parameters, most of which are related to defining sediment characteristics (size, density, erosion rate, critical shear and specific light attenuation) were used to calibrate the model against field observations. Long-term records of water quality from EPA's fixed-site grab sample monitoring sites - Barrallier Island, Hastings and Corinella - were used to assess the performance of the model over a decadal period (Figure 4.2). Short term, but temporally high resolution records from moored instrument sites were available from studies undertaken in 2011 (ASR and EPA, 2011). The selected simulation period was from 2003 to 2014.

Figure 4.2 Monitoring sites used for the development and calibration of the Western Port model, including EPA fixed-monitoring sites (red dots) for water quality, and sites for wind data (white dots).

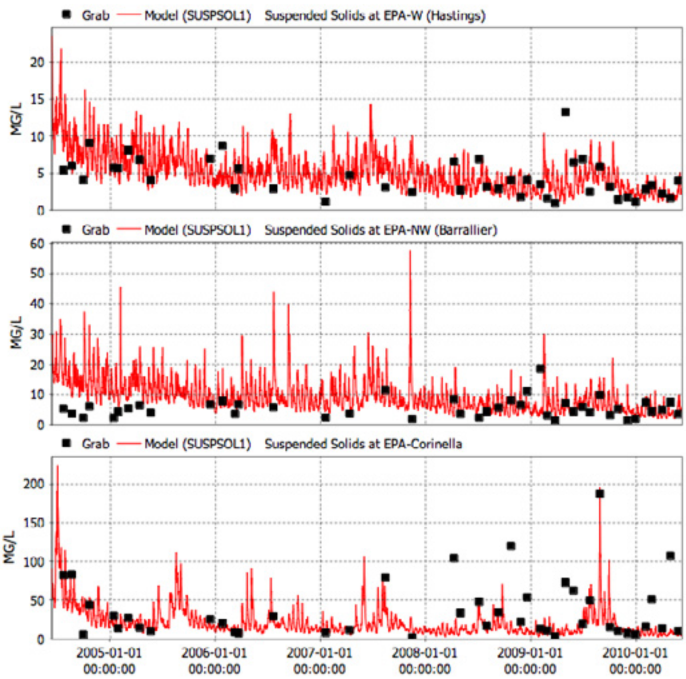

Figure 4.3 Long term EPA monitoring of turbidity at Hastings, Barrallier and Corinella (black squares) compared against model predictions (red lines). Note the changes in the scale for the three panels.

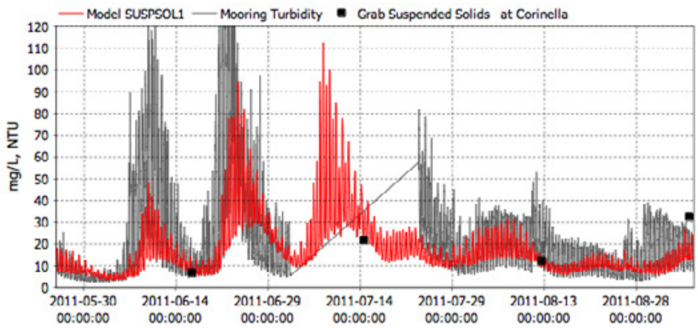

Figure 4.4 Turbidity from mooring data (grey line) at Corinella compared to model predictions (red line).

A critical test of the performance of the model is the ability to reproduce observed trends in concentration of suspended solids over the long term in the different regions of the bay. Figure 4.3 shows that the model is consistent with the observed trends in the long-term EPA monitoring data. It should be noted that model runs include an initial computational 'spin-up' period of 12-18 months after which the model 'settles'. It is the period after 'spin-up' that is considered ecologically relevant and likely to reflect what is happening in the system. In addition, the model can replicate the observed short-term cycles of high and low Total Suspended Solids (TSS) reflecting periods of resuspension and/or load in response to tidal fluctuations and inflow events (see Figure 4.4) that were observed in the 2011 field measurements.

The model was applied to simulate 10-year period scenarios that were selected to explore the effects of resuspension of fine sediment, the presence and absence of seagrass and incoming catchment loads on water turbidity in the bay. The model identifies suitable light climate for seagrass, rather than predicting seagrass extent. The four configurations included:

Baseline - current estimates of seagrass coverage, mud flat extent and depth, catchment loads and Lang Lang cliff erosion;

Sandy Bed - current estimates of seagrass coverage, no mud on the bed, current catchment loads and Lang Lang cliff erosion;

1970s Seagrass Coverage - 1970's seagrass coverage (more extensive coverage than current), current mud flat coverage, and catchment loads; and

No Catchment Inflows - current estimates of seagrass coverage and current mud flat coverage and no loads from the catchments.

Figure 4.5 Distribution of bed fine sediments (<4 µm) (left), catchment fine sediments (centre) and shoreline fine sediment (right) after 5 years of simulation. Units are in g/m^2.

Model results and discussion

Fate and transport of fine sediments

Sediments within the model can be distinguished by their origin meaning that the fate and transport of the major contributors to suspended fine sediment in the bay (catchments, shoreline erosion and bed deposits) can be separated. Figure 4.5 illustrates the extent and location of deposited bed sediments, catchments and shoreline erosion after a decade-long simulation. The figure shows that most sediment deposition is the result of movement of existing bed sediments in the bay. The model simulations indicate that fine sediment loads from the catchments (a total of 7 kt yr^{-1}) and from the shoreline erosion along the Lang Lang coast (an estimated 6 kt yr^{-1}, see Chapter 2) make up only a small contribution to sediment deposition in the bay compared to the resuspension of bay sediments (although these sediments would have historically been derived from catchment and shoreline sources).

The map of re-distributed bed sediment (Figure 4.6) indicates that there are regions along the northern and northeastern shorelines, and in deep channels, where net deposition has created an accumulation of fine material. Fine sediment has been removed from shallow channels and frequently wetted flats in the north ern region of the bay, then flushed from the bay or redeposited elsewhere. Highest deposition occurs on the northeast bank near Lang Lang and along the shoreline around Corinella, at rates of 0.1 to 1 cm per year. The distribution and rates of accumulation agree with results of radiochemical analysis at several locations undertaken by Hancock et al. (2001). Despite this localised accumulation there is a simulated net loss from the bay of 120 kt yr^{-1} which exceeds the current estimated contribution from the shoreline erosion and catchment flow combined. This suggests that there exists a net flushing effect on the bay that is driven by residual clockwise currents. Despite this, there are hot-spots within the bay that are likely to remain problematic in terms of potentially unstable fine deposits. These fine deposits could be mobilised by changes in climate or high energy events and lead to high turbidity and low light conditions that may interfere with seagrass recovery.

The concentration of suspended solids and bed load erosion and deposition in the model scenarios *1970s Seagrass Coverage* and *No Catchment Inflows* closely matched the *Baseline* simulation.

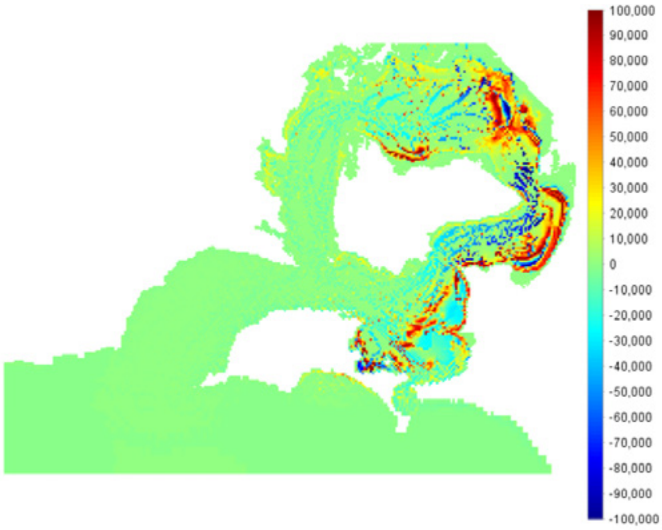

Figure 4.6 Redistribution of initial fine (<4 µm) bed sediment, 2004-2013. Units are in g/m^2

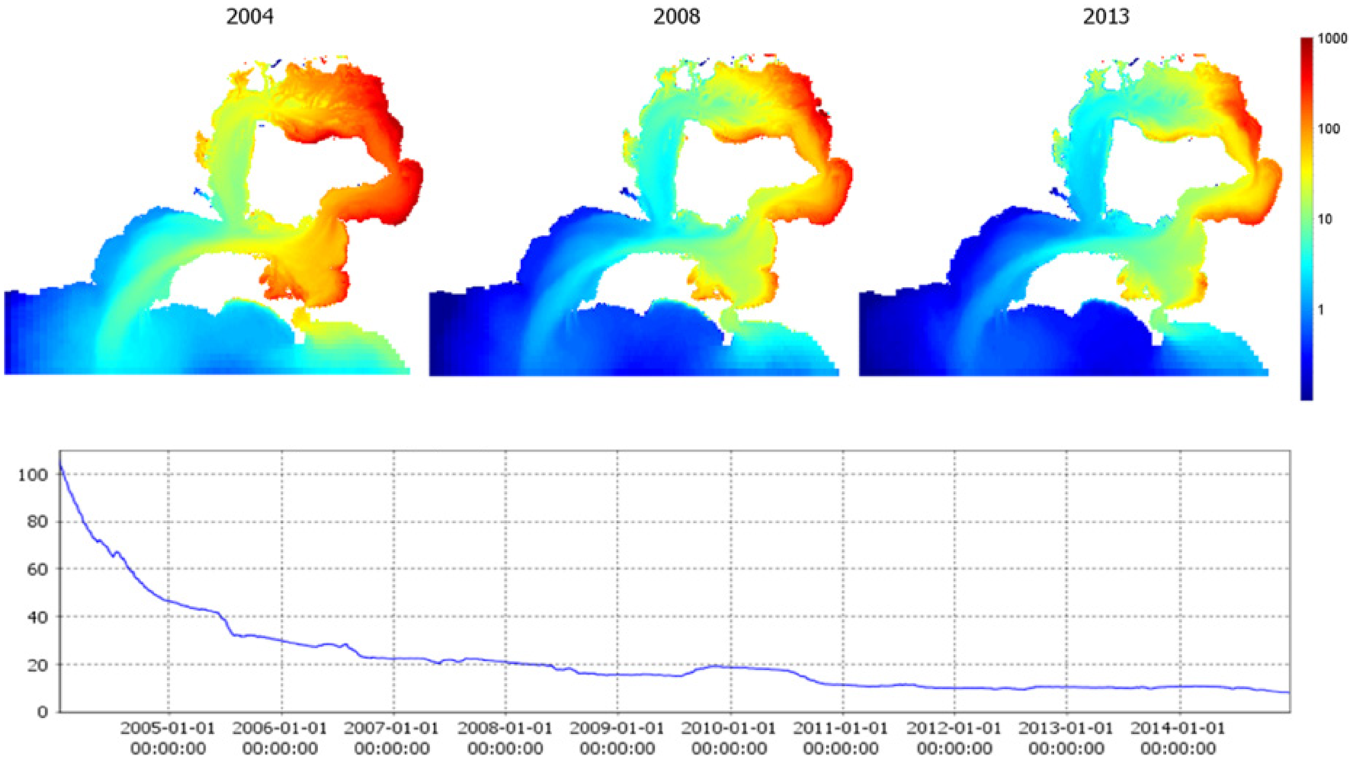

Figure 4.7 Top panel is 95th percentile of yearly simulated suspended clay (4µm) for the year shown at the surface. Bottom panel shows annual rolling mean of clay at the surface of the water at Corinella. Note that elevated suspended clay estimates in the initial 12-18mths of the scenario are associated with model 'spin up' and should be ignored. Units are in mg/L.

Flushing time-scales

By examining simulations over a long time frame, an estimate can be made of the time it may take to redistribute or flush the existing fine material out of the bay (Figure 4.7). The initial sediment bed load distribution was estimated from bed surveys of Hancock et al. (2001) and simulations show that sediment is redistributed within the bay and is also exported from the bay over a ten-year period.

Analysis of model results over the 10-year simulation period shows that there is a net decrease in the concentration of suspended sediment in the water column. This could indicate that - sediment is slowly being flushed out of the bay through the hydrodynamic processes and/or suspended sediment is being transported and deposited within the bay to less dynamic areas. Modelling shows that the region of highest suspended solid concentration is along the eastern shoreline, where the largest deposits of bed load occur, and highest contributions from inflow and cliff erosion are also located. As the model simulation progresses, suspended solids concentrations in the water column continue to decline (Figure 4.7).

The initial concentration of clay in the water column at Corinella after model spin-up is approximately 20 mg/L, reducing to 10 mg/L after 8 years of simulation. The *Sandy Bed* scenario, which has no initial seabed deposits of clay, indicates that with only current levels of shoreline and catchment erosion contributing to suspended clay the expected mean concentration at Corinella is approximately 1 mg/L. Similarly, the scenario without any catchment loads showed very little difference to the *Baseline* simulation, demonstrating the small influence that catchment sources of sediment have on the currently observed high turbidity events.

By simple linear extrapolation, the results suggest that within 20 years the contribution of re-suspended historical sediments to bay turbidity may be comparable to contributions from catchment inputs and shoreline erosion. However, Figure 4.7 shows a levelling out and reduction in concentrations of suspended clays with little change over the final five years of simulation. This means that the system may reach an equilibrium where sustained high turbidity remains a limiting condition for light availability. This can be linked back to the observed deposition maps (Figure 4.6) showing regions of net accumulation of fine sediments in the northeast from all sources.

These fine sediments may sustain turbid currents indefinitely due to a lack of complete removal. Under the *Sandy Bed* scenario, catchment and eroded sediments are transported and deposited in the same regions. While the supply of bed load from earlier inputs (e.g. draining of Koo Wee Rup swamp) may become exhausted, sediments from catchment and cliffs continue to accumulate in these regions.

Longer simulations, perhaps incorporating extreme events, will improve our understanding of the long-term flushing behaviour of the bay. This also requires a better understanding of the impact that changes to the initial condition of clay availability and mobility have on flushing rates.

The cyclic deposition and re-suspension of fine sediments combined with a net export of these legacy sediments from the bay suggests that as the fine sediments are flushed out (likely over decades), TSS levels will become more influenced by catchment inputs and shore-line erosion. A recent Melbourne Water investigation used hydrodynamic and water quality modelling to look at the influence of catchment and coastal sediment load reductions of 5%, 15% and 25% on water quality in the bay (Cinque, unpublished). Model simulations involved load reductions from three main inflows (Bunyip and Lang Lang Rivers and Cardinia Creek) and cliff erosion from the Lang Lang cliffs. Bass River was not included as sediment delivered from this catchment is unlikely to impact the major areas of seagrass loss.

Model results indicate that reducing the total loads from catchment or cliffs is unlikely to have a tangible impact on bay TSS in the short to medium term (at least within the 10-year simulation period, and most likely longer), as legacy sediments will continue to be resuspended and dominate the light climate. In order to reduce the time required to flush legacy sediments that will lead to a suitable light climate for seagrass in the northern and eastern parts of the bay, it is important that future sediment loads from the catchment and coast are less than the flushing rate of the bay. A focus on limiting the amount of cliff erosion is only likely to benefit areas of the bay that are immediately adjacent, especially in wet years when catchment flows have a much greater influence on TSS loads. Accordingly, it is recommended that the current sediment load target for Western Port does not exceed an average of 28kt yr^{-1} to ensure that conditions for seagrass do not continue to deteriorate, and that the period required for flushing existing legacy sediments is not extended. As well as managing sediment inputs from stream channel or coastal erosion, programs to manage the risks of sediment loads to Western Port will also need to consider the potential impacts of a progressively urbanised catchment and changing climate. This includes the movement of sediment from disturbed land during urban construction and rural activities during wet weather.

Light and seagrass coverage

The impact of resuspension and transport of existing bed sediments on the light climate at the seabed is shown by comparing light climates in the *Sandy Bed* and *Baseline* simulations (Figure 4.8). Both have current seagrass distributions but differ in their initial benthic mud load. Bulthuis (1983) observed a decline in seagrass density when available light was reduced to less than 20% of surface irradiance. The 2013 median for percentage of midday photosynthetically active radiation (PAR) reaching the seafloor is over 70% in the northeast region of the bay in the *Sandy Bed* simulation, but declines to less than 20% under current conditions of resuspended bed load, as modelled in the *Baseline* simulation. Regions of lower suspended solid concentrations in the *Baseline* simulation, where there is less impact on the light climate (such as in the central north region), correspond to current seagrass extent.

These results indicate that a broad scale improvement in the light climate - to a level that would enable recolonisation and growth of seagrass across areas of the north and northeast (where seagrass was observed in the early 1970s (Blake and Ball 2001)) - is likely to take decades, thereby making actions to improve water quality an imperative. Seagrass may regrow if a significant change in the mobility of the mud (predominantly clay and silt) in the beds occurs in a way that alters the sediment-light-seagrass feedback. One pathway to improving this long-term prognosis may be by re-establishing seagrass coverage in less impacted areas, thereby stabilising the seabed and reducing resuspension, (i.e. breaking the negative feedback loop). Hydrodynamic modelling can be used to identify where seagrass planting is likely to be particularly successful (i.e. based on light climate and location with respect to clockwise movement of sediments). Seagrass replanting is very difficult and has had limited success to date in Western Port and trials elsewhere but further planting trials using a range of methods are planned for Western Port in the coming years.

While improved management of catchment loads and a reduction in shoreline erosion are unlikely to have immediate benefits, they remain crucial elements to any long-term solution because they reduce further deposition and future mobilisation of fine material.

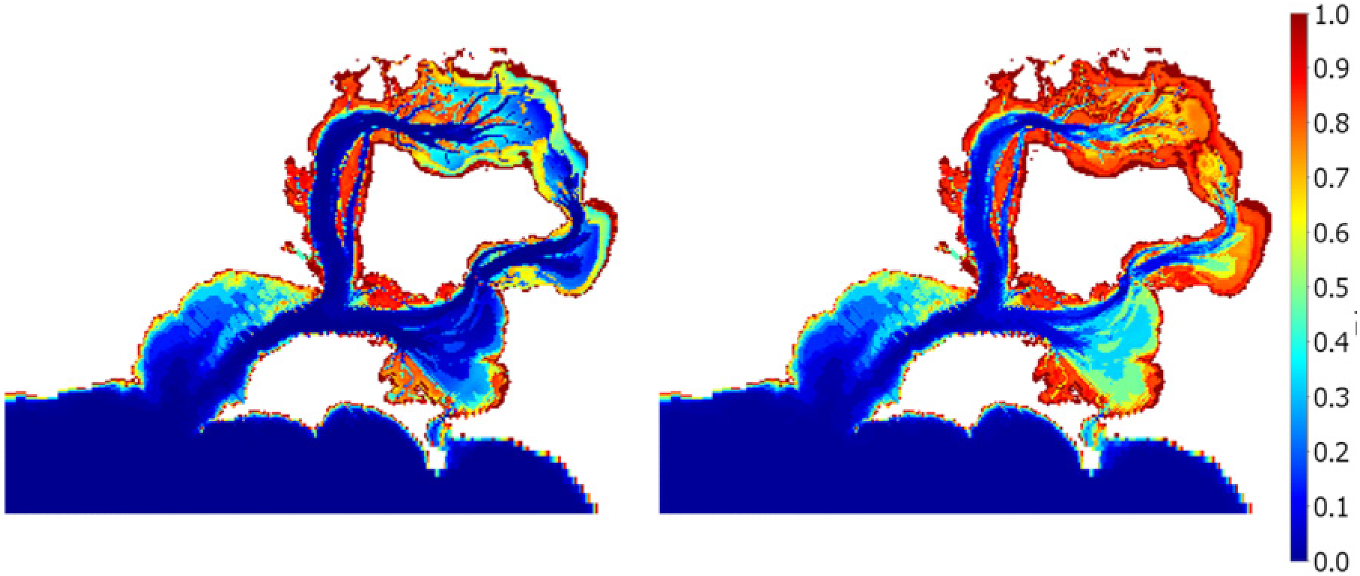

Figure 4.8 Fraction of midday surface PAR at bed, median for 2013, for Baseline (left) and Sandy Bed (right) simulations.

Future directions and opportunities

The sediment-seagrass-light feedback process has been identified as one of the major processes controlling current and future distribution of seagrass. Therefore, the fate and transport of sediments within models of Western Port are critical processes to understand and refine. Sediment monitoring also needs to undertaken to determine the spatial composition and characteristics of the bed sediments. A better understanding of the current sediment beds (amount available and mobility) around the bay will assist with determining a sediment mass balance and will allow a better estimate of the length of time it will take for the bay to flush itself of the legacy sediments. Similarly, updating the bathymetry, previously undertaken in the areas where there is still some uncertainty (Hancock et al. 2011) will ensure flow velocities and subsequent settling are more accurately predicted.

Additional areas that will be looked at in future modelling efforts include:

4.1 Longer model runs to determine the likely time frame for the bay to flush itself of legacy sediments and to establish the level of catchment and shoreline erosion that can be naturally managed by the bay. This would inform sediment load targets for Western Port. Longer model runs would also assist by identifying potential areas suitable for seagrass regrowth.

4.2 Incorporation of climate change scenarios such as sea level rise, increased sea temperature, and changed rainfall and streamflow patterns.

4.3 Integration of the dynamic seagrass algorithms developed by CSIRO which include the interplay between above ground and root biomass and incorporate feedback loops to flow and sediment erosion and deposition.

4.4 Collect wind data from additional sites across Western Port ,or employ BoM gridded meteorological model output, to test the sensitivity of a spatially variable wind field in the model.

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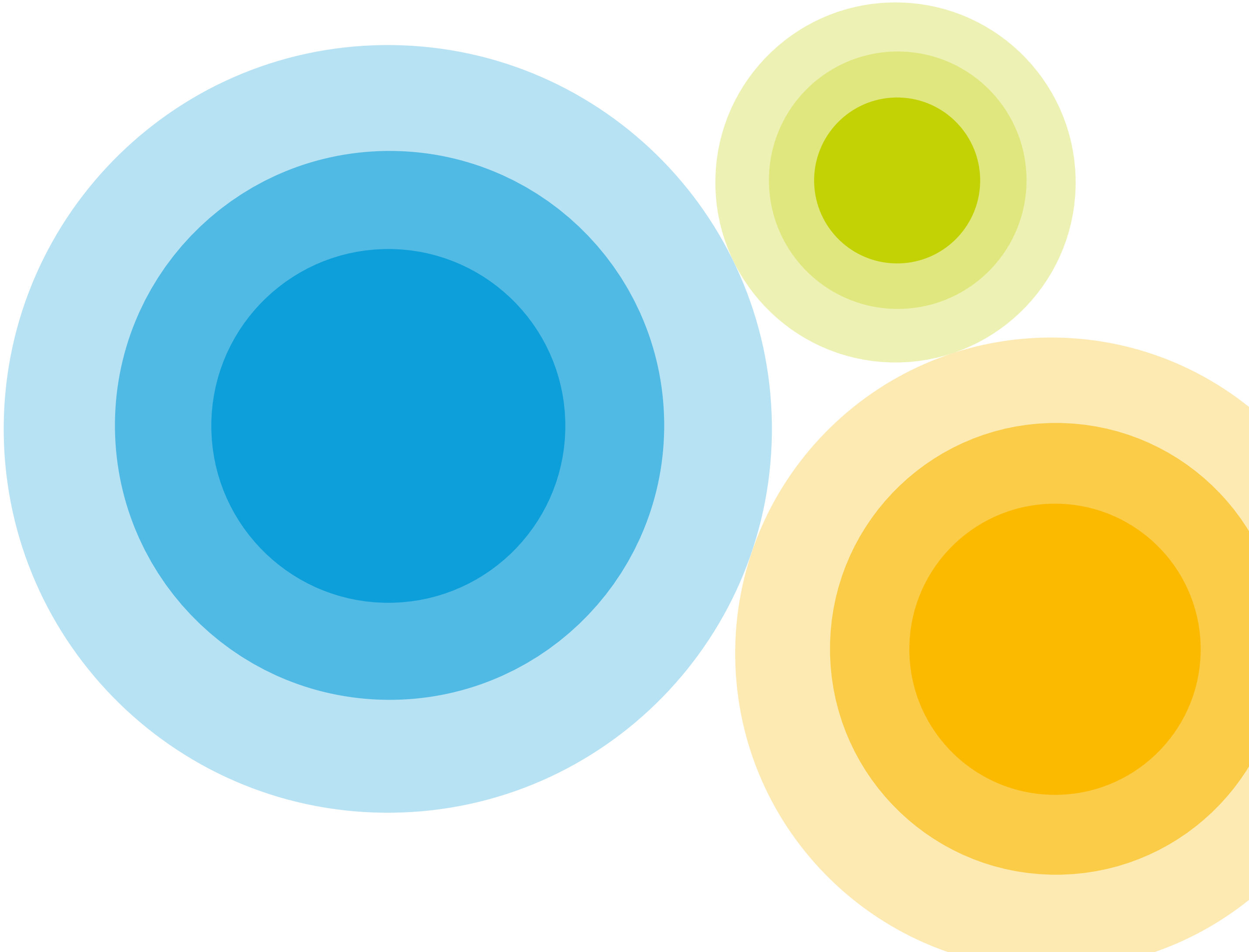

5 Ecological risks of toxicants in Western Port and surrounding catchments

Jackie Myers, Kathryn Hassell
and Vin Pettigrove, University of Melbourne

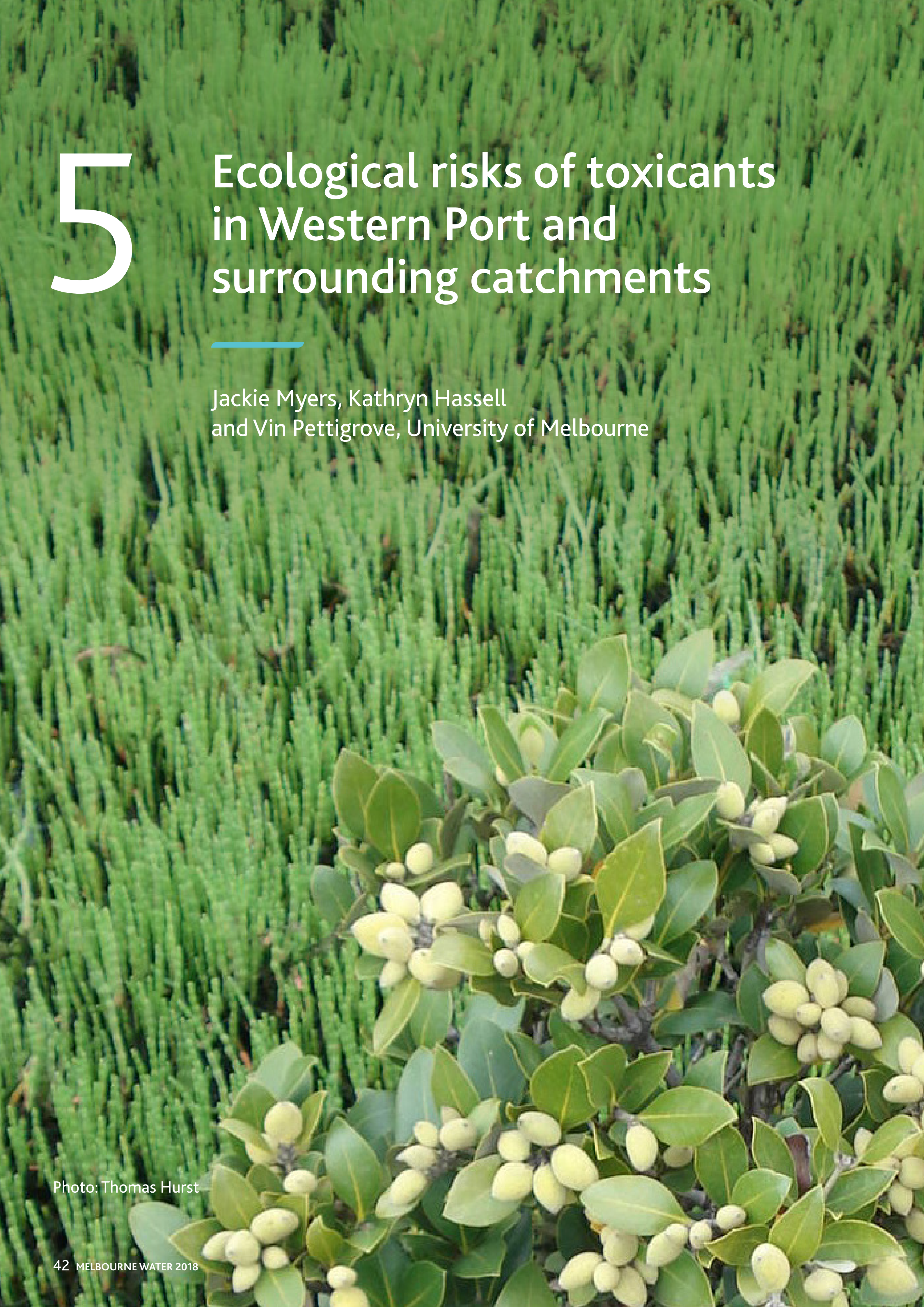

Photo: Thomas Hurst

Research Priorities

This project addresses a number of research priorities identified in the Western Port review (Keough et al. 2011) under the *Toxicant* research theme:

- Initial estimates of risk from toxicants (research priority 36)
- Impacts of toxicants on vegetation (research priority 37)
- Investigate toxicant effects (and climate change) on fish (research priority 38)

Key Findings

- Levels of toxicants including heavy metals, hydrocarbons and organotins are a low risk to the Western Port environment. Pesticides including fungicides, herbicides and insecticides pose a moderate risk to flora and fauna particularly within the freshwater reaches and estuaries, but also within the bay near the mouths of rivers and creeks.
- Storm events appear to increase the risk of exposure to pesticides, with increased rainfall linked to increased pesticide occurrence and concentrations in the catchments. Pesticide profiles during dry and wet weather showed that fungicides contributed the greatest number of individual compounds detected, while herbicides were most frequently detected and generally occurred at the highest concentrations. Insecticides formed a smaller percentage of total pesticide detections.
- The most common pesticides in Western Port surface waters were herbicides (simazine, prometryn, metolachlor and diuron) and fungicides (iprodione and metalaxyl). In sediments, the herbicide 2,4-D was the most common pesticide detected, followed by the fungicides azoxystrobin and boscalid, and insecticide fenamiphos. Several historical pesticides were also detected including DDT, DDE and dieldrin.
- For both mangroves and seagrasses, laboratory studies determined that diuron was the most toxic herbicide followed by prometryn and simazine. The results suggest potential for toxicity to seagrasses and mangrove seedlings from single herbicide exposure, however monitoring data shows that these plants may be exposed to as many as 22 different herbicides at one time.
- In Western Port, the risks from herbicides to aquatic plants are likely to be greatest in the upper estuarine and freshwater areas of the catchment, with significantly lower risks in the bay (based on Australian water and sediment quality guidelines ANZECC/ARMCANZ 2000).
- Studies to date have found that pesticides are primarily associated with agricultural, rather than urban, areas and that levels of these pollutants are having biological impacts (i.e. mortality and elevated biomarker responses in invertebrates, inhibition of algal growth, inhibited functional stream health).
- No site-specific impacts were apparent in the fish health study, although fish showed signs of general environmental stress.

Identity, distribution and impact of toxicants

The Centre for Aquatic Pollution Identification and Management (CAPIM) at the University of Melbourne has undertaken monitoring and research on behalf of Melbourne Water to better understand the risks from toxicants to the Western Port environment and provide evidence to inform management decisions. Initial screening studies showed that, in general, toxicants including heavy metals, hydrocarbons and organotins present a low risk to the health of Western Port. However, pesticides were identified as posing a moderate risk to flora and fauna, so more targeted research on pesticides and associated ecological risks was conducted. Several pesticides have been detected in surface waters and sediments including fungicides, herbicides and insecticides that occasionally exceeded available Australian water quality guidelines (ANZECC/ARMCANZ 2000) Monitoring results show while pesticides are most frequently detected in freshwater and estuarine surface waters, they do extend into the bay and are bioavailable. Biological assessments using fish, invertebrates and algae have identified priority areas to concentrate management efforts. This chapter describes the research and monitoring that has been conducted over the last five years to better understand risks from toxicants in Western Port, and makes recommendations for future research.

Figure 5.1 Blind Bight. Early morning fish sampling.

Initial estimate of sediment toxicants in Western Port

This program began in 2012 with a preliminary survey of toxicants in sediments (Stage 1), to identify and quantify heavy metals, hydrocarbons, organotins (TBT, DBT) and pesticides. Forty-nine sites were surveyed within four major zones of Western Port (Lower North Arm, Upper North Arm, Corinella segment, Rhyll segment) as well as inflowing streams and estuaries (Figure 5.2). These included 22 sites located within the bay, 16 in estuaries, four in freshwater and seven near marinas and boat ramps (the latter TBT only). Toxicant concentrations in sediments were assessed against available guideline values (ANZECC/ARMCANZ 2000 and State Environment Protection Policy Waters of Victoria (SEPP WoV)) and results from historical surveys used to assess changes in pollutants over time (see Sharp et al. 2013).

In general, toxicants in Western Port sediments were found to be at low levels and unlikely to impact resident flora and fauna but in localised areas several toxicants were at levels of concern (based on exceedance of guideline values or occurring at concentrations potentially toxic to flora and fauna). These contamination 'hotspots' were generally confined to estuarine sediments and areas that receive flows from catchment tributaries, whilst open bay sites had lower contamination levels. Of key concern was the presence of pesticide mixtures. A total of 17 pesticides were detected, including the historical chemical DDT (and its breakdown product DDE), and when considering their concentrations as a combined total, some types (i.e. herbicides) were at concentrations likely to impact flora and fauna.

Risks from other contaminants (heavy metals, hydrocarbons and organotins) were comparatively low, with the exception of isolated areas receiving catchment inflows or high boating activity where concentrations of several metals exceeded water quality guidelines. For instance, cobalt, nickel and zinc were detected in sediment pore waters at levels exceeding SEPP WoV guideline values, thus indicating moderate potential for toxic effects to benthic biota. Mercury was also slightly elevated in sediments from two estuarine sites when compared to interim sediment quality guidelines, but there is no evidence of any consistent mercury pollution in Western Port. Similarly, organotin concentrations were elevated at two sites situated near high boating activity. Comparison of 2012 survey concentrations with historical results indicated that, in general, concentrations are unchanged for most metals and hydrocarbons. For organotins, concentrations appear to have declined, indicating they are becoming less of a concern in Western Port since controls on their use.

Seven hotspot areas were identified through this initial study where single chemicals were elevated or occurred in complex mixtures that are a potential risk to ecosystem health (Table 5.1).

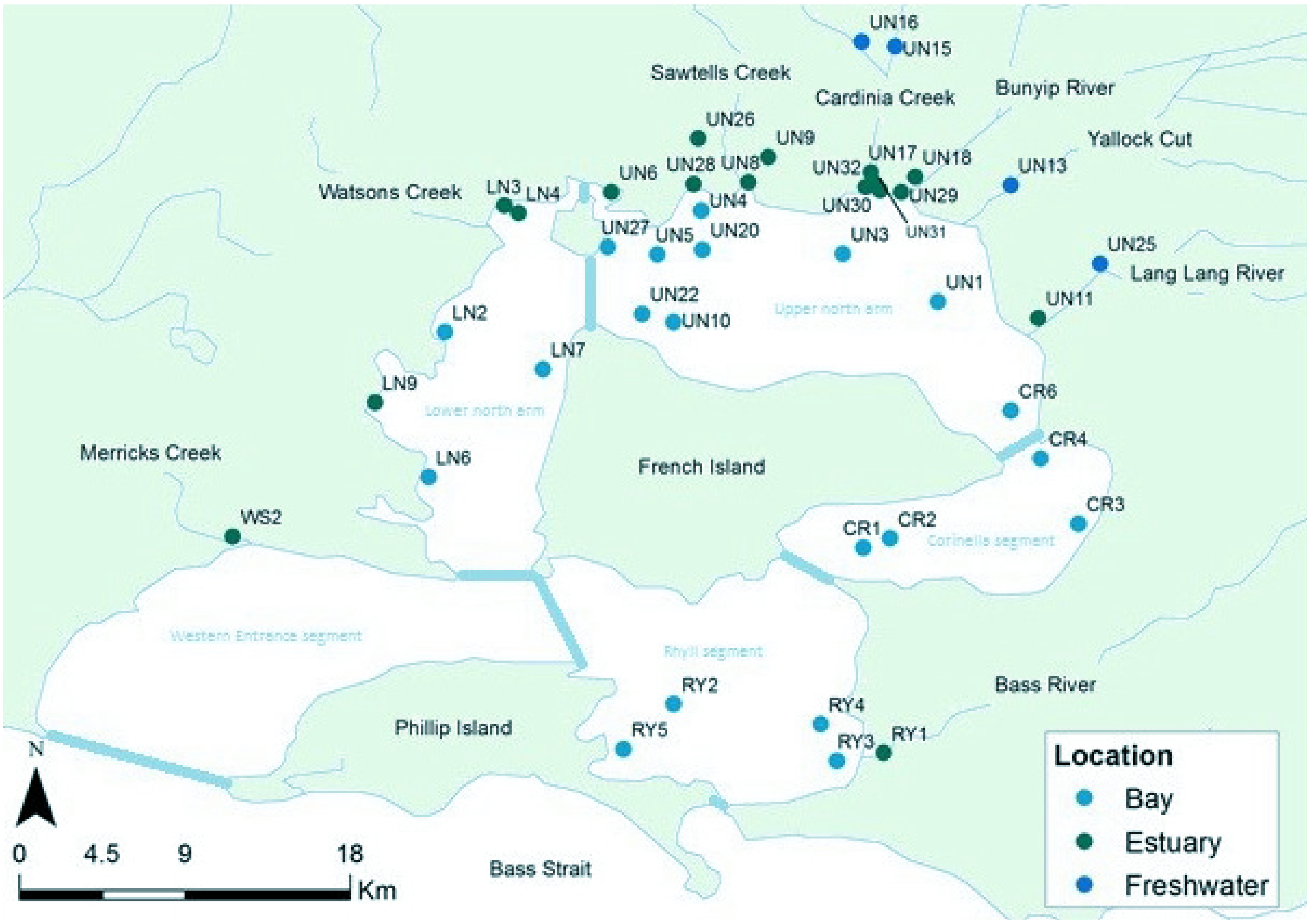

Figure 5.2 CAPIM sediment toxicant sampling sites across Western Port, 2012.

Table 5.1 Areas of Western Port identified as toxicant 'hot spots'.

Waterway or Area	Toxicants of concern	
	Pesticides	Metals & Total Petroleum Hydrocarbons
Western Contour Drain	Simazine, prometryn, linuron, metolachlor, boscalid, oxadixyl, azoxystrobin, cyprodinil	
Watsons Creek Estuary	Prometryn, linuron, metolachlor, boscalid, pp-DDE, pp-DDT, pirimicarb, fenamiphos	
Sawtells Creek Estuary	Simazine, diuron, pp-DDE, bifenthrin	Total Petroleum Hydrocarbons
Deep Creek Estuary	pp-DDE, pp-DDT, pirimicarb, simazine, triadimenol	Aluminium
Cardinia Creek Estuary		Mercury, Cobalt, nickel, zinc and copper
Warrangine Creek		Mercury, Cobalt, arsenic, copper, lead, nickel, zinc
Heavy boat use sites (Hastings and Warneet)		Tributyltin, dibutyltin

Monitoring and evaluation of herbicide risks to key habitats

Following the initial surveys, a more targeted research program was initiated in 2013 to assess temporal and spatial variability of pesticides and their ecological risk to seagrasses, mangroves and algae (see Myers et al. 2015). Two hotspot sub-catchments, Western Contour Drain and Watsons Creek, were monitored monthly over one year to assess the occurrence, concentrations and bioavailability of pesticides in surface waters of fresh, estuarine and bay areas. Surface water samples were also collected during multiple storm events to assess pesticide risks during high flows, and a more comprehensive sediment survey was conducted to understand pesticide risks in sediments from fresh and estuarine regions of the catchments and the bay.

Results of this second study indicated frequent and widespread occurrence of pesticides in the two sub-catchments. These included herbicides, fungicides and insecticides in fresh and estuarine surface waters and sediments of Watsons Creek and Western Contour Drain catchments, extending out into the wider bay. The extent of pesticide contamination could be seen in the number of different pesticides that were recorded at each site, the classes of pesticides detected, the mixtures in a sample, the concentrations detected, and their bioavailability. A total of 43 different pesticides were detected, including some that exceeded ANZECC/ARMCANZ (2000) guideline values in both dry and wet weather. Up to 22 different pesticides were found in surface waters at a single site with an average of six per site. Although sediments contained a lower number of pesticides, concentrations in sediments were generally greater than in surface waters. The use of passive samplers demonstrated that most of the pesticides occur in a dissolved form that is more available for uptake by plants and animals. Storm events appear to increase the risk of exposure to pesticides, with increased rainfall being linked to increased pesticide occurrence and concentrations in the catchments. Pesticide profiles during dry and wet weather showed that fungicides contributed the greatest number of individual compounds detected, while herbicides were most frequently detected and generally occurred at the highest concentrations. Insecticides formed a smaller percentage of total pesticides detected.

Herbicides that interfere with photosynthesis in plants were identified through experiments as posing a low to moderate risk to flora communities in Western Port. Laboratory trials were conducted to investigate the impact of three herbicides -simazine, diuron and prometryn – on photosynthesis in seagrass and mangroves. The seagrasses *Zostera muelleri* and *Z. nigricaulis* were particularly sensitive and photosynthesis was impaired at concentrations below current ANZECC/ARMCANZ (2000) trigger values and at concentrations less than, or equal to, concentrations of these herbicides observed in the environment.

Figure 5.3 Seagrasses in exposure tank system investigating impacts of herbicides.

Figure 5.4 Mangroves in Western Port and potted up ready for experiments assessing impacts of herbicides.

The sub-tidal species, *Z. nigricaulis*, was generally more sensitive than the intertidal species *Z. muelleri*. Single herbicides impaired photosynthesis of 20% of plants at concentrations as low as 0.02 μg L^{-1} for diuron, and up to 5 μg L^{-1} for simazine. Around 50% of plants would likely be impaired if environmental concentrations of exceed 0.4- 6, 1-10 and 11-45 μg L^{-1} of diuron, prometryn and simazine respectively.

The mangrove, *Avicennia marina*, was orders of magnitude less sensitive to these herbicides than seagrasses. Simazine and prometryn did not appear to significantly affect mangrove photosynthesis at concentrations observed in the environment or below trigger values. However, *A. marina* seedlings were sensitive to diuron, with 20% of seedlings having impaired photosynthesis at concentrations of 0.06 $\mu g\ L^{-1}$. This suggests there may be effects on early plant establishment at environmentally relevant concentrations and below trigger values (TVs).

For both mangroves and seagrasses, diuron was the most toxic herbicide, followed by prometryn and simazine. The results suggest potential for toxicity to seagrasses and mangrove seedlings from single herbicide exposure, however monitoring data show that these plants may be exposed to as many as 22 different herbicides at one time. Concentrations of total photosynthesis-inhibiting herbicides detected in monthly monitoring in estuarine and bay areas were 0.2-4.3 $\mu g\ L^{-1}$, compared to 10.9-11.4 $\mu g\ L^{-1}$ in freshwater reaches. Experimental results for seagrasses and mangroves indicate that current guideline values for individual herbicides may not be effective in protecting these habitats because they do not take into account the potential impacts of pesticide mixtures. In Western Port, the risks from herbicides to aquatic plants are likely to be greatest in the upper estuarine and freshwater areas of the catchment, with significantly lower risks in the wider bay (based on Australian water and sediment quality guidelines ANZECC/ARMCANZ 2000).

Hydrodynamic modelling conducted in this study indicated that pesticide inputs from Watsons Creek and Western Contour Drain are likely to have a localised influence in the bay. Inflow water is predicted to remain in the tidal channels and only move laterally during incoming tides where it would be diluted and likely limited to within 5 km of estuary mouths. Modelling also showed that herbicide concentrations in surface waters seaward of estuary mouths would likely be 10-20% of estuarine surface water concentrations and up to 50% following large rainfall events. This suggests that shoreline vegetation within 5 km of estuary mouths, such as seagrass and mangroves, may be impacted, depending on the extent of mixing, initial concentrations in the inflows and the resilience of particular species.

Pesticide sourcing study and fish health assessment

In 2015, a thorough pesticide sourcing program was conducted in the Watsons Creek and Western Contour Drain catchments to isolate and identify the main sources of pesticides (See Myers et al. 2016). Passive samplers were used to track pesticides in surface waters, while ecotoxicological studies were undertaken to determine the impacts of pesticides on fish, invertebrates, algae and functional stream health (FSH). Organic matter decomposition, as determined by the breakdown of leaves and cotton strips, was applied as the standardised indicator of functional stream health (FSH). The study found pesticides in both catchments were primarily associated with agricultural areas, particularly market gardens. A number of agricultural drains from irrigated market gardens that directly feed into the creeks were identified as point sources for pesticide runoff. In both catchments pesticides were detected more frequently, and in highest concentrations, in the mid and lower reaches. Herbicides and fungicides were most commonly detected, with insecticides low or absent. Pesticides were more frequently found in Western Contour Drain samples, and at elevated concentrations, compared to Watsons Creek. This may be due to stream morphology and catchment topography, land use, pesticide use (application rates and timing of application), as well as the chemical properties of the pesticides.

The sourcing study confirmed previous results which showed that elevated concentrations and increased detection frequency of pesticides are associated with wet weather events – as seen in spring monitoring when rainfall volume was 50% greater than in winter. While this may also relate to different rates of pesticide use between seasons, there is currently no data to confirm this. Furthermore, there appears to be potential for groundwater transport of pesticides in both Watsons Creek and Western Contour Drain. Elevated concentrations of pesticides were observed at sites in the mid reaches of both catchments. These significantly declined or disappeared at sites directly

Figure 5.5 Market gardens in Western Port (top), passive samplers to assess pesticides pollution and leaf litter bags to assess functional stream health deployed in Western Port (above).

downstream of the mid reaches, but became elevated again at the most downstream sites. With no major inputs between sites, it is possible that groundwater could be an important source. This requires further investigation.

Ecotoxicological investigations showed there were impacts on invertebrates (shrimp, amphipods), microalgae and FSH in the mid to lower reaches of both Watsons Creek and Western Contour Drain. The highest level of biological impairment was greatest in the mid to lower reaches of the catchment – measured as elevated activity in the general stress enzyme, glutathione-S-transferase (GST) in shrimp and amphipods, and inhibition of microalgal growth. These areas are dominated by intensive agriculture. In each catchment, one particular site showed significant biological impairment, with elevated GST, mortality of invertebrates, greater than 70% inhibition in algal growth and severe impacts to FSH. The high nutrient and pesticide inputs at these sites in the middle of each catchment are likely to have contributed to the observed biological impacts.

Summary of pesticide data 2012-2016

Since 2012, a total of 64 different pesticides have been detected in Western Port catchments, including 55 in surface waters and 20 in sediments (Tables 5.2 and 5.3). Most of individual compounds detected are fungicides (41.5%), followed by insecticides (38.5%) and herbicides (18.5%). However, the most frequently detected pesticides across all study sites were herbicides and fungicides (Tables 5.2 and 5.3) and these also occurred at the highest concentrations. In general, insecticides were detected less frequently and at lower concentrations. The most common pesticides in Western Port surface waters (present in approximately 50% of samples) were herbicides (simazine, prometryn, metolachlor and diuron) and fungicides (iprodione and metalaxyl) (Table 5.2). In sediments, the herbicide 2,4-D was the most common pesticide detected, occurring in 58% of samples, followed by the fungicides azoxystrobin and boscalid, and insecticide fenamiphos (all of which occurred in more than 30% of samples; Table 5.3).

Several historical pesticides were also detected including DDT, DDE and dieldrin. Concentrations and detection frequency were however much lower than for previous studies, indicating their occurrence is likely due to persistence rather than current use. Mixtures of multiple pesticide classes were common with between 2 and 22 different pesticides recorded at single sites. Complex mixtures of chemicals of different classes and different modes of action are of concern because they may interact and be more harmful to flora and fauna than one class of chemicals on its own.

Current Australian guidelines for surface waters and sediments state that, where identified chemical concentrations exceed their trigger values (TV), there is a moderate to high probability of toxicological effects occurring (ANZECC and ARMCANZ 2000, GBRMPA 2010, Simpson et al 2013). Since 2012, TVs have been exceeded for nine different pesticides in Western Port surface waters: the herbicides simazine, atrazine, diuron, metolachlor; insecticides chlorpyrifos, diazinon, dimethoate; and historical organochlorine insecticides pp-DDE and dieldrin (Table 5.3). In sediments, organochlorine insecticides pp-DDE and pp-DDT exceeded interim guidelines (Table 5.3). However, this assessment of potential risk may be an underestimate as no TVs are available for any of the other 46 pesticides detected in surface waters or the 18 in sediments. For many of the pesticides that do have a TV, only half have values that are of high reliability, i.e. derived from an adequate set of chronic toxicity data. Further, pesticides are most commonly detected in complex mixtures, which together have concentrations that frequently exceed single compound trigger values. In general, the total photosynthesis-inhibiting herbicide concentrations in Western Port exceeded TVs more frequently than individual compounds, and at concentrations 2 to 50 times greater. To more reliably determine the risk posed by pesticides in Western Port, a greater understanding of the toxicity of currently used pesticides and mixtures is required, along with the development of TVs of greater reliability.

Table 5.2 Pesticides detected in surface water grab and passive samplers from Western Port catchments from 2012 to 2016. Values in bold indicate a compound that was detected at a concentration exceeding trigger values (applicable to grab water samples only).

Pesticide	Group	Detects Waters %	Maximum Concentrations Detected (Ug/L)			Trigger Values (ug/L)			Maximum concentration in Passive samplers (ug/disk)
			Fresh	Estuarine	Marine	Freshwater[1] 95%	Marine[1] 95%	Marine[2] 95%	
Simazine	H	64	0.71	0.33	0.1	3.2	3.2*	0.2*	4.7
Prometryn	H	64	7.6	2.6	0.12	-	-	-	23
Metolachlor	H	64	**3.1**	**2**	**0.19**	0.02*	0.02*	-	11
Diuron	H	54	**1.1**	0.38	0.08	0.2*	1.6*	1.6	1.6
Iprodione	F	51	2.6	0.17	-	-	-	-	6.9
Metalaxyl	F	49	0.78	0.41	0.03	-	-	-	3.1
Boscalid	F	44	3.3	0.74	0.02	-	-	-	15
Linuron	H	41	1.5	0.88	-			-	0.66
Fenamiphos	I	39	2	0.85	0.08	-	-	-	1.3
Atrazine	H	36	4.8	0.91	0.02	13	13*	1.4	3.8
Dimethomorph	F	32	0.52	0.41	0.03	-	-	-	3.1
Procymidone	F	26	0.16	0.11	-		-	-	3.6
Tebuconazole	F	19	0.08	0.06	0.01	-	-	-	0.56
Chlorothalonil	F	17	-	-	-	-	-	-	110
Dimethoate	I	15	**2.8**	**0.17**	-	0.15	0.15*	-	0.39
Diazinon	I	14	**0.05**	**0.03**	-	0.01	0.01*	-	4.6
Pirimicarb	I	14	0.41	0.03	-			-	1.9
Propiconazole I	F	11	0.03	0.03	-	-	-	-	0.11
Cyprodinil	F	11	0.07	0.05	-	-	-	-	0.21
Carbaryl	I	11	0.14	0.13	0.11	-	-	-	8.4
Propiconazole II	F	10	0.02	0.02	-	-	-	-	0.11
Difenoconazole	F	9	0.04	0.04	0.04	-	-	-	0.27
Diphenylamine	F	8	0.06	-	-			-	0.18
Prochloraz	F	6	0.25	-	0.02	-	-	-	0.042
Chlorpyrifos	I	5	**0.06**	**0.06**	**0.05**	0.01	0.009	-	0.34
1-Naphthol	F	4	-	-	-	-	-	-	1.2
Pendimethalin	H	4	0.1	0.09	-	-	-	-	0.51
Hexazinone	H	4	0.05	-	-	75*	-	1.2*	0.023
Propiconazole_I_II	F	3	0.03	0.03	-	-	-	-	0.07
Metribuzine	H	3	0.08	0.07		-	-	-	0.047
p,p-DDE	I	3	**0.1**	**0.01**	**0.01**	0.03*	0.0005*	-	-
Buprofezin	I	2	-	-	-	-	-	-	0.015
Bifenthrin	I	2	0.04	-	-	-	-	-	-
Methidathion	I	2	0.04	-	-	-	-	-	-
Dioxathion breakdown	I	1	-	-	-	-	-	-	0.05
Methoprene	I	1	-	-	-			-	0.02
Permethrin	I	1	-	-	-	-	-	-	0.03
Triazophos	I	1	0.06	-	-	-	-	-	0.013
Azinphos_ethyl	I	1	-	-	-	-	-	-	0.021
Bupirimate	F	1	-	-	-	-	-	-	0.026
Dieldrin	I	1	**0.06**	-	-	0.01*	0.01*	-	0.04
Phorate	I	1	-	-	-				0.02
Thiometon	I	1	-	-	-	-	-	-	0.032
Azinphos_methyl	I	0.4	-	-	-	-	-	-	0.068
Deltamethrin	I	0.4	-	-	-	-	-	-	0.012
Ethion	I	0.4	0.04	-	-	-	-	-	-
Fenarimol	F	0.4	-	-	0.02	-	-	-	-
Fenchlorphos	I	0.4	-	-	-	-	-	-	0.05
Fenitrothion	I	0.4	-	-	-	-	-	-	0.026
Flusilazole	F	0.4	-	-	0.02	-	-	-	-
Hexaconazole	F	0.4	-	-	0.03	-	-	-	-
Imazalil	F	0.4	-	-	-	-	-	-	0.01
Malathion	I	0.4	-	-	-	-	-	-	0.018
Penconazole	F	0.4	-	-	0.02	-	-	-	-
Piperonyl Butoxide		0.4	-	-	-	-	-	-	0.03

1. ANZECC and ARMCANZ, 2000 2. GBRMPA 2010

* A low reliability trigger value (Derived from an incomplete data set, using either assessment factors or from modelled data. They should only be used as interim indicative working levels).

Table 5.3. Pesticides detected in sediments sampled from Western Port catchments from 2012 to 2016. ISQG – Interim Sediment Quality Guidelines (Simpson et al. 2013.)

Pesticide	Group	Detects(%)	Maximum concentration detected (ug/kg)	Trigger Values (ug/kg)	
				ISQG-low	ISQG-high
2,4-D	H	58	48	-	-
Azoxystrobin	F	37.0	1	-	-
Boscalid	F	37.0	22	-	-
Genamiphos	I	34.8	21	-	-
p,p'-DDE normalised	I	18.2	2.6	1.4	7
Prometryn	H	17.4	15	-	-
p,p'-DDT normalised	I	13.6	2.7	1.2	5
Flubendamide	I	13	5.7	-	-
Metolachlor	H	10.9	10	-	-
Methabenzthiazuron	H	8	18	-	-
Simazine	H	6.5	5	-	-
Bifenthrin	I	4.5	5	-	-
Linuron	H	4.3	2	-	-
Pirimicarb	I	4.3	1	-	-
Diuron	H	2.2	28	-	-
Oxadixyl	F	2.2	5	-	-
Pyrimethanil	F	2.2	3	-	-
Myclobutanil	F	2.2	2	-	-
Triadimenol	F	2.2	2	-	-
Cyprodinil	F	2.2	5	-	-

Figure 5.6 Fyke nets in Watsons Creek.

Figure 5.7 Seine netting at Churchill Island.

Fish Health Assessment 2013-2016

In 2012, a brief survey of fish health was conducted in Watsons Creek that identified indicators of biological stress in two fish species – both general stress indicators and indicators related to endocrine disrupting chemicals (Sharley et al. 2013). These results, together with evidence of elevated concentrations of pesticides, highlighted the need for more targeted surveys of fish fauna across Western Port.

A comprehensive survey of fish health was initiated in major inflows to Western Port in 2015. The Smooth Toadfish (*Tetractenos glaber*) was selected as the target species because it is abundant, widely distributed and exhibits traits that make it likely to reflect the environmental conditions (including pollution) in which it lives - and is therefore useful for detecting pollution impacts. These traits include limited movement, high site fidelity and foraging on the seafloor.

Fish were collected during winter and spring from four sites in Western Port and one reference site in Port Phillip in spring. Sites were selected to represent impacted and reference sites, based on previous toxicant surveys (Table 5.4). General markers of fish health were measured including a condition factor and organ indices, as well as vitellogenin induction in blood or surface
mucus (a specific biomarker of exposure to environmental estrogens).

Figure 5.8 Smooth Toadfish (*Tetractenos glaber*) used in fish health assessments.

Figure 5.9 Fyke nets in the Bass river.

Histological assessments were done for gonads and livers.

Table 5.4. Summary of Smooth Toadfish sampling locations

Site	Category	Dates sampled
Churchill Island (Phillip Island)	Reference site (internal)	Round 1 (winter, 2015); Round 2 (spring, 2015)
Edwards Point	External reference site	Round 2 (spring, 2015)
Watsons Creek	Impact site	Round 1 (winter, 2015); Round 2 (spring, 2015)
Western Contour Drain	Impact site	Round 1 (winter, 2015); Round 2 (spring, 2015)
Bunyip River	Impact site	Round 2 (spring, 2015)

Fish from all sites displayed changes in condition indicative of environmental stress. While a number of physiological and histological differences were observed in Smooth Toadfish collected from sites in and around Western Port and Port Phillip, there were no strong and consistent results indicating pollution effects at any particular site. General biological measures (condition factor and liver-somatic index) indicated differences in energy allocation in fish sampled from different sites; but these measures cannot be specifically linked to toxicant exposure. Nematode parasites and other infectious agents were observed in the livers of male and female toadfish from various sites. Gonad histology and vitellogen concentrations in blood and mucus showed no signs of endocrine disruption in any of the toad fish that were sampled.

Of potential concern to fish health across study sites was the occurrence pre-cancerous and cancerous lesions in fish livers. While this was observed in <20% of fish it indicates exposure to stressors during their lifetime. Occurrence of the benign or malignant liver tumours did not correlate with fish of specific age or sex and were observed at both reference and impacted sites.

While there was a lack of site-specific impacts, some fish did show signs of environmental stress demonstrating that the measures of fish health that we redeveloped and utilised in this study (i.e. histology and biomarkers) are sensitive enough to detect changes in individual fish. In order to adequately understand the potential impacts of toxicants on fish health in Western Port, collection of a greater number of fish, from a wider range of sites is recommended.

Future directions and opportunities

Research and monitoring of toxicants in Western Port has been conducted over the last five years. Whilst filling important knowledge gaps about broad-scale contaminant levels throughout the catchment and bay as well as likely impacts on specific flora and fauna, this work has identified some important gaps in our understanding of toxicant risks. Based on these findings, we recommend further research to better understand the scale of pesticide risks to Western Port including impacts on resident flora and fauna, risks from new and emerging toxicants and how different methods of agricultural pesticide application relate to pesticide transport into Western Port waterways.

5.1 Assess occurrence of pesticides in surface waters and sediments within additional sub-catchments.

Monitoring undertaken to date has been concentrated in waterways flowing into the north west of Western Port. While initial surveys of sediments suggested little to no contamination in other parts of the bay, more recent studies have indicated that pesticides more commonly occur in surface waters than sediments. Therefore, to understand the extent of pesticide contamination in Western Port, chemical analysis of surface waters more broadly across Western Port is recommended.

While current monitoring and research has indicated that intensive agricultural land uses (such as market gardens) are likely to be the main contributor to pesticides in the northwestern waterways of Western Port, land use in these catchments is significantly changing. Reassessment of pesticide risks is recommended in regions where significant land use changes have occurred or are occurring.

A project led by CAPIM and Melbourne Water will be investigating the temporal occurrence of pesticides in waterways flowing into the northeast of Western Port in 2017/18. This will provide a greater understanding of pesticide risks across the broader Western Port catchment.

5.2 Investigate pesticide effects on key fauna and flora of Western Port with a view to developing Western Port specific toxicant guidelines.

There are currently no guideline values for many of the commonly-used pesticides that have been detected. This makes it difficult to fully understand the risks posed to local flora and fauna by the elevated concentrations and complex mixtures of pesticides. Studies assessing impacts of three photosynthesis-inhibiting herbicides to seagrass and mangrove health indicated potential risks to these communities at concentrations recorded in waterways, and are likely to be sufficient to warrant management intervention. Further ecotoxicological testing on other locally relevant species would help identify the broader impacts of common pesticides that have been detected throughout this research program, as well as mixtures of these.

5.3 Assessment of risks from new and emerging contaminants: Pharmaceuticals and personal care products (PCPPs).

PPCPs are known to be present in waterways across Victoria and there is evidence of ecological impacts near wastewater treatment plant discharge sites (Richmond et al. 20126). There is a lack of information on the presence of these chemicals in Western Port, and so it is not known if they are an environmental issue in the bay or surrounding waterways. An initial screening of waterways for PPCPs is recommended, with a focus on potential hot-spots e.g. areas near substantial wastewater discharges or wastewater reuse.

5.4 Investigate the role of farming practices on the transport of pesticides to Western Port Waterways.

An understanding of chemical transport pathways is needed to develop management strategies that reduce the concentrations and occurrence of pesticides in Western Port. A study assessing the role of application methods on pesticide movement into Western Port catchments and the bay is recommended.

5.5 Fish surveys to be conducted more broadly throughout Western Port and additional external reference sites.

The studies reported here are amongst the most comprehensive studies done on fish health within Western Port to date, however, the existing samples collected from toadfish are not adequate to make a definitive assessment of toxicant risks. Further sampling is needed to characterise ‘normal’ health indicators in this species, as well as identify some key indicators of poor health or environmental stress. This may include physiological, morphological, anatomical, biochemical and histological indicators, and estimates of likely population impacts.

A 2017/2018 project led by CAPIM and Melbourne Water will be investigating fish health across a broader range of sites within Western Port plus multiple reference sites in other bays (Port Phillip, Anderson Inlet, Shallow Inlet). This will assist with interpretation of previous toadfish findings and our understanding of the effect of toxicants within Western Port on fish health. In addition, samples from the first two rounds of sampling have been preserved for future analysis using biomarkers previously used to detect pollution in other Australian locations.

5.6 Investigate health of freshwater and estuarine fish

Chemical analyses conducted to date indicate that most of pollution in Western Port is derived from upstream inflows and tributaries and an assessment of the health of fish living within these catchments is recommended. In estuaries, the Blue Spot Goby would be a good indicator species, while in freshwater reaches the Flat Headed Gudgeon would be appropriate. Both of these fish are small, native species that occur widely in Victorian rivers and estuaries and have life histories and behaviours that make them suitable as bioindicators. In addition to being valuable for field surveys, both of these species are also suitable for caging experiments. Thus 'clean' fish could be deployed within different waterways for a set period of time, from which a series of health indicators could be evaluated to determine if exposure to site waters and sediments is impacting fish health and populations.

5.7 Understand the connectivity of individuals and population structure of Smooth Toadfish throughout the bay

It is assumed that Smooth Toadfish sampled from different estuaries within Western Port represent discrete populations. This is an important assumption when trying to establish site-specific differences in fish health, but is based on limited knowledge of the ecology of this species. An investigation of the genetic structure of fish sampled from different locations may assist in identifying how much dispersal and mixing is occurring amongst Western Port toadfish populations. If populations from different sites are genetically distinct then it will be appropriate to link the health of Western Port toadfish to local water and sediment quality. This project is underway as part of the 2017/18 fish health study.

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6 Mangroves and Saltmarshes

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Photo: Thomas Hurst

Research Priorities

This chapter addresses aspects of the following research priorities identified in the Western Port review (Keough et al. 2011):

Characterise present biodiversity

- Estimate extent of invasion of key habitats (by introduced species) (research priority 22)
- Characterise importance of saltmarshes and mangroves for biodiversity (research priority 24)

Trends through time

- Use historical aerial photographs and extensive ground truthing to quantify changes in vegetation distributions since 1940s; compare with distributions apparent in mapping done in mid-19th century by Smyth etc. (research priority 25)

Resilience of habitat-forming species

- Identify determinants of saltmarsh and mangrove recovery and seedling establishment (research priority 27)

Functional links between organisms and habitat

- Relationships between sea levels, sedimentation/erosion rates and vascular plant communities (research priority 29)
- Mangroves and saltmarsh as habitat for animals and plants (research priority 31)

Toxicants

- Impacts of toxicants on vegetation (research priority 37)

Key Findings

- In Western Port, common cord grass (*Spartina anglica*) has been recognised as a significant threat to intertidal habitats and recent mapping has shown that the extent of Spartina has been significantly reduced following recent management efforts.
- A trial to control invasive tall wheat grass (*Lophopyrum ponticum*) showed a selective herbicide was ineffective and broad-spectrum herbicide resulted in undesirable off-target effects. Alternative control options need to be explored such as manual removal, burning, biological control and grazing.
- There was little difference in overall diversity or abundance of invertebrates in soft sediments within mangrove forests and adjacent non-vegetated areas. Mollusc diversity was greater on mudflats than mangroves while arthropod diversity was greater in the mangroves compared to the mudflats. This suggests that these soft-sediment invertebrate communities are influenced by factors (e.g. sediment type, bed elevation, wave and current energy) in addition to the presence/absence of mangroves.
- Historical aerial image analysis of mangrove forests at three sites in Pioneer Bay over 58-70 years demonstrated that the overall area and density of mangroves has increased, although the amount of change varied between sites. There has been little recolonisation of bare mudflats - potentially because higher wave energy reduces the ability of propagules to recruit in those areas.
- Survival of mangroves planted in Pioneer Bay tended to decrease with increasing exposure to wave and current energy, and was especially pronounced at sites exposed to prevailing winds.
- Mangrove planting was undertaken along the high-energy Lang Lang coastline with the aim of reducing coastal erosion. Survival of seedlings in the first 12 months after planting was found to substantially increase when PVC plant guards were installed, although additional protection measures are required as plants mature (e.g. height adjustable guards, complementary erosion protection structures). Potential risks from barnacle damage also need to be investigated.
- Mangrove field planting trials showed larger seedlings grown in the nursery generally had higher survival rates. The largest seeds collected from the ground had generally higher germination rates and produced larger seedlings than seeds picked from trees. It is recommended that largest seeds are collected in the middle of the summer fruiting season to optimise seedling growth and survival before the pre-winter planting.

Introduction

Mangroves and saltmarsh are intertidal vegetation types found along low-energy, muddy coastlines in many parts of the world and provide a range of ecosystem services including habitat for a variety of terrestrial and aquatic fauna and protection of coastlines against erosion. Mangroves are also increasingly recognised as highly effective at trapping and storing carbon (Mcleod et al. 2011). Mangroves in Western Port are represented by a single species - *Avicennia marina var. australasica*, the Grey or White Mangrove – and are close to their latitudinal limit with the southernmost occurrence 100 km southeast at Corner Inlet. Conversely, saltmarshes are known to increase diversity with increasing latitude (Specht and Specht 1999) and Western Port saltmarshes contain a wide variety of species organised in numerous sub-communities, each characterised by different wetting and drying regimes driving vegetation structure and composition (Boon et al. 2011).

Mangrove forests and saltmarshes have a history of being treated as an undesirable feature of the landscape and in the past, were commonly cleared for agriculture, coastal development, aquaculture and other purposes (Alongi 2002). The pre-European extent of mangroves and saltmarsh in Western Port has, however, remained relatively stable compared to other coastal vegetation communities around the world. Despite this, there have been some notable areas of coastal vegetation disturbance in the bay and ongoing threats, if left unmanaged, are likely to further compromise their health. Furthermore, intertidal ecosystems are expected to be amongst the most affected by the human-induced climate change (Loarie et al. 2009). Understanding the likely impact of current and future threats to coastal vegetation in Western Port is essential for catchment and bay managers in developing the most appropriate management responses.

This chapter will discuss progress in mangrove and saltmarsh as part of the Western Port Environment Research Program since research priorities were identified in the Western Port review (Keough et al. 2011). Other relevant research developments will also be discussed.

Invasion of coastal vegetation by weeds

Mangroves and saltmarsh inhabit harsh environments and this has led to the erroneous assumption that they are largely immune from infestation by exotic plant species. However, the Victorian Saltmarsh Study (Boon et al. 2011) identified 118 exotic species in Victorian saltmarshes. Of these, only two species -

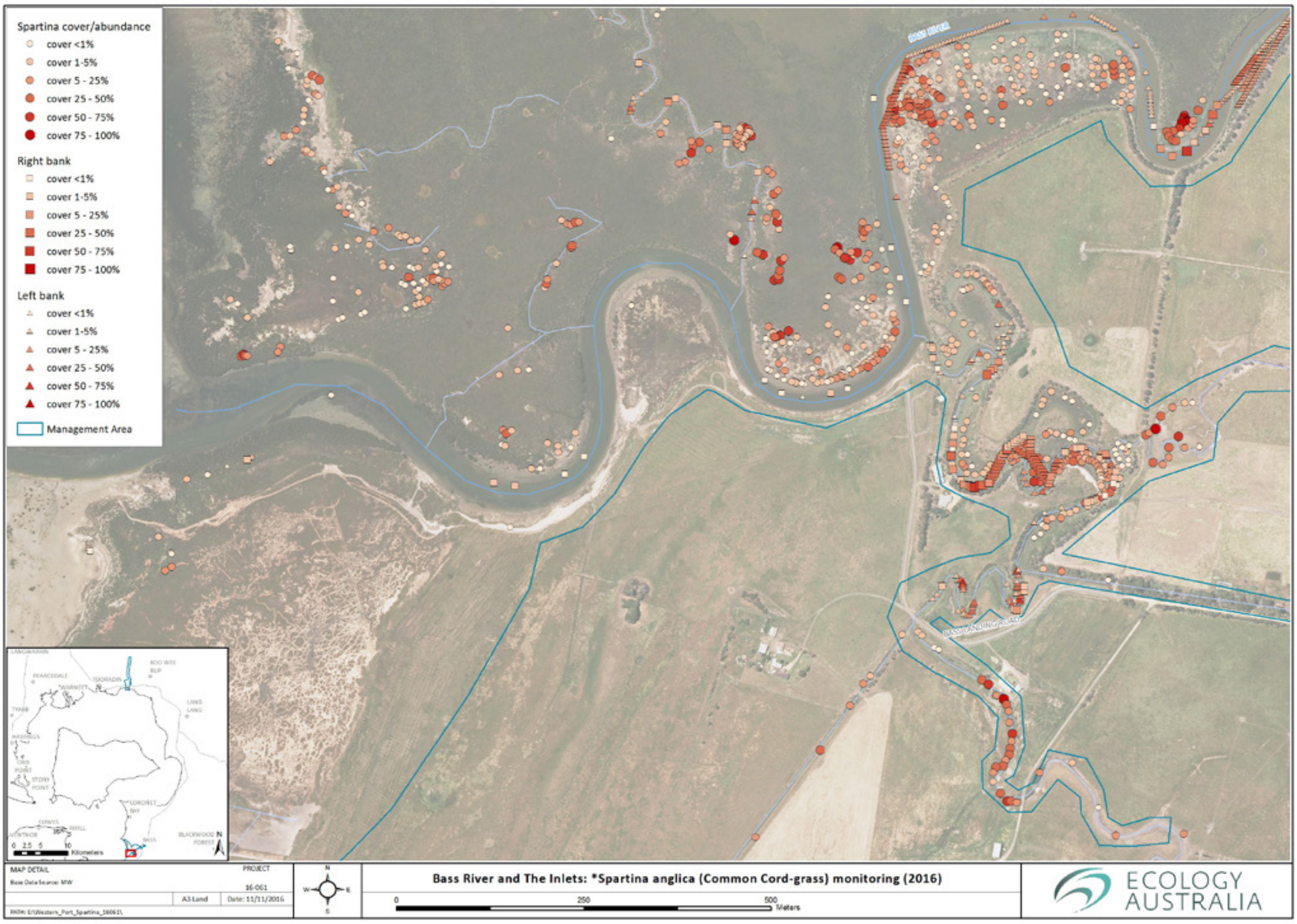

Figure 6.1 Sample map of Spartina distribution in the Bass Estuary (Ecology Australia 2016)

Figure 6.2 Changes to the Lang Lang field site at Western Port following the removal of stock grazing in (a) June 2009, when the site was unfenced and (b) grazed and March 2012, 2 years after fencing.

both from the genus *Spartina* - invade the lower intertidal areas occupied by mangroves in Western Port while the rest invade the drier middle and upper saltmarsh. Spartina is a well-known weed in Western Port and has been subject to intensive control programs by Melbourne Water and Parks Victoria for several years. The remaining invasive species have received far less attention since the Victorian Saltmarsh Study was completed except for a recent control trial of *Lophopyrum ponticum* (tall wheat grass).

Spartina anglica (Spartina or common cord grass)

Spartina spp. are halophytic (salt-adapted) grasses which form dense swards in estuarine areas. Spartina has been introduced in many parts of the world to aid in land reclamation due to its characteristic tendency to promote the deposition and binding of sediments. It has however, become a major problem in estuaries where it can exclude indigenous plants, alter hydrodynamic and sedimentary processes and reduce feeding grounds for wading birds (Hedge and Kriwoken 2000).

In Western Port, *S. anglica* has been recognised as a significant threat to intertidal habitats (DSE 2003). Melbourne Water and Parks Victoria have been managing the weed since approximately 2003 and recent mapping has shown that the extent of Spartina has been significantly reduced. Melbourne Water commissioned a ten-year management plan with the aim of eradicating the species from Western Port (Ecology Australia 2014). The management plan has emphasised the importance of collecting robust baseline data on the extent of Spartina and health of infested saltmarsh to ensure that appropriate analysis and adaptation of management activities can be undertaken (Figure 6.1). The results of the eradication program may assist coastal managers in other regions, as well as managers dealing with nearby Spartina infestations at Andersons Inlet, Corner Inlet and Lake Connewarre.

Lophopyrum ponticum (tall wheat grass)

Lophopyrum ponticum is a large tussock grass that is indigenous to Eastern Europe and Southern Russia and is one of the most serious invaders of coastal saltmarsh communities because of its robust lifeform and tolerance to saline conditions (Boon et al. 2011). Its salt tolerance led to its introduction to Australia in the 1940's as an alternate pasture species for salinity affected soils. Whilst it is still recommended for use by Agriculture Victoria, it has recently been listed as a 'Threatening Process' under the Victorian Flora and Fauna Guarantee Act 1988, so its use in agricultural settings should be re-evaluated.

The current distribution of *L. ponticum* in Western Port is restricted to a few locations including the lower Lang Lang River and Fisher's Wetland on Phillip Island. Infestation in the saltmarshes that line the estuary of the Lang Lang River increased when a fence was erected along the northern bank in 2009. Cessation of grazing along the top of the river bank resulted in *L. ponticum* - previously planted in the adjacent salt-affected paddock (and which had invaded the riparian zone) - flourishing in the fenced area (Figure 6.2).

A trial to examine the effects of different weed control techniques was established at the lower Lang Lang River site in 2012 (Hurst & Boon 2016). Two herbicide treatments (the systemic glyphosate and monocot-specific Fluazifop-P) and a site preparation treatment (pre-herbicide treatment slashing) were tested. The abundance of *L. ponticum* (and other invasive species on the site) and indigenous saltmarsh species was recorded four times over 18 months.

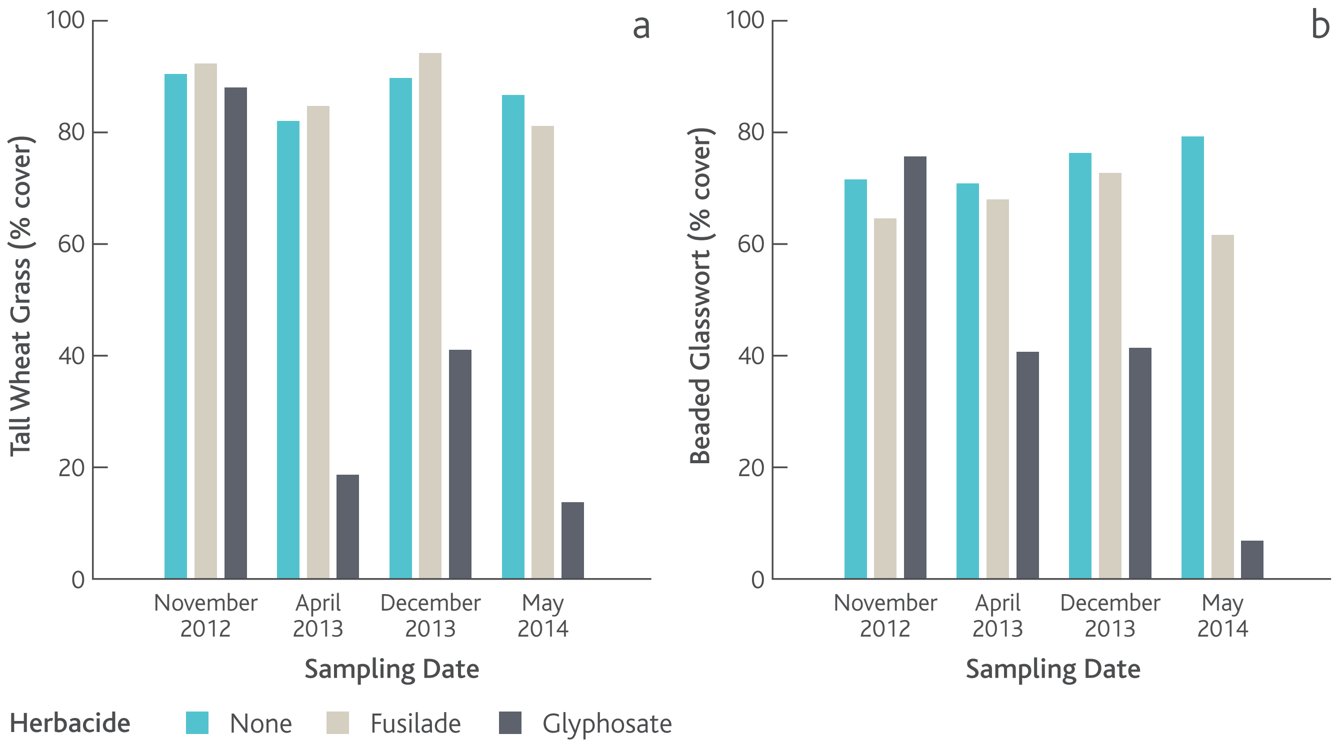

Figure 6.3 Abundance of (a) tall wheat grass, *Lophopyrum ponticum* and (b) the indigenous beaded glasswort, *Sarcocornia quinqueflora* over the period of control trial.

Unexpectedly, the Fluazifop-P did not have a significant impact on *L. ponticum*, potentially due to the intrinsic water-stressed environment in saltmarshes which is known to lessen the effect of the herbicide. Conversely, glyphosate was very effective at controlling *L. ponticum* but was also responsible for severe adverse effects on most of the indigenous saltmarsh species found at the site (Figure 6.3).

The lack of success with a selective herbicide and the undesirable off-target effects of the broad-spectrum herbicide indicates that alternative invasive species control options need to be explored. There are a range of potential options including manual removal, burning, biological control and grazing. While the ecological impact of these techniques is reasonably well understood in terrestrial and riparian settings, there is very little information on how these techniques may affect the integrity of coastal saltmarsh communities. Hurst and Boon's study (2016) focused largely on a single invasive species, but there are many more potentially harmful invasive species for saltmarsh environments. A good understanding of the magnitude of adverse ecological impacts for the most potentially threatening species is required to help coastal managers prioritise their control efforts.

Macroinvertebrate communities in mangrove soft sediments

A number of research priorities identified in the Western Port review (Keough et al. 2011) relate to improving knowledge of the biodiversity associated with mangrove and saltmarsh communities in the bay. A detailed understanding of the life histories and ecological function of the range of species which inhabit mangroves and saltmarshes can help to better understand and predict the effects of environmental change, including climate change. Since the release of the review, research in this area has focused on macroinvertebrate communities in mangrove and unvegetated sediments of Western Port.

Monk (2012) investigated differences in sediment infauna communities between unvegetated mudflats and mangrove habitats. Sampling was undertaken by taking sediment cores from paired mangrove and bare mudflat sites across Western Port. Forty-three taxa (528 individuals) were recorded. The diversity (number of taxa) and abundance of sediment fauna was generally lower than that recorded in a previous study by Butler and Bird (2010) although the latter included additional sites within Western Port Marine National Parks. Overall there was little difference in diversity or abundance between habitats but there was some variation at the site level. In general, mollusc diversity was greater on mudflats (16 taxa) than mangroves (10 taxa) with the reverse being true for arthropods (mangroves 11 taxa to 7 in mudflats). Monk (2012) concluded that infaunal communities are influenced by factors in addition to the presence/absence of mangrove vegetation. The author recommended future studies on soft sediment infauna be done at appropriate spatial scales and incorporate variability in physical conditions (e.g. sediment types, elevation and hydrodynamics) to better understand the drivers of soft sediment communities across mangrove and unvegetated intertidal habitats.

Historical changes in coastal vegetation distributions

Change in the distribution of coastal vegetation has been a significant research focus in Australia and New Zealand. Some studies have found the occurrence of landward encroachment of mangrove forests at the expense of saltmarshes, especially in the estuaries of NSW (Rogers and Saintilan 2008, Rogers et al. 2006). Conversely, mangroves have been found to expand seaward in New Zealand in response to accretion of mudflats due to accelerated catchment sediment inputs – a result of European colonization and agricultural and forestry development (Morrisey et al. 2007). In Western Port, Boon et al. (2011) estimated that around 60-100% of pre-1750 mangrove forest and coastal saltmarsh remain across the bay. They compared current distribution with historical mapping produced by Smythe in 1842 and found that the largest areas of loss were caused by coastal development, agricultural conversion and drainage works. This vegetation mapping was undertaken at a relatively coarse scale and therefore could not detect seaward or landward migration of mangrove and saltmarsh. Another study (Rogers et al. 2005) used aerial photos to determine that mangroves are encroaching into saltmarshes in some areas of Western Port, although this appears to have occurred in the more developed areas around Rhyll and Koo Wee Rup when compared to the less disturbed French Island. This study was restricted to a few sites around Western Port, and there is a current project being undertaken by Deakin University which will expand the use of historical photography to undertake a bay-wide analysis of historical changes in coastal vegetation. This study will also utilise models to predict future distribution under various climate change projections.

Analysis of historical extent of mangrove forests in Western Port

Historical photography has been collected and analysed to complement a mangrove restoration research project (Hurst et al. 2015) in patchy mangrove forests along shores of Pioneer Bay in the eastern part of Western Port (Hurst, unpublished data). These mangrove forests were subject to clearing in the 19th century for shore access, firewood and/or to produce barilla - an alkaline substance used for soap-making (Bird 1975).

Hurst (unpublished) used image analysis software to classify and digitize the remnant mangrove forests shown in the earliest available aerial photographs (1939-1951) at three sites in Pioneer Bay. These were then compared with current distributions shown in recent orthographic imagery (Figure 6.4). As well as displaying change in extent of the forest, this analysis allowed changes in forest density to be detected with the disturbed forests seen to recover over a period of 58-70 years. The analysis revealed that overall area and density increased, although the amount of change varied between sites. Aerial extent of mangrove forest increased (125-328%) across the sites as did density (78-240%). Some of the sites also displayed a seaward expansion of the forest as well as migration of saltmarsh into the formerly forested areas. There was significant lateral spread of the forest at some sites while at others, there was very little recolonisation of adjacent bare mudflats
– potentially because higher wave energy reduces the ability of propagules to recruit in those areas.

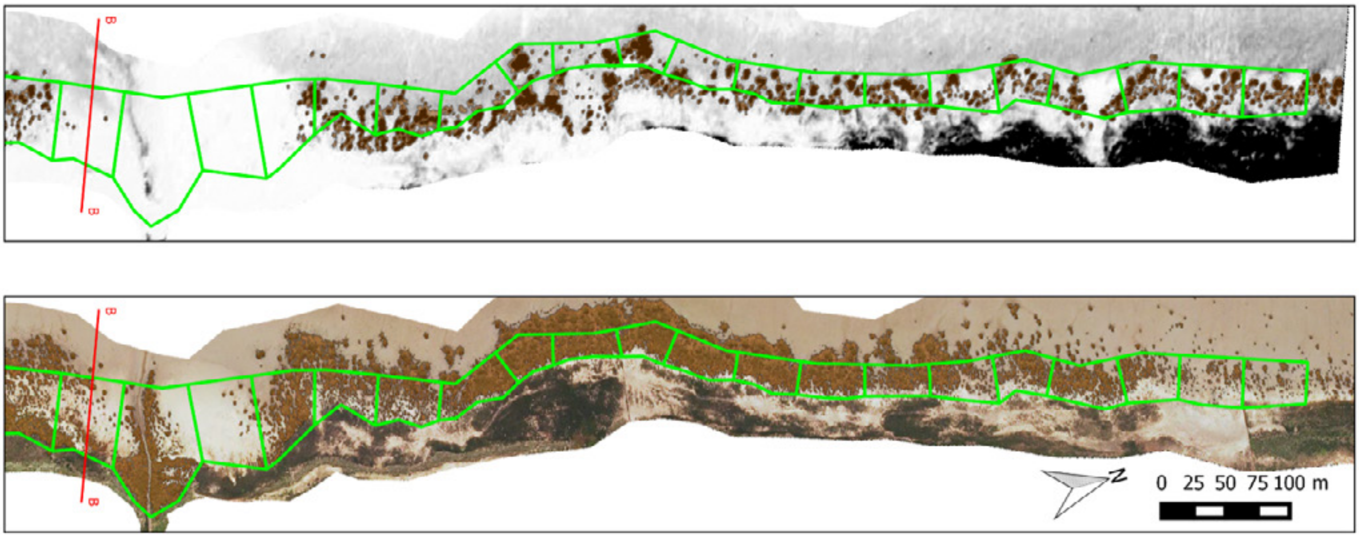

Figure 6.4 Mangrove distribution and density change between (a) 1951 and (b) 2009 along a section of shoreline north of Grantville, Western Port.

Figure 6.5 Evenly aged stands of mangrove saplings growing seaward of older mangrove forests at Stony Point.

The quality of aerial photography in the Pioneer Bay area was patchy, meaning that a thorough time-series analysis of mangrove forest change was not possible. Examining the change in mangrove distribution over time in other locations has shown that mangroves sometimes regenerate on a large scale when periods of calm weather allow mass-recruitment of propagules on bare mudflats (Balke et al. 2015). This phenomenon has been observed in Western Port (Figure 6.5) and further investigation of such recruitment events may lead to greater understanding of the drivers of mangrove regeneration and distribution.

In addition to current research examining historical and predicted future distribution of mangroves and saltmarsh across Western Port, it would also be useful to identify regeneration 'events' and see if these are driven by weather conditions.

Determinants of saltmarsh and mangrove recovery and seedling establishment

Mangrove and saltmarsh vegetation provides a range of ecosystem services, from nursery habitats for aquatic species to coastal protection and carbon sequestration (Mcleod et al. 2011). The growing recognition of the importance of mangroves has seen increased restoration efforts focused on mangroves in Western Port over the last decade. Most mangrove restoration activity in Western Port has focused on establishing a protective band of mangroves along the eroding coastline in the northeast of the bay (the Lang Lang cliffs) as well as some smaller scale projects aimed at restoring mangroves around Grantville. Successful mangrove restoration is notoriously difficult to undertake (Lewis 2005, Primavera and Esteban 2008) and community groups that have initiated mangrove restoration in Western Port have had mixed success to date. Research in this area since the Western Port review has largely focused on understanding the factors contributing to low restoration success rates in Western Port and developing more effective restoration techniques. Efforts have also focused on building knowledge of natural mangrove seed dispersal, recruitment and forest regeneration.

The effect of hydrodynamic energy on mangrove seedlings

Mangrove planting experiments were undertaken in previously disturbed, patchy, mangrove forests near Grantville and Pioneer Bay (Hurst et al. 2015). Nursery-raised mangrove seedlings and mangrove propagules were planted at a number of sites and at different heights on the shore to test the effect of inundation duration. Exposure to hydrodynamic energy (i.e. wind-driven waves and currents (see Figure 6.6) was investigated by planting at increasing distances from remnant mangrove forests. Survival and growth of the seedlings were monitored over two years.

Figure 6.6 Wind-driven waves near Grantville with remnant stand of mangroves.

Results showed that survival generally decreased with an increase in exposure to wave and current energy, and this was especially pronounced at sites that were exposed to prevailing winds. This indicates that stress caused by higher levels of wave and current energy causes higher rates of mortality, although it is unclear what specific mechanism is most important in this process (e.g. defoliation vs. weakening of stems). Results from this study suggest that the initial clearance of the mangrove forests around Pioneer Bay has created an 'alternative stable state'. That is, bare mudflats that were formerly occupied by mangrove forest are not able to recover due to exposure to high wave and current energy – physical forces that were previously dampened by the mangrove forest itself. The implication of this finding is that to restore these patchy mangrove forest areas, planted seedlings (and indeed naturally recruiting seedlings) need to be protected from wave and current energy in order to successfully re-establish.

Mangrove planting for coastal stabilisation at the Lang Lang cliffs

An eight kilometre stretch of coastline in the northeast corner of Western Port consists of an eroding 1-2 m high cliff composed of peaty swamp deposits. The cliff is eroding relatively quickly and is retreating at a rate of about 0.42 m per year, resulting in a large input of sediment to Western Port (Tomkins et al. 2014). An earlier sediment study by CSIRO recommended planting mangroves to reduce the erosive effects of wind-driven waves and reduce the rate of erosion (Wallbrink et al. 2003). Community groups had been attempting to establish mangroves along this coastline for a number of years and Melbourne Water, with the support of the Port Phillip and Westernport CMA, initiated a large-scale mangrove planting project between 2011 and 2013 (Hurst 2013).

For this large scale planting, approximately 25,000 mangrove seedlings were raised in a nursery before being planted along the Lang Lang coast over a three-year period. Very low success rates in the first year, coupled with the findings from the planting experiment in Pioneer Bay (discussed above), led to the trial of a range of measures designed to protect planted seedlings from the high levels of wave energy. Two large-scale protective measures were tested: a PVC pipe pile field and rolled wire mesh. Four hundred seedlings were planted at the landward edge of each trial – half of which had the additional protection of a PVC pipe around individual seedlings from 2013 (Figure 6.7). Three plots of 400 seedlings (half guarded) without a large-scale protective measure were also established as a control.

Figure 6.7 Mangrove seedling protective measures; a) PVC pipe pile field; b) wire mesh roll; c) PVC pipe guard; d) experimental setup along Lang Lang coast.

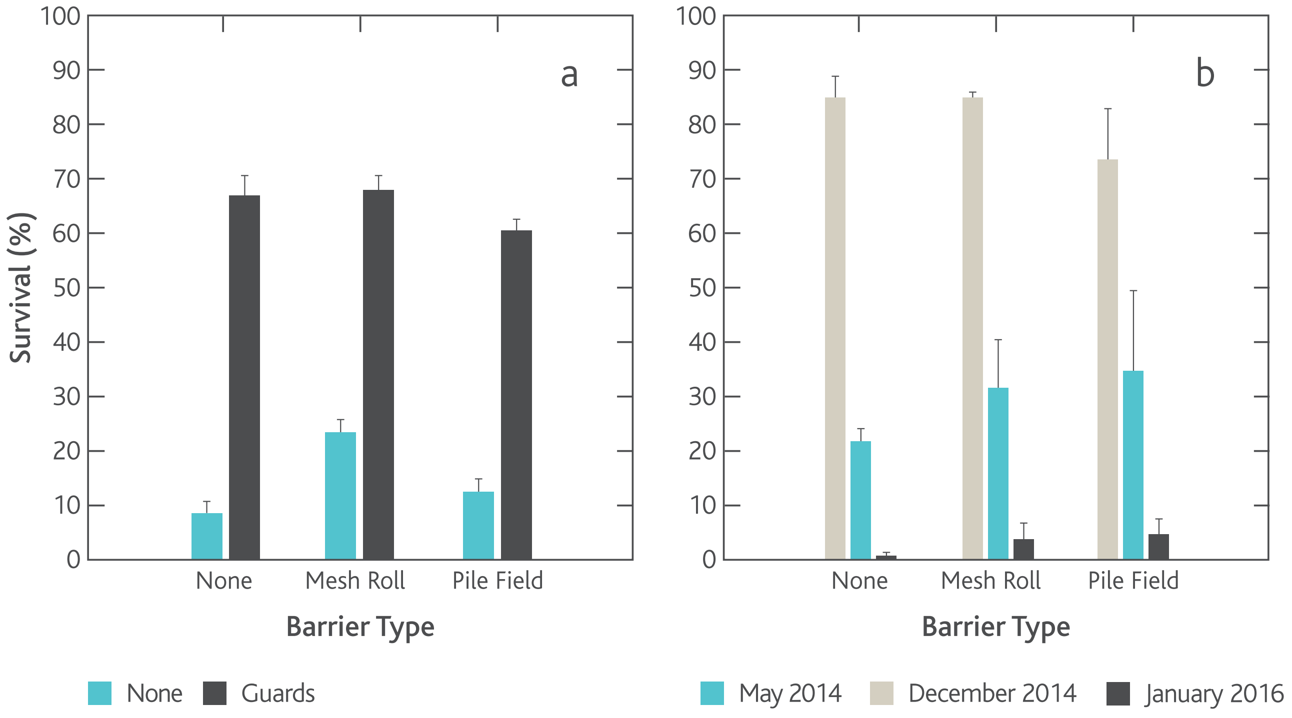

Figure 6.8 Survival of planted mangrove seedlings a) after 12 months and, b) over the length of the monitoring period (all seedlings guarded).

Individually-guarded seedlings were found to have much higher survival than unguarded seedlings at 12 months, while the large scale protective measures had a much lower influence on survival (Figure 6.8a). Survival dropped dramatically over the winter of 2014 and no unguarded seedlings had survived by the following summer. Survival again dropped sharply over the winter of 2015 and less than ten percent of the original number of seedlings survived to summer 2016 (Figure 6.8b). One possible reason for the heavy loss of guarded seedlings is that, while small seedlings were well-protected from wave energy, taller plants that emerged above the top of the guard became susceptible to defoliation, stem damage and breakage as they were pushed against the rim of the guard by strong waves and currents. This indicates that there is an opportunity to develop a mangrove seedling guard that can be adjusted as plants grow.

The Western Port Seagrass Partnership has continued mangrove planting activities along the Lang Lang coast (as well as Grantville). They have modified the PVC pipe guard for individual seedlings by cutting slots in the sides of the guard to prevent sediment accumulation and smothering of seedlings within the guards (Western Port Seagrass Partnership 2016). They have also trialed the use of netting to anchor seeds within guards until they become established to reduce the logistical difficulties and costs of raising seedlings in a nursery. While survival of seedlings within guards has been higher than those without, death of older seedlings emerging from the guards (through defoliation or damage against guard rim) has been detected, consistent with observations from previous plantings. To address this, experimentation with timing of guard removal is continuing. The Western Port Seagrass Partnership has also raised concerns about seedlings becoming infested by barnacles, reducing seedlings photosynthetic ability and weakening plants through the additional weight of the barnacles on the plant.

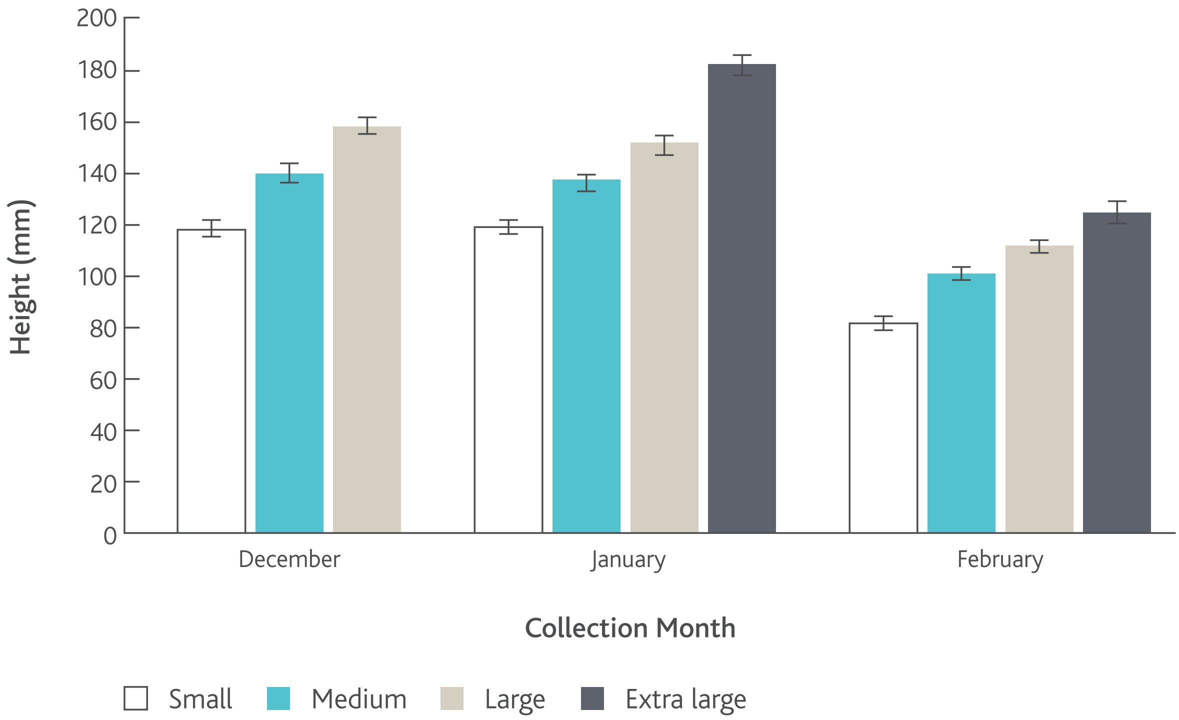

Figure 6.9 The effect of size of propagules and month of collection on the height of seedlings grown in a nursery over 5-7 months.

Mangrove propagule size and harvesting techniques

During the planting experiment in Pioneer Bay and planting activities at Lang Lang it was noted that, in general, larger seedlings grown in the nursery had higher survival rates when planted in the field. A project was initiated to determine the optimal seed size, collection method (picking from the tree versus collecting fallen seeds from the ground) and harvest time (early fruiting season vs. mid and late) for producing larger seedlings in the nursery (Hurst et al. In prep). Seedlings grown in the nursery were also planted in the field to determine whether the largest seedlings survive in higher numbers and what growth characteristics are most important for their successful establishment (e.g. height, number of leaves, stem diameter).

Seeds collected from the ground germinated in the nursery in very high numbers, while seeds picked from trees had germination rates that were lower and more variable. This was most likely due to the difficulty in determining the ripeness of propagules while still attached to the parent tree, whereas natural detachment is a much more reliable indicator.

The largest propagules collected from the ground yielded the largest seedlings at six months when grown under nursery conditions (Figure 6.9). The lack of large seeds early in the season (December) suggests it is optimal to wait until the middle of the season (January) to harvest. The largest seeds were still available late in the season (February), but due to the propensity of Western Port mangroves to slow their growth over colder months of the year, seedlings grown from these seeds were unable to match the size of those grown from seeds collected earlier in the season in time for planting before winter.

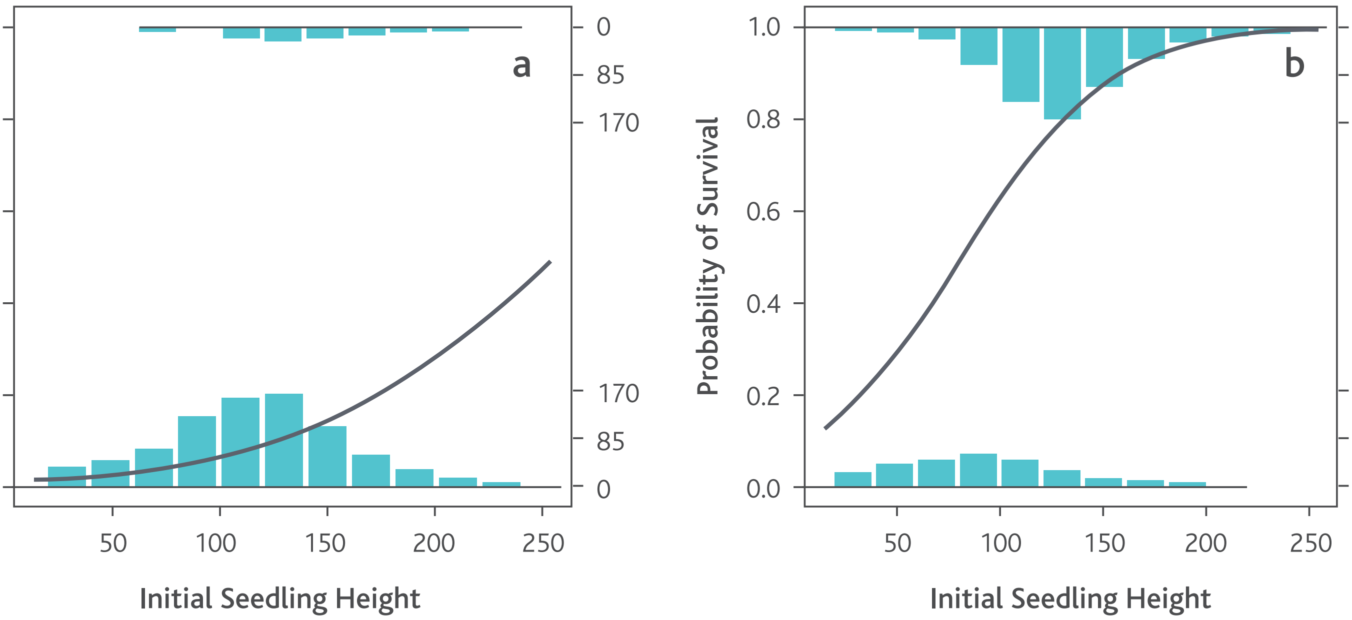

Figure 6.10 Probability of mangrove seedling survival, based on their initial height when planted, after a) ten months and b) thirty months. The histograms represent frequency of observed survival thirty months after planting (0 = dead, 1 = alive) and the line indicates predicted probability of survival.

Seedling height was positively linked to survival ten months after planting in bare mudflats near Grantville (a high wave energy environment (Figure 6.10a)). Of the 70% of surviving seedlings, 80% of seedlings greater than 125mm in height survived while survival of seedlings <100 mm was 50%. After thirty months, overall survival decreased from 70% to just over 10% but initial seedling height still appeared to influence survival (Figure 6.10b). This study demonstrates that early survival of seedlings can be improved by maximizing their size (height) at planting, but longer term survival is driven by local environmental conditions.

Natural mangrove recruitment and regeneration processes

Much of the mangrove restoration in Western Port has been undertaken through the planting of seeds and seedlings at a given restoration site. However, mangrove restoration could also include facilitating the natural regeneration of mangrove forests (Lewis 2005). This was investigated in a study examining mangrove seed dispersal and factors important for recruitment and seedling survival. Stranded seeds and seedlings were counted at approximately 250 sites across Western Port during the height of the seed dispersal season in January, and twice afterwards over a 12-month period. Seed dispersal at a whole-of-bay scale is now being investigated using hydrodynamic modelling. Given the importance of wave and current energy on mangrove seedling survival, this includes wave modelling to determine likely recruitment and survival patterns.

Future directions and opportunities

Invasive species

Exotic plant species remain a key threat to coastal vegetation communities in Western Port and their distribution and impact need to be better understood to ensure that management activities can be tailored appropriately. Spartina infestation is an issue for coastal managers across the globe and there is much to learn from Melbourne Water's program to eradicate the species from a system the size of Western Port. Recommended actions are:

6.1 Collate and analyse data from Western Port Spartina eradication program for publication in broader scientific literature.

6.2 Review high threat saltmarsh weeds identified in the Victorian Saltmarsh Study and determine extent, impact and potential management opportunities across Western Port saltmarshes.

Mangrove and Saltmarsh Biodiversity

Increased research in this area would provide vital information for management regarding the value of coastal vegetation to support biodiversity and is likely to strengthen the justification for protection and restoration of coastal habitats in the bay. There is also a need for baseline data to enable monitoring of the effects of environmental change, including climate change, which is expected to have serious consequences for mangrove and saltmarsh ecosystems.

6.3 Undertake a bay-wide biodiversity study of coastal habitats in Western Port to understand areas of important habitat for protection, and to determine both historical and potential future change in condition. A number of actions are recommended, including research priorities identified in the Western Port review (highlighted in italics):

- An inventory of flora and fauna found within coastal habitats to identify areas of important habitat for protection;
- *A comparison of the resulting inventory to historical data, to detect changes in condition since earlier records* (research priority 21);
- *Comparison of biodiversity to other similar southeast Australian systems* (research priority 24) - to understand the potential state or national significance of the Western Port invertebrate community;
- Modelling of food webs and key dependencies amongst ecosystem components to determine the role of invertebrate communities in maintaining the broader ecological health of Western Port (relates to research priority 18); and
- Identification of indicator species that can be monitored over time to determine the rate and magnitude of the effects of climate change on biodiversity.

Mangrove and saltmarsh distribution

A Deakin University project has recently been initiated which includes further examination of changes in mangrove and saltmarsh distribution. An additional knowledge gap is around the hydrodynamic conditions that drive mangrove regeneration and recovery.

6.4 Historical weather records be examined to determine if lengthy periods of low winds have led to mangrove regeneration 'events' as identified in other locations (Balke et al. 2015). This could provide further information about hydrodynamic conditions that drive mangrove regeneration and recovery.

Restoration and regeneration of mangroves and saltmarsh

The extensive mangrove restoration activities that have been undertaken in Western Port over the last 10-15 years have been hampered by low success rates, although progress has been made towards determining the key factors that are influencing survival rates of planted seedlings. It appears that protection from wave and current energy is a key requirement for mangrove seedling survival. While it is well known that young mangrove seedlings are susceptible to toppling due to erosion of surrounding sediments (Balke et al. 2011), this was rarely observed in Western Port plantings and it is not clear what other specific impacts of hydrodynamic energy are occurring at the seedling physiological level. For example, it is not known if mortality is caused by defoliation during sever weather events, or through long-term weakening of stems and other plant parts by water movement. Insights into these processes would help to inform the design of protective measures (especially at the individual seedling scale) used to help establish mangrove seedlings.

6.5 Investigate the physiological impact of wave and current energy on seedling survival.

Mangrove restoration along the Lang Lang coast has been particularly difficult and is potentially unfeasible without the construction of some complementary infrastructure. Coastal erosion control structures are used widely around the world. A feasibility study could address whether the structures could prevent coastal erosion in the short-term and allow mangrove (and potentially saltmarsh) establishment to provide a long-term stabilisation solution for this coastline.

6.6 Undertake a feasibility study of the use of coastal erosion control structures to complement and assist mangrove restoration along the Lang Lang coast.

Long-term mangrove and saltmarsh monitoring

Climate change-related processes are a major threat to coastal vegetation communities. Understanding the current and future impacts is vital for coastal managers to adapt their management responses in timely a manner.

6.7 Implement a long-term monitoring program to identify the impact of rising temperatures and sea-levels along with more frequent extreme weather events. For example, this could be done by analysing remotely sensed images and tide height data to validate predictions from coastal vegetation models being developed as part of the current Deakin University project.

Monitoring sites could be established at mangrove-saltmarsh interfaces, seaward and landward boundaries of vegetation types, restoration sites, and at dynamic areas revealed by historical photo analysis. Measures could include vegetation extent and condition as well as indicator fauna species identified in the above biodiversity studies. Some of this monitoring could potentially be undertaken by the community (similar to the 'Mangrove Watch' monitoring program which operates in Queensland) which could also help to raise the profile of Western Port's coastal ecosystems.

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7 Fish habitats, fish biodiversity and recreational fisheries

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Photo: Bill Boyle

Research Priorities

This chapter addresses the following research priorities identified in the Western Port review (Keough et al. 2011):

- Determine linkages between fish and habitats (research priority 28)
- Investigate the state of fish populations in Western Port and the effects of recreational fishing on fish stocks (research priority 39).

Key Findings

- Fish species previously found to be common in *Zostera* seagrass were also found in *Caulerpa* algal habitat, and to a lesser extent in *Amphibolis* seagrass habitat. Most species were able to utilise habitats other than *Zostera* seagrass, although some were more specific, such as the Weedy Seadragon (*Phyllopteryx taeniolatus*) that was only recorded in *Amphibolis* habitat.
- Although some species previously recorded in *Zostera* seagrass habitat can utilise other habitats, *Zostera* habitat is nevertheless the most critical for fish biodiversity in Western Port because of its extensive spatial cover (relative to alternative habitats) and important role for larval settlement/development in shallow areas. *Zostera* also supports some unique species, in particular, pipefish and seahorse species.
- Some species such as King George Whiting, Southern Calamari and Southern Sea Garfish had higher catch rates (indicating greater abundance) in areas of higher seagrass cover. In contrast, species such as Snapper and Gummy Shark had higher catch rates in the deeper reef habitats of the Western Entrance Segment and the Lower North Arm.
- An area of high catch rates for most species was the Rhyll Segment, which is strongly influenced by water quality and sedimentation entering the northeast of the bay from the catchment. Catchment management to maintain water quality entering the bay is therefore likely to be critical to maintaining fish biodiversity and sustaining recreational fishing in the bay.
- Overall, King George Whiting stocks in Western Port appear to be improving, stocks of Snapper and Flathead are considered stable, and Gummy Shark stocks in good condition. On the other hand, it appears that stocks of Elephant Fish have significantly declined.
- An analysis of long term trends in Snapper, King George Whiting and Elephant Fish populations and environmental conditions suggests that changes in population abundances are predominantly associated with El Niño and La Niña events, and to a lesser extent recruitment pulses and cessation of commercial netting.
- On a local scale, nitrogen loads and planktonic algae concentrations were found to affect fish through the food web and via seagrass cover which provides essential habitat for juveniles. Analyses of long-term trends show that on a regional scale, sea surface temperature in Bass Strait is important, especially in promoting catches of Snapper and King George Whiting.

Summary

Since the publication of the Western Port review (Keough et al. 2011), three high priority research projects on fish and their habitats have been completed. A field-based project investigated the specificity of fish-habitat relationships with implications for the management of marine habitats supporting fish populations in Western Port. Another high priority research project focussed on the implications of recreational fish harvesting in Western Port and was undertaken in two phases. Phase 1 utilised an extensive recreational fishing survey database to increase our understanding of fish biodiversity and habitat relationships, while phase 2 was a formal assessment of the status of the Western Port recreational fishery. The third major project addressed links between three key fish species and their habitats by analysing trends and change-points in historical data sets on fish catch statistics, fish growth, and environmental variation. This chapter provides a summary of the research outcomes from these projects, including their aims, results and management implications. The chapter also identifies further research needs.

Introduction

A major theme of the Western Port review was fish and their habitats (Jenkins 2011). Western Port is characterised by a high diversity and abundance of fish species, mainly related to the extensive and diverse habitats available. These habitats include benthic (sea bed) habitats such as seagrass, reefs, algae, invertebrate isolates (bryozoans), mangroves, unvegetated sand and mud habitats as well as the pelagic (water column) habitat (Jenkins 2011). The high productivity of fish in Western Port is of importance for higher order consumers such as birds and marine mammals (Dann 2011), as well as supporting a highly significant recreational fishery for key species such as King George Whiting, Snapper and Elephant Fish (Jenkins 2011).

The Western Port review recommended a number of fish research needs (Keough et al. 2011). The report assigned research needs to one of three priorities, ranked from highest (1) to lowest (3). Priority 1 research needs included two that were related to fish and fisheries: *Determine Linkages between fish and habitats* (research priority 28) and *Investigate the state of fish populations in Western Port and the effects of recreational fishing on fish stocks* (research priority 39). Since the publication of the review, three research projects have been completed that address these two research needs. These projects have included field studies on the specificity of relationships between fish and habitats (Jenkins et al. 2013, 2015), analysis of recreational fishing survey data to increase knowledge of fish ecology and biodiversity (Jenkins and Conron 2015) as well as assess the status of fish stocks (Conron et al. 2016), and analysis of historical fishing and environmental data sets to better understand links between fish populations and the Western Port environment (Morrongiello and Jenkins 2016).

Determining the specificity of fish-habitat relationships in Western Port

The Western Port review concluded that fish assemblages associated with *Zostera* seagrass and mangroves were relatively well studied, but assemblages associated with alternative habitats such as *Amphibolis* seagrass, the alga, *Caulerpa*, reef-macroalgae, rhodolith beds and sedentary invertebrate isolates were poorly known (Figure 7.1). Given the vulnerability of *Zostera* to declines in periods of adverse environmental conditions, it was important to understand the ability of fish species to utilise these alternative habitats, and therefore whether these habitats could act as a refuge habitat in the case of *Zostera* loss. The objectives of this study (Jenkins et al. 2013, 2015) were:

1) To determine the specificity of fish habitat relationships in Western Port.
2) To determine the resilience of fish populations to habitat loss through the use of alternative habitats.

A.
B.
C.
D.
E.

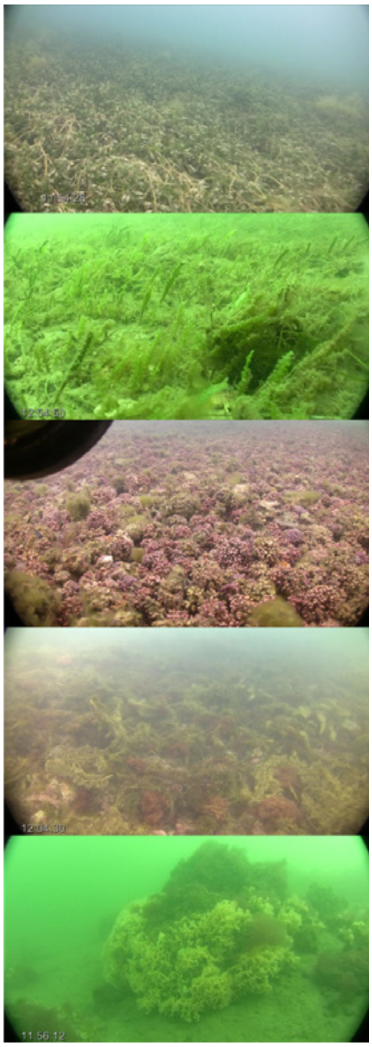

Figure 7.1 Habitats recorded by underwater survey in Western Port (Blake et al. 2012). A. *Amphibolis antarctica*, B. *Caulerpa cactoides*, C. Rhodolith bed, D. Reef with macroalgae, E. Invertebrate (Bryozoan) isolate.

Two primary methods were used for sampling alternative habitats. Underwater stereo video (Figure 7.2) was used to sample habitats with higher water clarity and in some cases high-relief bottom: *Amphibolis* and reef-macroalgae in the Western Entrance and sedentary invertebrate isolates and rhodolith beds in the Rhyll Segment. A mini otter trawl (Figure 7.3) was used to sample habitats with low water clarity and low relief bottom: *Caulerpa* habitat near the eastern coast of the Rhyll Segment and also *Caulerpa* habitat and a reference *Zostera* location north of Hastings in the Lower North Arm. Underwater video sampling in the Western Entrance segment was conducted in autumn and spring 2012, while the remaining sampling was conducted in spring 2012 and autumn 2013.

The results showed that species previously found to be common in *Zostera* seagrass were also found in *Caulerpa* habitat, and to a lesser extent in *Amphibolis* habitat. Most species were able to utilise different plant and algal habitats although some were more specific, such as the Weedy Seadragon (*Phyllopteryx taeniolatus*) that was only recorded in *Amphibolis* habitat. Multivariate analysis indicated that fish community structure was very similar between *Caulerpa* and previously published results for *Zostera*, as well as the reference sub-tidal *Zostera* location. One distinct difference, however, was much higher abundances of pipefish in *Zostera* habitat, supporting previous studies showing that syngnathids prefer seagrass over *Caulerpa* habitats. Although there was significant overlap of species amongst sub-tidal *Zostera* and the alternative vegetative habitats, this may not be the case for the fish associated with intertidal and shallow sub-tidal *Zostera* (which show differences from those in deeper sub-tidal *Zostera*) because these alternative habitats tend not occur at shallow depths.

The presence of many fish species in *Caulerpa* and *Amphibolis* habitat that have been previously recorded in *Zostera* habitat suggests that these habitats may provide a refuge for these species in the case of *Zostera* loss. However, evidence from commercial fish catches after the major *Zostera* decline in the mid-1970s suggests that species capable of utilising multiple habitats still showed significant population declines. This may be partly explained by the larger area of *Zostera* habitat relative to the alternative habitats. For example, habitat mapping in 1999 indicated that *Zostera* covered an area of approximately 100 km^2 compared with 20 km^2 for *Amphibolis* and < 10 km^2 for *Caulerpa* algae. Additionally, the depth and location of alternative habitats may not be as suitable for larval settlement as *Zostera* habitat but this requires further evaluation. Many of the juvenile fish in the alternative habitats may have initially settled in *Zostera* habitat. Thus, alternative habitats may provide some measure of resilience by providing a refuge for a low level of fish populations in the face of *Zostera* decline, however, they will not provide protection from major population declines, and also may not provide a refuge for species where larvae settle primarily in shallow habitats or in locations dominated by *Zostera*.

Figure 7.2 Deployment of stereo video camera.

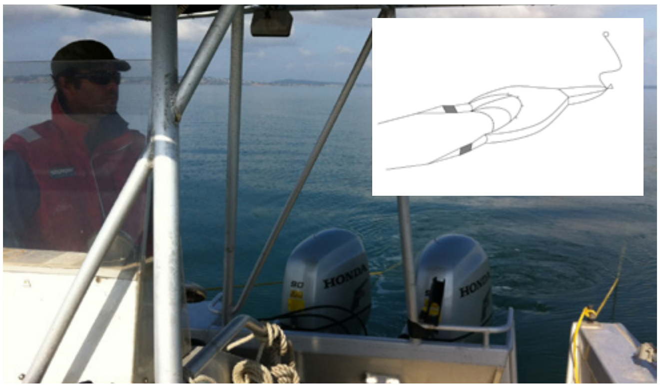

Figure 7.3 Mini otter trawl net deployed in Western Port.

In terms of biodiversity value, *Amphibolis* beds on the western coast of the Western Entrance segment were found have significant biodiversity value for fish, including the only populations of Weedy Seadragons recorded in the study. The *Caulerpa* beds on the eastern side of the Rhyll Segment were also found to have high fish species richness and abundance, indicating significant biodiversity value for fish. In contrast to *Amphibolis* and *Caulerpa* beds, very few fish species were recorded on rhodolith beds, suggesting this habitat has low value from the perspective of fish biodiversity.

The results in relation to fish biodiversity associated with sedentary invertebrate isolates in the deeper channels of the Southeast Basin were inconclusive due to poor visibility in the area. The relatively high turbidity and significant current movement would explain the prevalence of sedentary invertebrates as opposed to algae or seagrass in these deeper areas. Sampling of fish in this habitat is problematic because low visibility affects video sampling while trawl nets would become snagged on the large isolates (Figure 7.1). One potential sampling method for fish that could be used in the future is an acoustic sonar camera that creates high-quality video images that can define the outline, shape and even fins of target fish. Importantly, the technology is particularly effective in dark or turbid conditions where visibility is otherwise poor, such as in the deeper channels of Western Port.

The main conclusion of the study was that although some species previously recorded in *Zostera* habitat can also utilise alternative habitats, *Zostera* habitat is nevertheless the most critical for fish biodiversity in Western Port because of its extensive spatial cover and unique role for larval settlement/development in shallow areas, as well as supporting some unique species, in particular, pipefish and seahorse species. Thus, although alternative habitats provide a potential refuge for older juveniles and adults of some fish species in the event of *Zostera* loss, the maintenance of fish biodiversity in Western Port relies on the persistence of significant areas of *Zostera*, particularly in the intertidal, shallow sub-tidal zone. The key findings of the study in relation to the major habitats are summarised below (Table 7.1).

Characterising the status of the Western Port recreational fishery in relation to biodiversity values

This project was divided into two phases: Phase 1 was an analysis of the extensive recreational fishing survey data from Western Port to better understand the biodiversity of key fish species and their habitats (Jenkins & Conron 2015), while phase 2 was a formal stock assessment of the Western Port Fishery (Conron et al. 2016). The project was carried out in collaboration with Fisheries Victoria.

Table 7.1. Summary of key findings in relation to major habitats (Project 1)

Zostera seagrass:

- Major fish habitat in Western Port covering approximately 100 km^2.
- High fish species richness, dominant species include the Spotted Pipefish, Grass Whiting, Little Weed Whiting and leatherjackets (Acanthaluteres).
- Unique species include a diverse assemblage of conservation listed syngnathid species (pipefish and seahorse).
- Occurs in intertidal and shallow subtidal areas where it provides habitat for settling larvae of key species such as King George Whiting.

Amphibolis seagrass:

- Main fish habitat in the western entrance area of Western Port covering approximately 20 km^2.
- High fish species richness, dominant species include the Sixspine Leatherjacket, Little Weed Whiting, Weedy Seadragon, and leatherjackets.
- Unique species include the conservation listed Weedy Seadragon.
- Habitat for economically important species such as Calamari and King George Whiting.

Caulerpa algae:

- Important fish habitat on the eastern coast of the Rhyll segment of Western Port covering < 10 km^2.
- High fish species richness, dominant species include the Cobbler, Wood's Siphonfish, leatherjackets, and Southern Pigmy leatherjacket.
- *Caulerpa* occurs deeper than *Zostera*, with mostly older juvenile or adult fish.
- Habitat for economically important Rock Flathead and Sand Flathead.

Reef/algae:

- Small area of habitat mostly in the entrance areas of Western Port.
- High fish species richness, dominant species include the Bluethroat Wrasse, Silver Trevally, Toothbrush Leatherjacket, and Sixspine Leatherjacket.
- Fish generally larger than in low-relief *Amphibolis* habitat in the same area.
- Habitat for economically important species such as Silver Trevally and Australian Salmon.

Rhodolith beds:

- Small area of habitat occurring immediately inside the eastern entrance of Western Port.
- Low fish species richness, common species were the Red Mullet and Smooth Toadfish.
- Generally appears to be of limited value as fish habitat.
- Low diversity and abundance of fish may relate to low habitat complexity.

Phase 1

Recreational fishing research data obtained from boat ramp interviews has detailed information on numbers and lengths of species caught, as well as location, depth and habitat of capture. In this study, data collected over a 15-year period from 1998 to 2013 was analysed with a view to increasing knowledge on the ecology and biodiversity of key fish species in Western Port. The results also provided base-line information for a stock assessment of important recreational fishing species in Western Port that was conducted in Phase 2 of the project.

Interviews with boat-based fishers returning from fishing trips were conducted by Fisheries Victoria on weekends from approximately October - November to April - June each year. Nearly 11,000 interviews were conducted at ten ramps, with most information coming from Corinella, Cowes, Hastings, Newhaven, Rhyll, Stony Point, Tooradin and Warneet. Information provided included number of fishers, hours fished, fisher avidity (i.e. frequency of fishing trips), fishing method/bait, species caught and released, and fish length. The information also included the area fished based on the catch cells (areas) previously used for commercial log book recording.

The spatial distribution of catch rates (an indicator of abundance) was visualised using Geographical Information System (GIS) mapping for key species. Spatial information was supplemented with data on habitat and depth fished. Some species, such as King George Whiting, Southern Calamari and Southern Sea Garfish had higher catch rates (indicating greater abundance) in areas of higher seagrass cover. An example of this is shown for King George Whiting in Figure 7.4. Fishing for these species tended to be in relatively shallow depths and habitats that included seagrass. In contrast, species such as Snapper and Gummy Shark had higher catch rates in the deeper reef habitats of the Western Entrance Segment and the Lower North Arm. An area of high catch rates for most species was the Rhyll Segment, a broad subtidal sedimentary plain with habitats such as seagrass, macroalgae and sedentary invertebrate isolates. The Rhyll Segment is also strongly influenced by water quality and sedimentation entering the northeast of the bay from the catchment, so catchment management to maintain water quality entering the bay is likely to be critical to maintaining fish biodiversity and sustaining recreational fishing in the bay.

In terms of changes to catch rates and length distributions over the survey period, there was a common pattern for several species of strong fluctuations at the scale of a few years. For species such as King George Whiting and Snapper, research has shown that these fluctuations are related to variability in recruitment that is driven by environmental fluctuations. Long term trends were also evident for some species across the survey period. Snapper showed an increasing trend that was most likely related to a series of successful recruitment years in Port Phillip in the 2000's following poor recruitment in the 1990s. Flathead showed a slightly decreasing trend in catch rate that may be related to the much more significant decrease in Sand Flathead catch rates in Port Phillip over the same period. This decline is also thought to be mainly driven by a period of poor recruitment related to environmental conditions. An area of uncertainty in this analysis is the extent to which Snapper and Flathead spawn inside Western Port as opposed to immigrating into Western Port after spawning in Port Phillip. Plankton sampling for fish eggs and larvae in Western Port is recommended to determine the extent of spawning by these species within the bay.

Although catch rates of Elephant Fish were relatively stable across the survey period, a contraction of the spatial distribution in the catch rates to the Rhyll Segment may be a cause for concern because decline in the population is masked by increased aggregation. Further research on the biology of Elephant Fish in Western Port would be valuable to understand the major population fluctuations of this species in the bay. In particular, understanding more about the key habitats in the Rhyll Segment for Elephant Fish spawning and juveniles would be of benefit to the management of the species and its supporting habitats.

Overall the study provided new information on the spatial distribution and habitat use of important fish populations in Western Port that will inform management of the marine environment in relation to catchment inputs, coastal development, recreational fishing and marine protected areas. The results suggested that variation in catches by recreational fishers was primarily influenced by the environmental drivers of recruitment of young fish to the Western Port ecosystem.

Phase 2

In phase 2, a formal assessment of the Western Port fishery was undertaken at Hastings in August 2015 and this was followed by the publication of a fishery assessment report.

The assessment workshop was attended by:

- Representatives from recreational and commercial fishing sectors
- Fisheries Victoria managers, scientists and compliance officers
- Catchment management, university and conservation representatives.

The assessment process used a weight-of-evidence approach that, for the first time, was based solely on recreational fishery data, including trends in catch rates, catch size structures, pre-recruit surveys, effort and social indicators.

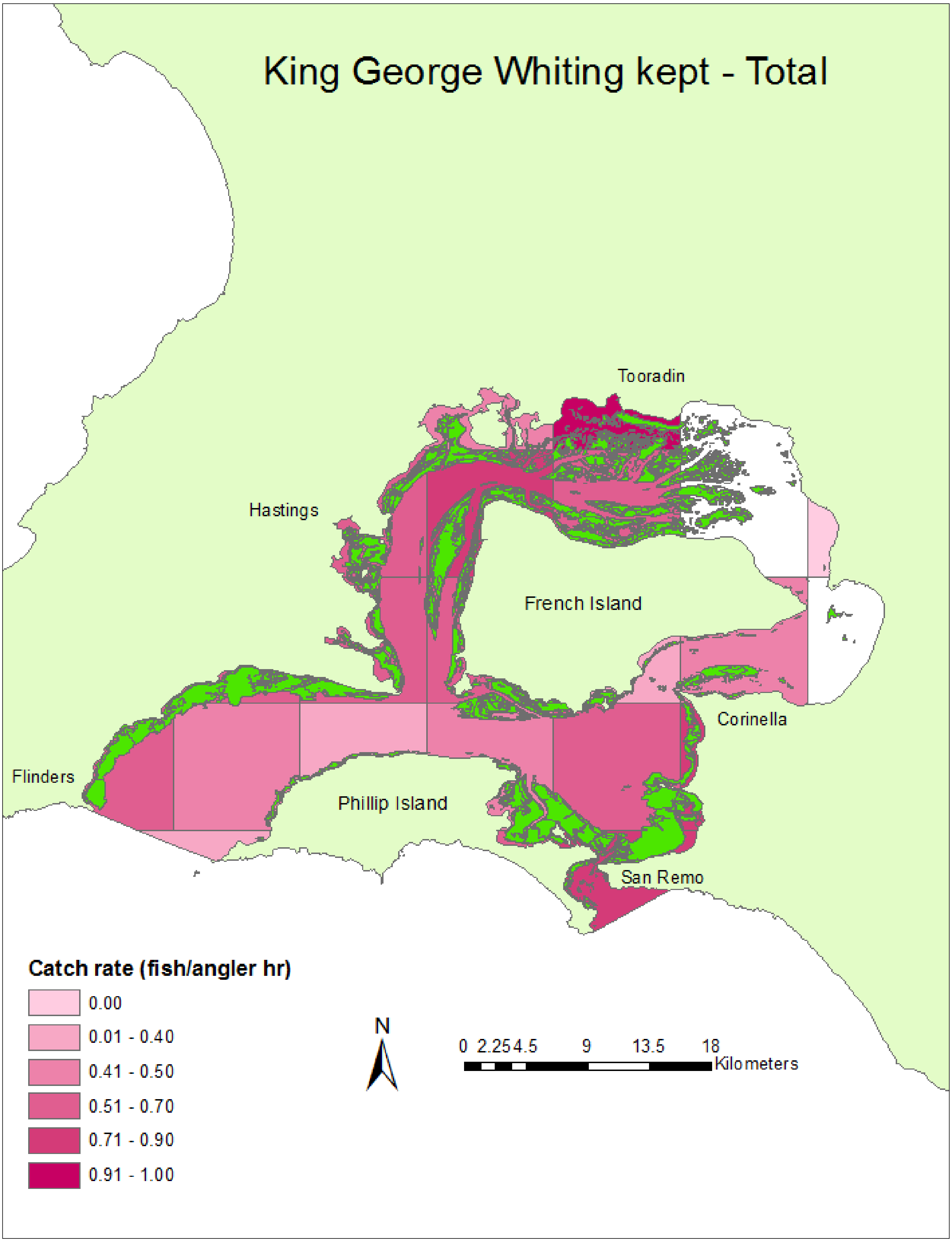

Figure 7.4 Catch rate of King George Whiting (number caught per fisher hour) by area with the distribution of seagrass overlaid on the map

Status of the Western Port fishery

The workshop considered the status of key recreational fishing species and made the following assessments.

King George Whiting

The recent trend in recreational catch rates for King George Whiting has been upwards even though the most recent year (2014/15) was below average. Recent higher catch rates correspond to good recruitment of post-larvae in the Port Phillip pre-recruit survey in 2007 and 2008. Length-frequency data is also consistent with a group of larger fish moving through the fishery. Although declining in the most recent year, catch rates are predicted to improve again due to above average abundances of pre-recruits in 2013. Overall, based on pre-recruit and catch rate indicators the stock is considered to be improving.

Snapper

The recent trend for recreational catch rates for Snapper has been stable even though the most recent year (2014/15) was below average. Pre-recruit surveys in Port Phillip show above average abundances of 0+ age Snapper in 2012/13 and 2013/14 that should support stable to increasing catch rates of Snapper in Western Port in the coming years. Length-frequency data is within historic variation; as yet there is little evidence of the small fish coming through as a result of the recent strong recruitment in Port Phillip. Overall, based on pre-recruit and catch rate indicators the stock is considered to be stable.

Flathead

The recent trend for recreational catch rates for Flathead has been stable even though the most recent year (2014/15) was below average. The recent stable period comes after a longer term decline that mirrors the decline in Port Phillip (thought to be due to environmental factors). Length-frequency data is within historic variation; and there is evidence of undersize fish from good recruitment in 2011/12 that may support the fishery in coming years. Overall, based on catch rate and size-frequency indicators the stock is considered to be stable.

Gummy Shark

The recent trend for recreational catch rates for Gummy Shark has been increasing and the most recent year (2014/15) was above average. This upward trend comes after low catch rates in the mid 2000's, particularly in evidence in the angler diary data. Interpretation of catch rates for Gummy Sharks is complicated by the low daily bag limit of two Sharks. Length-frequency data for Gummy Shark is within the historic range. Overall, based on catch rate and size-frequency indicators the stock is considered to be in good condition.

Elephant Fish

The recent trend for recreational catch rates for Elephant Fish has been downward and the most recent year (2014/15) was well below average. There is a limitation to interpreting this data, however, because the daily bag limit was reduced from 3 to 1 in 2008. Data on length-frequency was also limited by reduced targeting and low catches. This includes evidence of a contraction in the area where Elephant Fish are caught to the Rhyll Basin, and the opinion of expert fishers that catch was already in sharp decline before the new bag limit. Overall, the assessment is considered to be data limited but the weight of evidence supports a significant decline in the fishery. Prior to the 1980s there were also very low abundances of Elephant Fish in Western Port and this long-term variation may be linked to environmental change (e.g. seagrass cover).

Satisfaction and Perception

A social survey of satisfaction and perception amongst Western Port anglers in 2014/15 indicated that the primary motivations for fishing were 'enjoyment of the sport' and 'fish for food'. Over 80% of anglers were either 'very satisfied' or 'somewhat satisfied'. The main issue for those who were not satisfied was 'lack of fish' (caused by 'bad season/weather') and to a lesser extent 'boat ramp busy/no parking' (caused by 'lack of facilities' and 'too many boats').

Fishing Effort

Fishing effort based on standardised trailer counts at ramps shows relatively stable effort in recent years (approximately 25 trailers per ramp) with increased numbers in 2014/15 (approximately 35 trailers per ramp). Trailer counts in recent years have been lower than in the early to mid-2000's, particularly for the western ramps. These trends in fishing effort tend to be correlated with catch rates, that is, higher catch rates lead to greater fishing effort. The interpretation of the data is limited by the fact that counts are only undertaken on fine, weekend days.

Management arrangements

Fishery data presented at the August 2015 stock assessment workshop did not indicate the need for a review of fishery management arrangements, and participants supported maintaining the current management regime.

Understanding Western Port's past to manage its future: Investigating the drivers of long-term change in key biological systems

This project further addressed the priority 1 research need: *Determine Linkages between fish and habitats* by bringing together historical time series of fish populations and environmental factors in sophisticated analyses to determine the drivers of long term change in key fish species.

Coastal and estuarine environments provide a range of valuable ecosystem services, including supporting commercial and recreational fisheries, nursery habitats for marine species, and filtering and detoxification services. These systems have, however, already been impacted by human activities, which in conjunction with climate change will continue to threaten the condition of these environments. For many coastal and estuarine systems we currently lack the knowledge to link particular environmental drivers or events to observed biological changes. This understanding is essential if we are to sustainably manage coastal and estuarine environments and protect their valuable ecosystem services in a changing world.

Although Western Port is in relatively good condition, we have a limited understanding of how its ecology responds to a range of local and regional environmental drivers. Developing such insight is of relevance to Melbourne Water and other local natural resource managers, as it would help them manage the surrounding catchments, creeks and major drainage systems to promote Western Port's ecosystem health.

In this project, the drivers of long-term change in key Western Port fisheries were investigated to inform future management (Morrongiello and Jenkins 2016). Firstly, we developed a series of conceptual models illustrating hypothesised links between environmental factors and the life history of three key fisheries species, Snapper, King George Whiting and Elephant Fish. An example of these conceptual models is shown for King George Whiting (Figure 7.5). Secondly, we collated a database of fisheries information, including commercial and recreational catch records, recruitment indices and two novel growth time series (using otoliths). Thirdly, we used the conceptual models to identify potentially important local and regional environmental drivers. Finally, we performed a series of statistical analyses on the long term data to explore whether there were any similarities in species abundances through time, identify any distinct changes, and then relate major trends to environmental conditions.

We curated 13 biological time series spanning the last 100 years in Western Port, including the development of two new growth chronologies for King George Whiting and Snapper (as an example, the growth chronology for King George Whiting is shown in Figure 7.6). We successfully reduced this data into three common trends, identifying responses across different species and aspects of life history. Step changes (change points) in these trends were predominantly associated with El Niño and La Niña events, and to a lesser extent recruitment pulses and cessation of commercial netting.

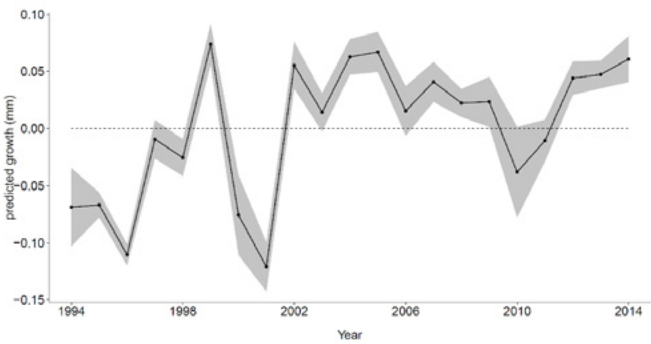

Figure 7.6 Novel growth time series for King George Whiting in Western Port based on otolith growth increment chronology

The three common trends were associated with both local and regional drivers. On a local scale, nitrogen loads and Chlorophyll *a* (an indicator of the amount of algae in the water column) concentrations affected fish through the food web and via seagrass cover which provides essential habitat for juveniles. On a regional scale, sea surface temperature in Bass Strait was important, especially in promoting catches of Snapper and King George Whiting. Further details of the influence of drivers for these trends can be found in Morrongiello and Jenkins (2016).

This research could be readily expanded to include biological data such as long-term bird counts and catch data for other fish species, and additional environmental variables such as seagrass cover to provide a more holistic view of the drivers of long-term biological change in Western Port's marine environment.

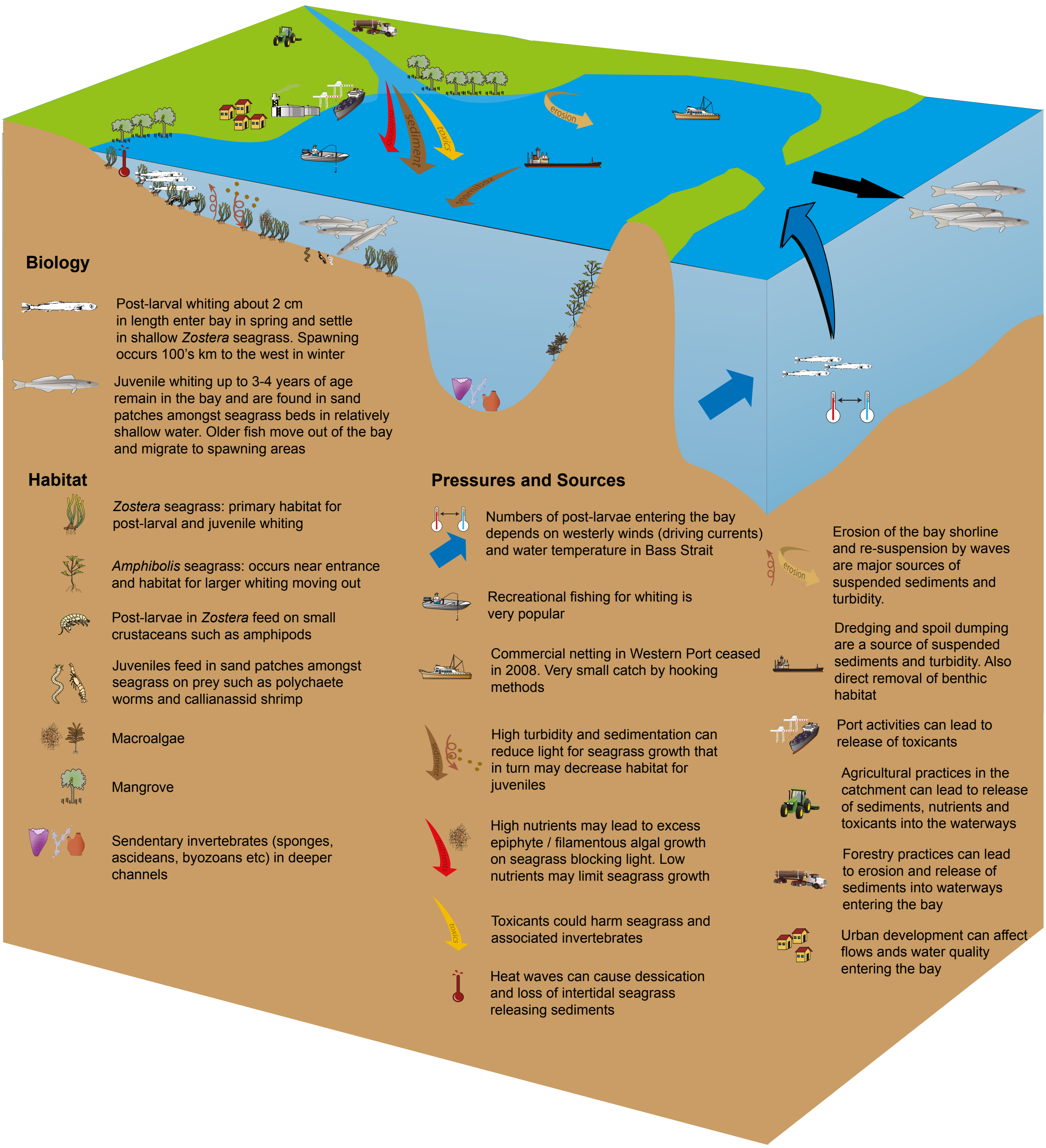

Figure 7.5 Conceptual model of King George whiting life history and important environmental drivers

Future directions and opportunities

Research into the status of Western Port fish stocks

7.1 Although data is limited, there is significant evidence of a continued decline in the Elephant Fish fishery. It would be prudent to review the status of the larger offshore stock for comparison.

In species such as Gummy Shark and Elephant Fish with very low bag limits there is a difficulty in interpreting catch rates, and collection of detailed data on discards is recommended so that catch rate can be estimated more accurately. Confusion over the use of 'partial length' to measure Gummy Sharks is also a cause for concern as it can lead to the retention of undersize sharks.

7.2 Pre-recruit surveys for King George Whiting in Western Port are required, rather than relying on results from Port Phillip surveys. This will provide a much clearer picture of the status and trajectory of the Western Port populations, and improve our ability to identify local processes and threats.

Fish life histories and habitat relationships

7.3 A research need that has yet to be addressed from the Western Port review is *Determining the species, locations and timing of fish spawning* (research priority 33) (Keough et al. 2011).

Little is known about the Western Port in relation to fish spawning, including the extent of spawning of key species such as Snapper and Flathead. Only limited egg and larval sampling has been conducted from the southern part of the bay in the past. Fish eggs and larvae are the life stage most sensitive to exposure to poor water quality, and therefore understanding the spawning activities of fish in the northern part of Western Port is a key catchment management issue. Further, the distribution and abundance of fish eggs and larvae is an important aspect of biodiversity that is poorly understood.

Finally, interpretation of trends in the fishery, as emphasised in the recreational fishing research (project 2 above), depends strongly on interpreting recruitment patterns of young fish. At present this information comes from surveys of recruitment of key species in Port Phillip, but interpretation is hindered by a lack of knowledge of whether the same species also spawn in Western Port.

A fish egg and larval sampling program would also contribute another priority 2 research need from the Western Port review:

7.4 'Investigate the marine and estuarine requirements of the listed Australian Grayling' (research priority 32) (Keough et al. 2011). Fish larval sampling could potentially provide information on the marine larval stage of this species within Western Port.

7.5 Quantify the fish-habitat relationships for the young stages of Elephant Fish.

An area of uncertainty in the understanding of fish-habitat relationships is for the young stages of Elephant Fish (project 2 above). Anecdotal evidence suggests female Elephant Fish may lay eggs in bare silty sediments near seagrass beds while young juveniles (neonates) are thought to utilise seagrass habitat. There is no quantitative information on these relationships and field surveys are required to provide the necessary evidence on habitat use for spawning and early juvenile life of this species.

7.6 Investigate fish communities associated with sedentary invertebrate isolates.

In the fish habitat project we found that surveying the fish communities associated with the sedentary invertebrate isolates that occur extensively in the deeper channels of Western Port is not practical using typical sampling techniques as nets will become snagged while high turbidity and low light reduce the effectiveness of underwater video. Acoustic sonar camera techniques may be one option to survey fish in these habitats in the future.

Trends and change points

7.7 Further investigate drivers of long-term biological change by including additional biological data (e.g. birds and other fish species) and environmental variables.

Research into drivers of long-term change provided a proof of concept for analysing trends and change points in historical data for three key fish species and associated environmental variables. The analysis, however, could be significantly expanded to include biological data such as that from long-term bird counts and catch data for other fish species available from Western Port, and additional environmental variables such as seagrass cover and turbidity (using recently developed remote sensing analysis techniques by CSIRO) to provide a more holistic view of the drivers of long-term biological change in Western Port's marine environment. The inclusion of additional environmental variables that have a clear link to management needs (e.g. turbidity and seagrass cover) will be of particular value.

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8 Population trends in waterbirds in Western Port: what do they tell us?

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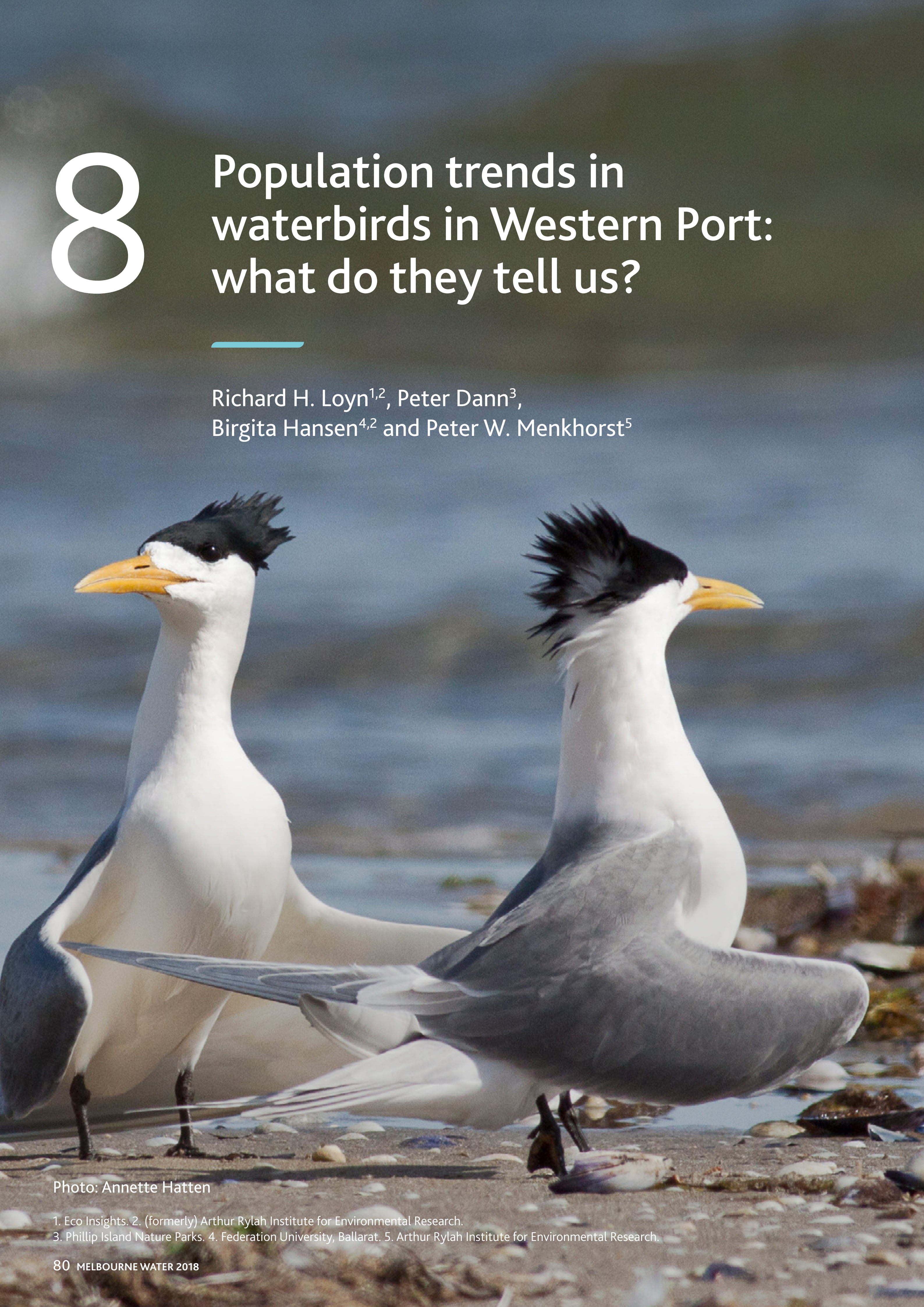


Photo: Annette Hatten

1. Eco Insights. 2. (formerly) Arthur Rylah Institute for Environmental Research.
3. Phillip Island Nature Parks. 4. Federation University, Ballarat. 5. Arthur Rylah Institute for Environmental Research.

Research Priorities

This chapter addresses the following research priorities identified in the Western Port review (Keough et al. 2011):

- Examine the trends of fish-eating birds in Western Port and Corner Inlet. (research priority 35).
- Effects of sea-level rise on shorebirds (research priority 42).

This chapter also contributes to a broader picture of ecological changes in different parts of Western Port, some mediated by local events and others by events operating on a global scale. Some of these research priorities were addressed through the Western Port Welcomes Waterbirds project (Hansen et al. 2011).

Key Findings

- Population trends were determined for 39 of the 85 observed waterbird species (excluding seabirds). Populations of 22 waterbird species in Western Port declined between 1973-2015, 15 species remained stable (despite fluctuations and some changes in distribution) and two of the 39 species have increased.
- The main declines were associated with trans-equatorial migratory shorebirds. These declines may be due to habitat loss on their migratory flyways in east Asia, particularly the Yellow Sea.
- Fish-eating terns, cormorants and pelicans have decreased in Western Port (and increased in West Corner Inlet). Little Pied Cormorant decreased in the late 1970s and early 1980s, in association with seagrass dieback in Western Port. Crested Tern decreased later in the 1980s and 1990s, to a greater extent than other fish-eating species and despite establishing a large new breeding colony at the bay entrance on the Nobbies (Phillip Island). Numbers of two smaller and less numerous tern species (Fairy Tern and Little Tern) declined at the same time.
- Crested Terns, Little Terns and Fairy Terns have made less use of the bay since a decline in small fish and a larger predatory fish (i.e. Australian Salmon that drive small fish to surface waters where terns feed). The decline of threatened Fairy Terns is of particular conservation concern because (unlike Crested Terns) no redistribution has been observed for Fairy Terns and they now breed only intermittently in the bay.
- Several species declined in the central-east part of the bay along with major loss of seagrass. That area is now consistently used by four waterbird species that were formerly rare in the bay (Red-necked Avocet, Banded Stilt, Whiskered Tern and Gull-billed Tern), suggesting a local switch to a new type of habitat (open mudflats and shallow waters without seagrass).
- Black Swans form 69% of the waterbird biomass in the survey area, and may be useful as a highly visible indicator of seagrass distribution and health.
- Despite declines in some waterbird species over the last two decades, Western Port continues to be an extremely important habitat for waterbirds and attention should be given to the needs of all species so that they continue to thrive.

Summary

Waterbirds have been counted in Western Port at least three times per year since 1973 in a co-ordinated citizen science survey run by BirdLife Australia (formerly Bird Observers Club of Australia). Surveys have focused on strategic sites including high-tide roosts and associated stretches of coast and nearby wetlands. Data have been analysed at various times, most recently through three projects commissioned by the Central Coastal Board and Melbourne Water. This paper summarises the results of those three projects.

Many waterbird species have declined over the 43 years of the survey (22 species from 39 analysed), but a few have increased and many have remained stable (despite fluctuations and some changes in distribution). The main declines have been among trans-equatorial migratory shorebirds, perhaps because of habitat loss in the Yellow Sea, east Asia. Crested Terns, Little Terns and Fairy Terns have made less use of the bay since a decline in small fish and a larger predatory fish (Australian Salmon): possibly because the latter drive small fish to surface waters where terns feed. Crested Terns have established a large breeding colony on the Nobbies, Phillip Island, with most resident birds now feeding outside the bay. The decline of Fairy Terns remains a serious conservation concern. Several species declined early in the survey with major loss of seagrass, especially in the central-east part of the bay. That area has now been colonised by four waterbird species that were formerly rare in the bay (Red-neck Avocet, Banded Stilt, Whiskered Tern and Gull-billed Tern), suggesting a local switch to a new type of ecosystem (open mudflats and waters without seagrass). Australian Pied Oystercatchers using tidal mudflats have increased steadily, and two species that feed in pasture, Cape Barren Goose and Straw-necked Ibis, have also increased. Black Swans form 69% of the waterbird biomass in the survey area, and may be useful as a highly visible indicator of seagrass abundance; seagrass is their main food on tidal mudflats, and they are the only bird to feed extensively on seagrass in this area.

The survey highlights the value of a long-running citizen science project, especially when intermittently funded projects such as those discussed here allow extra value to be extracted from the data.

Introduction

Western Port covers 680 km^2 with 270 km^2 of tidal mudflat and 263 km of coast, of which 107 km is lined by mangroves *Avicennia marina* (Shapiro 1975). It is renowned as a habitat for waterbirds, attracting large numbers of species that feed from the extensive mudflats. In recognition of these and other values, most of the bay was declared a Ramsar-listed wetland, and now forms part of the Western Port Biosphere Reserve.

Numbers of waterbirds have been monitored in Western Port since 1973 as part of a citizen science project now run by BirdLife Australia (previously the Bird Observers Club of Australia). This is the longest-running program of its kind in Australia. It was initiated to provide data on waterbirds for the Westernport Bay Environmental Study (Shapiro 1975), and focused on birds that made use of the extensive intertidal mudflats in the shallow northern and eastern parts of the bay from Sandy Point (near Somers) round French Island and extending to Observation Point near Rhyll on Phillip Island. Hence it included most of the bay except the exposed south-western arm and the southern coasts of Phillip Island and the Mornington Peninsula facing Bass Strait.

The survey located the major high-tide roosts around the bay and teams of observers counted birds at each of the main roosts (simultaneously where possible) and associated stretches of coast on selected days each year (Loyn 1975, 1978; Dann et al. 1994, Loyn et al. 1994, 2001; Heislers 2003, Chambers and Loyn 2006, Hansen et al. 2011, 2015). Over the whole study, 11 sites (or groups of sites) were covered on most counts; eight other sites were counted as often as possible and another 14 sites were counted intermittently or discontinued.

In recent years, Victorian government agencies have funded three studies examining long-term trends, based on this dataset, two of which have been commissioned by Melbourne Water as part of its Western Port Environment Research Program. The Western Port review (Keough et al. 2011) placed a high priority on research to *Examine the trends of fish-eating birds in Western Port and Corner Inlet*. The resulting project examined trends in fish-eating birds in Western Port from 1974 to 2012 and West Corner Inlet from 1987 to 2012, recognising that numbers of these birds have been reported to have declined in Western Port, and may act as useful indicators of trends in fish stocks (Menkhorst et al. 2015). Another project collated data on Black Swans, *Cygnus atratus*, from 1973-2015 (Loyn 2016), with the aim of correlating the data with information on seagrass distribution, and is currently being analysed for Melbourne Water. The third study, conducted by Arthur Rylah Institute on behalf of the Central Coastal Board, was a broad-ranging investigation looking at trends in the data from 1974 to 2009 and comparing them with trends observed nationally and internationally (Hansen et al. 2011, 2015).

This paper summarises these recent studies and places them in the context of the broader set of data collected over more than forty years. The long-term monitoring program is a model for long-term citizen science and emphatically illustrates the value of long-term data and citizen science projects.

Long-term monitoring

Counts of waterbirds were made at the main sites on the selected date: monthly from 1973-75, five times per year to 1994, then three times per year from 1994. Main survey sites are listed in Table 8.1. Various additional sites were counted less frequently. All waterbird species were counted, including cormorants, pelicans, grebes, ducks, swans, gulls, terns, shorebirds (waders), ibis, spoonbills, herons, egrets and coots. The five seasonal visits were in late summer (January or February), autumn (April or May), winter (June, July or early August), spring (September or October) and early summer (November or December). The spring and autumn counts were discontinued in 1994.

On each count, teams of observers counted all waterbirds by species at the designated site, including birds at high-tide roosts and nearby stretches of coast. If particular sites could not be visited on the appointed day (e.g. if high winds prevented boat crossings) they were counted soon afterwards if possible.

Various subsets of the data have been analysed for different purposes and to answer key questions including population trends over time. The trend analysis considered data from the main sites from 1974-2009 (Hansen et al. 2011, 2015). The study of fish-eating birds (Menkhorst et al. 2015) focused on counts from two seasons (late summer and winter) because comparable data were available for those seasons from West Corner Inlet, which was seen as a useful benchmark having less human disturbance. It examined trends over 38 years from 1974 to 2012 in Western Port and 25 years from 1987 to 2012 in West Corner Inlet, and related those trends to data on commercial fish catch per unit effort supplied by Fisheries Victoria. The study of Black Swans (Loyn 2016) considered all data from 1973-2015, from 33 sites. Analyses were done to test the influence of location within the bay and season over five-year periods for 11 main sites counted on more than 120 dates. Correlations with seagrass will be examined for a subset of years when suitable satellite imagery data on seagrass distribution becomes available from the CSIRO project (discussed in Chapter 2 of this document).

These results refer to total numbers of waterbirds, and total numbers of selected species, across all sites counted, with no allowance for sites missed on particular counts.

Table 8.1 Sites counted regularly for BLA Western Port waterbird survey.

The main focus is on high-tide roosts, but Hanns Inlet is counted at low tide.

Site group	Region and location
Hanns Inlet (& formerly Sandy Pt)	SW, Mainland
Barralliar Is and nearby reefs	NW, French Is.
North-west French Island (Bullock & Decoy Swamps etc)	NW, French Is.
Yallock Ck (formerly also Bunyip River)	NE, Mainland
Stockyard Pt (Jam Jerrup)	NE, Mainland
North Pioneer Bay (Red Bluff Ck/GMH drain/Tramcar coast)	NE, Mainland
Central Pioneer Bay (Mottons/Blackneys Rd/Grantville)	NE, Mainland
Reef Is and nearby parts of Bass Bay	SE, Mainland
Observation Pt (also beach, estuary)	SE, Phillip Island
Rams Is (and nearby parts of Bass Bay)	SE, French Is.
Tortoise Head (and saltmarsh, coast from Tankerton)	SW, French Is.
Fairhaven (and saltmarsh, coast from Tankerton)	SW, French Is.

During the 168 survey-sessions, 85 waterbird species were recorded on a regular or occasional basis, along with several others recorded as vagrants or seabirds seen in the more exposed ocean-facing parts of the bay. The 30 most common species, and others for which Western Port provides important habitat, are listed in Table 8.2 and 8.3, along with their abundance in the survey area (mean and maximum counts) and basic information about their needs for habitat, food, nesting and migration.

Table 8.2 Mean and maximum counts of waterbirds at all sites counted on the BirdLife Australia Western Port survey, also showing main foods and habitats used for feeding, roosting and nesting by each species. The table includes the 30 most numerous species, plus a few less common species for which the bay may be important. A key to symbols is given at the foot of the table.

Species	Scientific name	Mean total count	Max count (all sites)	Main food	Main feeding habitat	Main roosting habitat	Main nesting locations
Musk Duck	*Biziura lobata*	32	264	IV	SW	SW	wetlands mainly inland
Cape Barren Goose	*Cereopsis novaehollandiae*	1.8	92	P(L)	P	W, P	Phillip Is & French Is
Black Swan	*Cygnus atratus*	1974	10506	P	SW, SM	SW, SM	local wetlands & saltmarsh
Australian Shelduck	*Tadorna tadornoides*	123	1576	IP	SW	SM	saltmarsh & trees southern Vic
Grey Teal	*Anas gracilis*	150	2835	IP	SW	SW	inland Australian wetlands
Chestnut Teal	*Anas castanea*	463	2887	IP	SW	SW	local wetlands
Pacific Black Duck	*Anas superciliosa*	42	428	IP	SFW	SW	local wetlands
Hoary-headed Grebe	*Poliocephalus poliocephalus*	24.0	205	IF	SW	SW	inland Australian wetlands
Little Pied Cormorant	*Phalacrocorax melanoleucos*	151	844	FV	SW	M, BI, J, T	trees in local wetlands
Pied Cormorant	*Phalacrocorax varius*	59	230	F	SW	M, BI, J, T	Mud Is; local wetlands & previously mangroves
Australian Pelican	*Pelecanus conspicillatus*	67	253	FD	SW	BI, U, J	Duck Splash (French Is) & Mud Is
Great Egret	*Ardea alba*	15.6	92	FV	TM, W	M, J, T	inland Australian wetlands
White-faced Heron	*Egretta novaehollandiae*	169	1188	FV	TM, SM, W, P	SM, M, T	trees near local wetlands
Little Egret	*Egretta garzetta*	0.3	7	FV	TM	M, R, BI	wetlands mainly inland (also Mud Is & Corio)
Australian White Ibis	*Threskiornis molucca*	511	1807	I	TM, P, U	M, SM, T	local wetlands, Mud Is
Straw-necked Ibis	*Threskiornis spinicollis*	159	2812	I	P	W, T	trees in local wetlands, Mud Is
Royal Spoonbill	*Platalea regia*	88	395	IF	TM, W	W, P, T	trees in local wetlands
Yellow-billed Spoonbill	*Platalea flavipes*	1.1	70	IF	SFW	W, P, T	wetlands mainly inland
Purple Swamphen	*Porphyrio porphyrio*	10.6	80	P(W)	AV	W	local wetlands
Eurasian Coot	*Fulica atra*	10.4	281	P(W) FA	SFW	W	wetlands mainly inland
Australian Pied Oystercatcher	*Haematopus longirostris*	194	503	I	TM	BI, M	low vegetation behind local beaches
Red-necked Avocet	*Recurvirostra novaehollandiae*	61	850	I	SW	BI, SW	inland Australian wetlands
Banded Stilt	*Cladorhynchus leucocephalus*	2.8	180	I	SW	SW	inland Australian wetlands
Pacific Golden Plover	*Pluvialis fulva*	24.6	217	I	TM	M, R, BI	Arctic tundra
Red-capped Plover	*Charadrius ruficapillus*	95	466	I	TM, W	BI	local beaches; inland wetlands with broad shores
Double-banded Plover	*Charadrius bicinctus*	212	1172	I	TM, P	BI, SM	New Zealand braided rivers

Species	Scientific name	Mean total count	Max count (all sites)	Main food	Main feeding habitat	Main roosting habitat	Main nesting locations
Lesser Sand Plover	*Charadrius mongolus*	2.2	42 #	I	TM	BI	Asian semi-deserts
Greater Sand Plover	*Charadrius leschenaultii*	0.6	4	I	TM	BI	Asian semi-deserts
Hooded Plover	*Thinornis rubricollis*	1.1	13	I	B	BI	ocean-facing beaches
Masked Lapwing	*Vanellus miles*	242	778	I	TM, P	SM	local grassland
Bar-tailed Godwit	*Limosa lapponica*	228	607 #	I	TM	BI	Arctic tundra
Whimbrel	*Numenius phaeopus*	12.5	143 #	I	TM	BI, R	Arctic tundra
Eastern Curlew	*Numenius madagascariensis*	617	2251	I	TM	SM	Arctic tundra
Terek Sandpiper	*Xenus cinereus*	1.9	14	I	TM	M, R, BI	sub-Arctic rivers
Common Sandpiper	*Actitis hypoleucos*	0.3	3	I	TM (narrow creeks)	J	sub-Arctic rivers & streams
Grey-tailed Tattler	*Heteroscelus brevipes*	10.3	80	I	TM	M, R, BI	Arctic mountain streams
Common Greenshank	*Tringa nebularia*	85	372	IF	TM, W	W	north Asian lakes
Ruddy Turnstone	*Arenaria interpres*	50	211	I	TM	R, BI	Arctic tundra
Red Knot	*Calidris canutus*	41	571	I	TM	BI	Arctic tundra
Red-necked Stint	*Calidris ruficollis*	3317	12608	I	TM, W	BI, W	Arctic tundra
Sharp-tailed Sandpiper	*Calidris acuminata*	173	1856	I	TM, SM, W	BI, SM, W	Arctic tundra
Curlew Sandpiper	*Calidris ferruginea*	1744	7098	I	TM, W	BI, W	Arctic tundra
Fairy Tern	*Sternula nereis*	16.4	128	F	SW	BI	islands south of French Is; Mud Is, etc
Gull-billed Tern	*Gelochelidon nilotica*	7.2	126	IF	SW, TM	BI	inland Australian wetlands; also SE Asia
Caspian Tern	*Hydroprogne caspia*	19.9	61	F	SW	BI	islands south of French Is; Mud Is, etc; also inland
Whiskered Tern	*Chlidonias hybridus*	4.0	230	IF	SW	BI	inland Australian wetlands; probably also SE Asia
Crested Tern	*Thalasseus bergii*	118	870	F	SW	BI, J	Phillip Is; Mud Is
Pacific Gull	*Larus pacificus*	210	800	IVD	TM, U (tips)	BI, J	Bass Strait islands (inc off Wilsons Prom & a few on Phillip Is)
Silver Gull	*Chroicocephalus novaehollandiae*	2154	11707	IVD	TM, U	BI, U	Phillip Is & Mud Is

Main food: I, invertebrates; F, fish; V, other vertebrates (eg frogs & reptiles); D dead matter or garbage; P, plant matter in water; P(L) plant matter on land; P(W) plant matter in non-tidal wetlands.

Main feeding habitats: TM, tidal mudflats; SM, saltmarsh; SW, shallow water (sea or fresh); SFW; shallow fresh water; AV, vegetated non-tidal wetlands B, sandy beaches; P, pasture; U, utilities (eg tips, towns).

Main roosting habitats: BI, banks, islets or spits; R, rocks; M, mangroves; P, pasture; SM, saltmarsh; SW, shallow water; W, wetlands; J, jetties, moored boats or posts; T, trees in or near wetlands; U, utilities.

The count of 42 lesser Sand Plover includes 40 at an unusual location for the species (Bunyip/Yallock Feb 1990): otherwise up to 18 recorded regularly south coast of French Island to 1997, few subsequently.

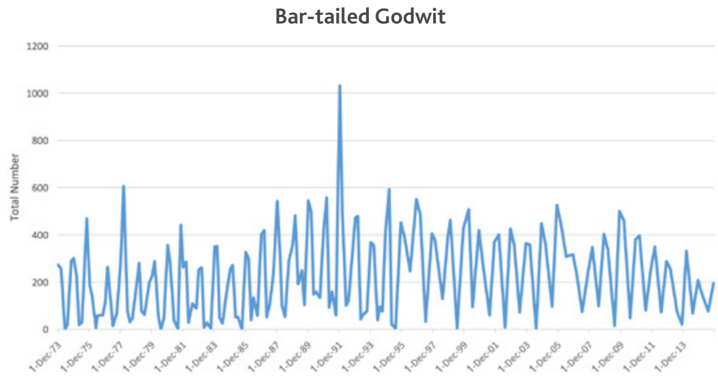
The count of 143 Whimbrel includes flocks of 73 at Observation Point and 70 at Bunyip/Yallock Feb 1995: if they were the same birds, the max would be 73, with no other records exceeding 55.

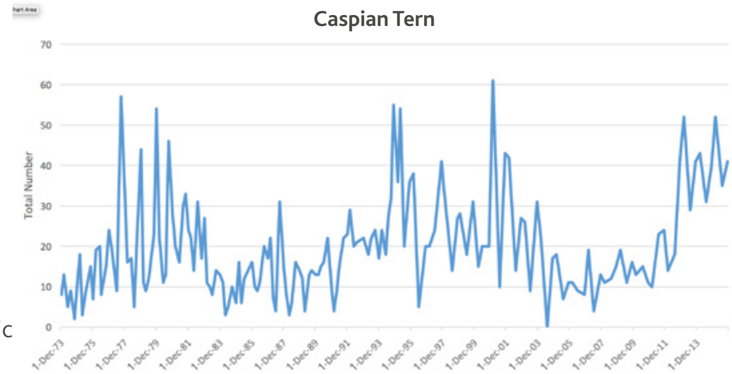
A higher count of Bar-tailed Godwits on one occasion (1031 in Dec 1991) has been excluded as it may have involved double counting when the flock moved between roosts at Observation Point and Rams Island.

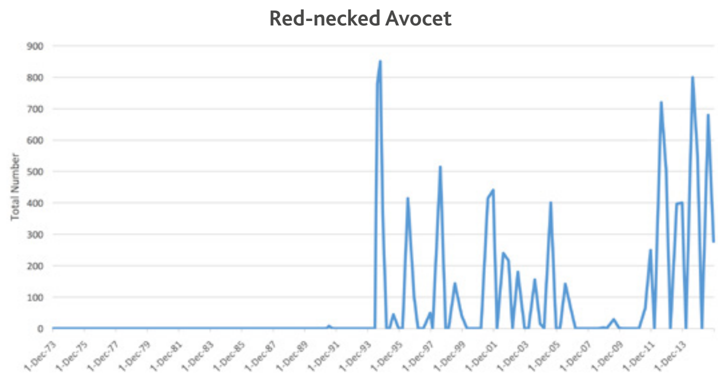
Changes in total numbers of selected waterbird species on each of the 168 counts are shown in Figure 8.1. Almost all species showed strong seasonal patterns, with consistent maxima at one season and minima at another. For non-migratory species, the minimum numbers generally occurred at seasons when they were known to be breeding at local wetlands outside the survey sites, or further afield in Australia. In many cases the peak breeding season was from late winter to spring and early summer, and the maximum counts were often recorded in late summer, autumn and winter. Hence, an unfortunate consequence of dropping the spring and autumn counts in 1994 was the loss of information about seasonal variation. No waterbirds other than shorebirds were consistently absent at particular seasons, with Cattle Egrets as the sole exception (this species was a winter visitor, recorded erratically from autumn to spring but usually absent in summer).

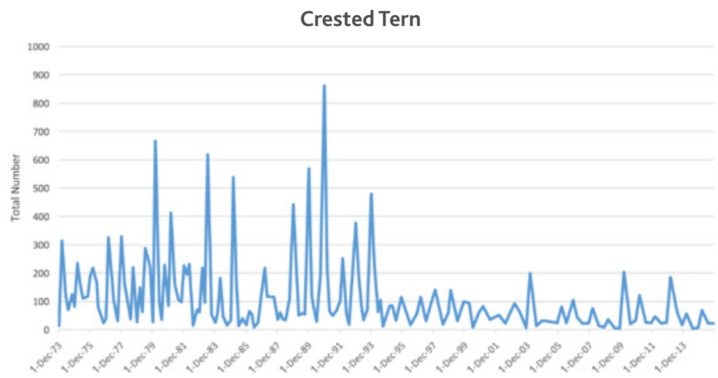
For trans-equatorial migratory shorebirds (breeding in North East Asia or Alaska), the patterns generally involved maximum numbers in summer and minimum numbers in winter (as expected, because they breed in the northern hemisphere when it is summer there, during the austral winter). Small numbers of most species remained over winter (young birds), but this was not the case for Sharp-tailed Sandpiper *Calidris acuminata*, Pacific Golden Plover *Pluvialis fulva*, Lesser Sand Plover *Charadrius mongolus* and Greater Sand Plover *Charadrius leschenaultii*, which were absent (or extremely rare) in winter. Numbers tended to be higher in late summer than early summer, especially for species such as Common Greenshank *Tringa nebularia* and Sharp-tailed Sandpiper which may visit ephemeral inland wetlands before resorting to coastal waters as these wetlands dry over summer. Numbers of Double-banded Plover (*Charadrius bicinctus*) which breeds in New Zealand were, as expected, highest in winter, with just a few early arrivals, usually juveniles, in late summer.

Numbers of most species fluctuated between years, and some appeared to increase over time while some decreased and others showed more complex patterns. These trends are considered further in the next section.









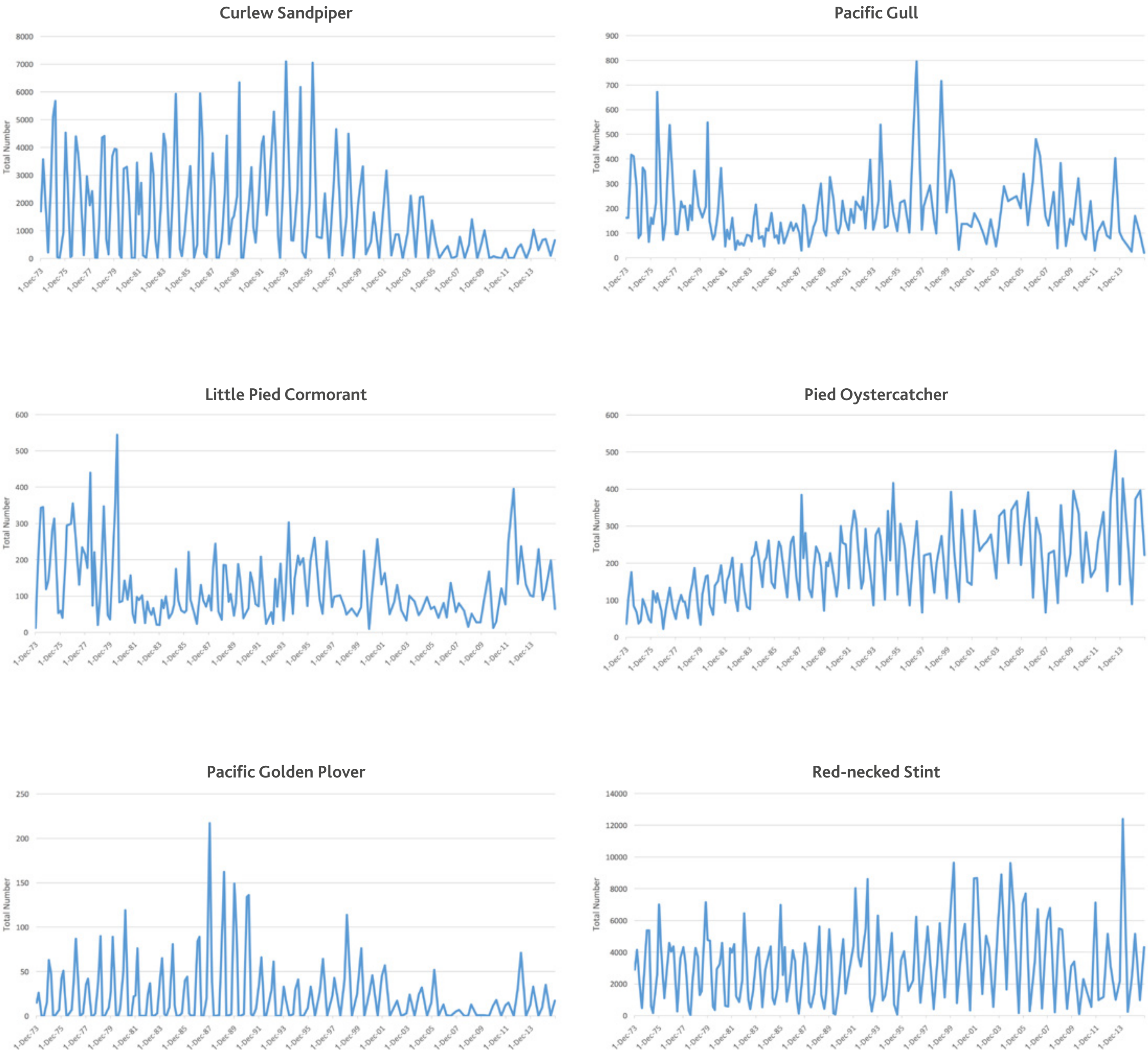


Figure 8.1 Numbers of selected waterbirds counted in Western Port 1973-2015, on 168 counts for the BirdLife Australia Western Port survey. Numbers shown are raw totals from the main sites, with no allowance for missing counts. The species selected include three of many that declined (Little Pied Cormorant, Crested Tern and Curlew Sandpiper); two that increased and then declined (Pacific Golden Plover and Bar-tailed Godwit); two that increased (Pied Oystercatcher and Red-necked Avocet) and three that showed stable or complex patterns with no clear trend over time (Red-necked Stint, Caspian Tern and Pacific Gull).

Broad analyses of waterbird trends 1974-2009

Data on 39 waterbird species were sufficient for statistical analysis of trends over 35 years (1974-2009) (Hansen et al. 2015). They included 17 species of shorebird, eight species of specialist fish-eating bird (four cormorants, a pelican and three terns), five ducks, a swan, a grebe, two gulls and five large wading birds (a heron, an egret, a spoonbill and two ibis). Of those 39 species, 13 were found to have declined significantly during the 35 years and nine other species increased initially and then declined (Table 8.4). The first group included Australian Pelican *Pelecanus conspicillatus*, Great Cormorant *Phalacrocorax carbo*, Little Pied Cormorant *Phalacrocorax melanoleucos*, Crested Tern *Thalasseus bergii*, Silver Gull *Chroicocephalus novaehollandiae*, White-faced Heron *Egretta novaehollandiae*, Grey Teal *Anas gracilis*, Masked Lapwing *Vanellus miles* and five migratory shorebird species (Common Greenshank *Tringa nebularia*, Curlew Sandpiper *Calidris ferruginea*, Eastern Curlew *Numenius madagascariensis*, Grey-tailed Tattler *Tringa brevipes* and Ruddy Turnstone *Arenaria interpres*). The second group included Hoary-headed Grebe *Poliocephalus poliocephalus*, three ducks (Australian Shelduck *Tadorna tadornoides*, Chestnut Teal *Anas castanea* and Pacific Black Duck *Anas superciliosus*) and four migratory shorebirds (Pacific Golden Plover *Pluvialis fulva*, Red Knot *Calidris canutus*, Bar-tailed Godwit *Limosa lapponica* and Whimbrel *Numenius phaeopus*). Two additional species of migratory shorebird (Lesser Sand-Plover *Charadrius mongolus* and Greater Sand Plover *Charadrius leschenaultii*) were regularly found in low numbers at the start of the survey, but have become extremely rare or absent.

Just two of the 39 species analysed (Australian Pied Oystercatcher *Haematopus longirostris* and Straw-necked Ibis *Threskiornis spinicollis*) were found to have increased significantly in the 35-year period. Four additional species increased markedly in the central-eastern part of the bay (Red-necked Avocet *Recurvirostra novaehollandiae*, Banded Stilt *Cladorhynchus leucocephalus*, Whiskered Tern *Chlidonius hybridus* and Gull-billed Tern *Gelochelidon nilotica*), but because they had extremely low initial population sizes it was not practical to include them in the formal statistical analyses. Another species (Cape Barren Goose *Cereopsis novaehollandiae*) also increased markedly on Phillip Island (where numbers are now in the thousands) and parts of French Island, but again it had a very small initial population that excluded it from formal analyses. Both Straw-necked Ibis and Cape Barren Goose feed mainly from pasture, and not from tidal mudflats.

No significant nett trends were found for a range of common species such as Black Swan, *Cygnus atratus,* although localised declines associated with seagrass loss in particular parts of the bay are being investigated for this species. Similarly, no significant trends were observed for Australian White Ibis *Threskiornis molucca*, Royal Spoonbill *Platalea regia*, Pacific Gull *Larus pacificus*, Caspian Tern Hydroprogne caspia, Red-necked Stint *Calidris ruficollis* and Double-banded Plover *Charadrius bicinctus*. The Red-necked Stint and Double-banded Plover were the only common migratory shorebirds not to show significant declines in number (Table 8.4).

Table 8.3 Mean counts, weight (from HANZAB) and biomass of the 30 most numerous waterbirds observed during the BirdLife Australia Western Port survey on 168 counts from 1973 to 2015.

Species	Mean total count	Rank by count	Weight (g)	% of bird biomass	Family	Breeding range
Red-necked Stint	3317	1	25	0.53	Shorebird	NE Asia & Alaska
Silver Gull	2154	2	290	4.0	Gull	Australia
Black Swan	1974	3	5500	69.1	Swan	Australia
Curlew Sandpiper #	1744	4	57	0.63	Shorebird	NE Asia
Eastern Curlew #	617	5	900	3.5	Shorebird	NE Asia
Australian White Ibis	511	6	1950	6.3	Ibis	Australia
Chestnut Teal	463	7	650	1.9	Duck	Australia
Masked Lapwing	242	8	315	0.48	Shorebird	Australia
Bar-tailed Godwit	228	9	350	0.51	Shorebird	NE Asia & Alaska
Double-banded Plover	212	10	70	0.09	Shorebird	New Zealand
Pacific Gull	210	11	1350	1.8	Gull	Aus (coastal islands)
Pied Oystercatcher $	194	12	700	0.87	Shorebird	Australia (coasts)
Sharp-tailed Sandpiper	173	13	65	0.07	Shorebird	NE Asia
White-faced Heron	169	14	550	0.59	Heron	Australia
Straw-necked Ibis	159	15	1300	1.3	Ibis	Australia
Little Pied Cormorant	151	16	645	0.62	Cormorant	Australia
Grey Teal	150	17	650	0.62	Duck	Australia (inland)
Australian Shelduck	123	18	1450	1.1	Duck	Australia
Crested Tern #	118	19	310	0.23	Tern	Australia (coasts)
Red-capped Plover	95	20	38	0.02	Shorebird	Australia
Royal Spoonbill	88	21	1650	0.93	Spoonbill	Australia
Common Greenshank	85	22	170	0.09	Shorebird	NE Asia
Australian Pelican	67	23	5400	2.3	Pelican	Australia
Red-necked Avocet $	61	24	310	0.12	Shorebird	Australia (inland)
Pied Cormorant	59	25	1750	0.66	Cormorant	Australia
Ruddy Turnstone	50	26	115	0.03	Shorebird	NE Asia & Alaska
Pacific Black Duck	42	27	1000	0.27	Duck	Australia
Red Knot	41	28	120	0.03	Shorebird	NE Asia
Musk Duck	32	29	1975	0.41	Duck	Australia

Species marked # have declined markedly and would no longer rank as high. (Crested Terns have declined in the inner parts of the bay where the survey is conducted, but increased dramatically as a breeding species on the west coast of Phillip Island.)

Species marked $ have increased markedly and would now rank substantially higher.

Trends of fish-eating birds in Western Port and Corner Inlet

This study (Menkhorst et al. 2015) identified opposing trends in two bays, with terns, cormorants and pelicans decreasing in Western Port and increasing in West Corner Inlet (Figure 8.2). Little Pied Cormorant decreased in the late 1970s and early 1980s, in association with seagrass dieback. Crested Tern decreased later in the 1980s and 1990s, to a greater extent than other fish-eating species. Numbers of two smaller and less numerous tern species (Fairy Tern *Sternula nereis* and Little Tern *Sternula albifrons*) also declined in the 1980s and 1990s. Small numbers of Fairy Tern breed erratically in the bay (Lacey and O'Brien 2015) whereas Little Terns are scarce, non-breeding visitors. The overall decline suggests that feeding conditions for fish-eating birds have declined in Western Port and improved in West Corner Inlet. The latter trend correlated with increases in fish catch per unit effort (CPUE) at West Corner Inlet. Fish CPUE also increased in Western Port until 2008-10 when it declined before commercial fishing ceased. Hence, numbers of fish-eating birds in Western Port did not correlate with total fish CPUE. Nevertheless, some of the small fish favoured by terns (but not a target of the commercial fishery) are known to have declined sharply in the mid-1990s in Western Port and elsewhere (Dann et al. 2000, Neira et al. 1999). Commercial catches of Australian Salmon, *Arripis trutta,* declined sharply in Western Port in the 1980s, but increased steadily throughout the period in West Corner Inlet. This predatory fish often drives schools of anchovies to the surface where they may be caught by Crested Terns. Numbers of Crested Terns correlated positively with CPUE of Australian Salmon in both bays. This raises the intriguing possibility that terns benefit from the presence of predatory fish in shallow embayments, perhaps relying on them to drive schools of small fish close to the surface where they can be seen and caught by plunge-diving.

While terns were declining in the survey area in Western Port, a large breeding colony of Crested Terns became established from 1994 on vegetated slopes at the western end of Phillip Island (Chiaradia et al. 2002). This colony built up to its recent size of 5000 pairs in 2011-12 (PINP 2011) and though it was not used in 2016 or 2017, a number moved their breeding to Seal Rocks in this period. (P.Dann pers. obs.). So, while the numbers of Crested Terns recorded in the survey area were much smaller (a few hundred in early years), the overall story is positive for that species. No such compensatory event has been observed for Fairy Terns in Western Port. Fairy Terns nest on low islands close to the tide-line, at just a few scattered locations in the state (with more in South Australia and Western Australia). They are the subject of a current study by BirdLife Australia (funded by DELWP), recognising the low breeding numbers in Victoria and the vulnerability of their nest-sites to storm surge and predation.

Two other tern species (Whiskered Tern and Gull-billed Tern) were not captured in this analysis of fish eating birds, because they were rare in Western Port in the first two decades of the survey, and fish form only part of their diet. Whiskered Terns feed extensively on invertebrates and small fish picked from the surface of fresh water or calm shallow seawater, and Gull-billed Terns make characteristic shallow dives to take invertebrates such as crabs from mudflats, or fish from shallow water. Both species were recorded mainly in early summer (and much less often in the two seasons considered in the main analysis). Both species increased markedly in one part of Western Port (the Corinella segment from Stockyard Point to Grantville) in the 2000s, with records of up to 230 Whiskered terns and 126 Gull-billed Terns. Red-necked Avocet (a shorebird species that takes crustaceans from the water surface) colonised the same area from the early 1990s, with records of up to 850 birds, mainly in winter, spring or early summer, along with up to 180 Banded Stilts. The local influx of these four species (formerly rare across the bay) suggests a major change in the ecology of that part of the bay e.g. an abundant new supply of an undetermined food such as crustaceans, or small fish, for the terns. These species may also be influenced by conditions, or habitat loss, elsewhere in Australia.

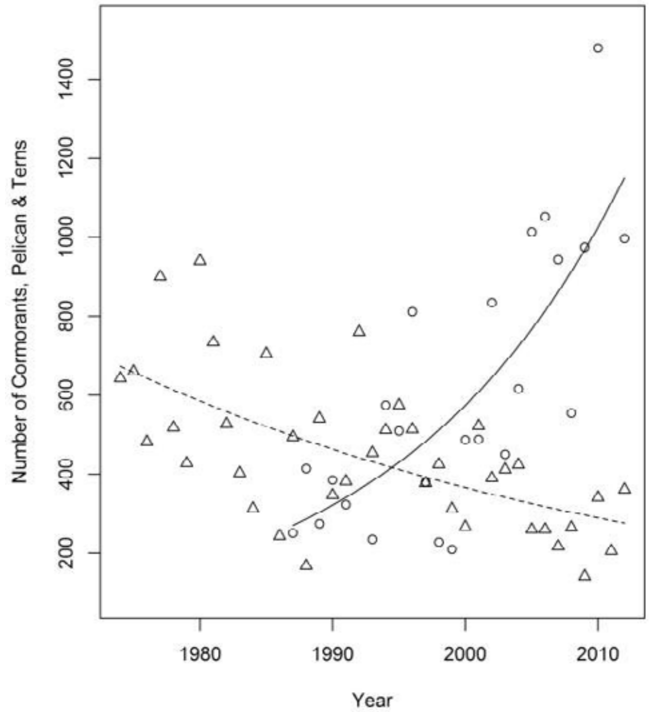


Figure 8.2 Annual combined summer counts of cormorants, pelican and terns in Western Port (triangles, dashed line) and in West Corner Inlet (circles, solid line) between 1974 and 2012. Both time trends are significant (Western Port $p < 0.0001$, R-sq (adj)=0.362; Corner Inlet $p < 0.0001$, R-sq (adj)=0.59). [taken from Fig 7 in Menkhorst et al. 2015.

Table 8.4 Population trends for 39 Western Port waterbird species over 35 years (1974-2009) surveyed by BirdLife Australia.

Observed trend in abundance	Common Name	Species
Significant decline in abundance relative to start of survey period	Australian Pelican Great Cormorant Little Pied Cormorant Crested Tern Silver Gull White-faced Heron Grey Teal Masked Lapwing Common Greenshank Curlew Sandpiper Eastern Curlew Grey-tailed Tattler Ruddy Turnstone	*Pelecanus conspicillatus* *Phalacrocorax carbo* *Phalacrocorax melanoleucos* *Thalasseus bergii* *Chroicocephalus novaehollandiae* *Egretta novaehollandiae* *Anas gracilis* *Vanellus miles* *Tringa nebularia* *Calidris ferruginea* *Numenius madagascariensis* *Tringa brevipes* *Arenaria interpres*
Increased initially followed by decline	Hoary-headed Grebe Australian Shelduck Chestnut Teal Pacific Black Duck Pacific Golden Plover Red Knot Bar-tailed Godwit Whimbrel	*Poliocephalus poliocephalus* *Tadorna tadornoides* *Anas castanea* *Anas superciliosus* *Pluvialis fulva* *Calidris canutus* *Limosa lapponica* *Numenius phaeopus*
Initially found regularly in low numbers, becoming extremely rare - absent	Lesser Sand Plover Greater Sand Plover	*Charadrius mongolus* *Charadrius leschenaultii*
Significant increase	Australian Pied Oystercatcher Straw-necked Ibis*	*Haematopus longirostris* *Threskiornis spinicollis*
Marked increase in the central-eastern part of the bay from an extremely low base	Red-necked Avocet Banded Stilt Whiskered Tern Gull-billed Tern	*Recurvirostra novaehollandiae Cladorhynchus leucocephalus* *Chlidonius hybridus* *Gelochelidon nilotica*
No significant net trends	Black Swan Australian White Ibis Royal Spoonbill Pacific Gull Caspian Tern Red-necked Stint Double-banded Plover Pied Cormorant Red-capped Plover	*Cygnus atratus* *Threskiornis molucca* *Platalea regia* *Larus pacificus* *Hydroprogne caspia* *Calidris ruficollis* *Charadrius bicinctus* *Phalacrocorax melanoleucos* *Charadrius ruficapillus*
Marked increase on Phillip Island and parts of French Island from a low base	Cape Barren Goose*	*Cereopsis novaehollandiae*

*Forage mainly from pasture, not mudflats.

Black Swan as a potential indicator of seagrass cover

Data on Black Swan were collated from 33 sites individually from 1973 to 2015 (168 count sessions) (Loyn 2016). A concurrent project by CSIRO for Melbourne Water is expected to provide comparable data on seagrass distribution from satellite images, at least for some of these sites and dates. Black Swan was the only bird species seen to feed on seagrass as a primary consumer, and seagrass was the main food that Black Swan were seen taking in tidal waters. Further information on its feeding behaviour and ecology has been collected by P. Dann (in prep.). All other obligate avian herbivores fed in nearby freshwater wetlands or from pasture, not from tidal mudflats.

Black Swans were the third most numerous waterbird species counted in the survey area, but contributed 69% of the biomass of all waterbirds observed during the survey. This was about ten times more than any other species (Australian White Ibis 6.3%, Silver Gull 4.0% and Eastern Curlew 3.5%) (Table 8.2). Note that these figures apply to the mudflat-dominated parts of the bay, not to the exposed waters outside the survey area, where marine seabirds including penguins, shearwaters, gannets and albatrosses would contribute the bulk of the avian biomass.

Analysis of survey data on Black Swans showed highly significant effects of site, season, five-year period and the interactions between these variables (Loyn 2016). Swans virtually disappeared from one part of the bay (Corinella segment) in the early 1980s, following a major documented loss of seagrass from that area. Numbers of swans elsewhere in the bay fluctuated in various ways, with no uniform linear trend in either direction.

This shows that counts of swans (a highly visible species) are providing strong signals about ecological change at the local level, and it will be valuable to correlate these changes in swan abundance with satellite data on seagrass distribution when it becomes available. If correlations are found, counts of Black Swan may provide a useful, cost-effective citizen science tool for detecting changes in seagrass distribution.

Discussion

What do the changes mean?

Various reasons have been suggested for some of the observed changes. Declines in migratory shorebirds reflect those observed elsewhere in Australia, and may be driven in part by loss of habitat at migratory staging sites in Asia (Hansen et al. 2015, Clemens et al. 2016). Fluctuations in inland-breeding birds relate to variable rainfall in inland Australia. Some of these birds (notably Grey Teal, Great Egret and Hoary-headed Grebe) vacate coastal wetlands to breed in newly filled ephemeral wetlands. These species become most numerous in Western Port in subsequent years after successful breeding (Dann et al. 1994, Loyn et al. 1994, Chambers and Loyn 2006), before declining gradually as the inland waters dry out during prolonged drought (presumably because recruitment from limited breeding is less than annual mortality). Declines in fish-eating birds in the1980s and 1990s may be driven partly by crashes in numbers of small fish, perhaps exacerbated by the decline in Australian Salmon. Declines in many waterbird species in the late 1970s and early 1980s were attributed to seagrass loss at the time, especially in the central-eastern Corinella segment (Dann et al. 1994, Loyn et al. 1994, Hansen et al. 2015). Although the seagrass loss was a negative event, part of the affected area developed new characteristics and now supports a seasonal influx of four bird species that were previously rare in the bay (Red-necked Avocet, Banded Stilt, Whiskered Tern and Gull-billed Tern). This potentially represents a rare example of a switch in ecological state that has added to regional diversity and deserves further investigation.

Value of the citizen science project

The BirdLife Australia survey has provided important insights into waterbird population longitudinal trends (increased, decreased or remained stable over time), and allows us to focus on possible reasons for the changes or stability. The survey has run with minimal input of public funds, and continues to do so. However, intermittent investments of public funds have enabled the data to be used to answer important questions to help manage these areas and their resources (including seagrass and fish, as well as birds). The three projects summarised here illustrate the application of these data and highlight its great value for addressing important conservation and management issues.

Many factors have contributed to the longevity of this survey, including the dedication of many counters and organisers over more than forty years. Several people have contributed consistently over the whole period. A crucial factor is that many people find counting waterbirds an enjoyable exercise (Heislers 2003). Part of its attraction is that there are always surprises to be found, including unexpected species and local changes in numbers, all of interest to the counters. Survey sites are defined broadly, as they must be: birds usually feed and roost in different areas, depending on tide, weather and many other factors. Counters must have flexibility to search for them each time within a general area, and this challenge adds to the enjoyment.

Future directions and opportunities

Black Swans and seagrass

8.1 Investigate whether the presence of Black Swans *(Cygnus atratus)* is a useful indicator for determining long-term trends in seagrass cover.

Swans virtually disappeared from the Corinella segment in the early 1980s, following a major documented loss of seagrass from that area. It will be valuable to correlate changes in swan counts with satellite data on seagrass distribution which is currently being compiled by CSIRO. Counts of Black Swan may provide a useful citizen-science tool for detecting changes in seagrass distribution, and complement remote sensing imagery or more labour intensive seagrass mapping undertaken through field surveys.

Ecological processes and seagrass cover

8.2 Characterise ecological processes in parts of the bay where seagrass has been lost, notably in the Corinella segment.

Evidence from waterbird monitoring shows that there has been a state change in the Corinella segment as seagrass has been lost, but little is known about the main drivers of productivity in the novel ecosystem that appears to have become established there, or the potential broader ecological implications. This change in habitat in the Corinella segment has brought benefits to some waterbird species (some of which feed mainly on crustaceans) and may have broader benefits, for example to the population of Elephant Fish, which has prospered there in recent decades, until a relatively recent decline (Chapter 7). The segment has attracted some vagrant bird species rarely seen in Victoria (e.g. Ringed Plover, South Island Pied Oystercatcher, Little Stint and the Asian subspecies of Gull-billed Tern), and congregations of four bird species that were previously rare in Western Port (Red-necked Avocet, Banded Stilt, Gull-billed Tern and Whiskered Turn). Improved understanding of ecological processes of this area and its importance for a range of birds and fish may assist in species conservation and management.

Utilisation of sediment mounds

8.3 Investigate use of new sediment accumulations by waterbirds for roosting or nesting.

The artificial sediment mound at Long Island (near Hastings) was a valuable high-tide roost for many years but appears to have become less suitable. Fairy Terns are known to have used artificial sediment mounds elsewhere and their status as a breeding species in Victoria has become precarious, mainly reliant on one or two sites on French Island. Sea-level rise and possible port developments are likely to exacerbate the situation, but also provide opportunities to restore or create new habitat by judicious and informed use of dredge spoil or other material.

New survey approaches for determining long-term population trends across diverse habitats

8.4 Broad survey of birds using aquatic and saltmarsh habitats around the coasts of Western Port.

The survey described here focuses strongly on birds that use intertidal areas, and generally gather at defined roosts at high tide. An additional suite of birds inhabits a range of other wetland habitats around the coasts, and little systematic information has been collected on most of those species. These habitats include ocean beaches, mangroves, saltmarsh, vegetated freshwater wetlands, wet heathlands, creeks and their estuaries and artificial habitats such as farm dams. The bird species include some that are listed as threatened in Victoria or nationally (e.g. Lewin's Rail, King Quail and Australasian Bittern, which are cryptic and poorly understood in this area; and Orange-bellied Parrot and Hooded Plover, which have been more intensively studied). Other species are of interest because of their specialised habitats and restricted local distributions (e.g. Common Sandpiper in narrow creeks and beaches; Blue-winged Parrot, Striated Fieldwren and Southern Emu-wren in saltmarsh and associated habitats). Cape Barren Geese have increased substantially in pasture with farm dams. Marked differences have been noted anecdotally between the bird faunas of saltmarsh on French Island and the mainland, and it has been speculated that some species may have been lost from French Island (e.g. Superb Fairy-wren and Striated Fieldwren) because of the abundance of feral cats, while others may have benefited from the absence of foxes on French Island. More systematic research and documentation is needed to explore these relationships further. This could form part of the coastal biodiversity study proposed in Chapter 6.

8.5 Drone survey of low-tide feeding habitats for swans, shorebirds and other waterbirds.

Despite various attempts, our knowledge of preferred low-tide feeding areas remains fragmentary. This could be a crucial gap in knowledge if new proposals are made for port or other developments involving dredging or other damage to particular areas of intertidal mudflat. Helicopter drones provide a new opportunity to systematically explore the distribution of waterbirds at low tide. They are used routinely by Phillip Island Nature Park (PINP) staff for observing large animals (seals) and would certainly be useful for swans. With suitable modifications it is likely that this technology will also be able to provide useful data on a wide range of waterbird species that feed from open mudflats.

Conservation management

8.6 Ecological research on locally breeding shorebirds and terns to identify possible management interventions to improve breeding success.

Locally breeding shorebirds and terns are generally present in low numbers and rely on limited numbers of breeding sites, mainly on or near beaches. Some have prospered in recent years, notably Pied Oystercatcher (breeding mainly on fox-free French Island) and Hooded Plover (breeding on ocean beaches of Phillip Island — this species has benefited from fox control and targeted management). Other species have declined (e.g. Fairy Tern) or have precarious localised populations. Further research is needed to determine whether they would benefit from targeted management interventions.

8.7 Reintroductions of selected bird species to "island ark" habitats being created on Phillip Island, Churchill Island and French Island by controlling foxes and cats.

Foxes and feral cats have been removed from Churchill Is. and foxes removed from Phillip Is. (there are no foxes on French Is.). Control programs are in place for feral cats on Phillip Is. (PINP) and French Is. (PV and other agencies). Some of the work is being done to protect breeding penguins and to provide habitat for at-risk mammals. However, these efforts will also provide potential habitat for birds that may have been lost from the region historically, including ground-nesting species that may be especially vulnerable to predation, e.g. Bush Stone-curlew and Spotted Quail-thrush, and others that persisted locally round Western Port into the 1990s (notably Grey-crowned Babbler, which has now been lost from southern Victoria). This is considered a management action rather than a knowledge gap requiring research.

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9 Western Port Environment Research Program revised research needs and prioritisation

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Photo: Annette Hatten

Overview

Important gaps in our understanding of how best to protect and restore Western Port were originally captured and prioritised in Chapter 15 of the Western Port review (Keough et al. 2011), and formed the basis of the integrated Western Port Environment Research Program that followed.

The Western Port review identified a total of 43 strategic research needs that were prioritised from highest priority ('1') to lowest priority ('3') based on their expected benefit to scientific understanding and management, urgency, chance of a successful outcome or dependency on other research needs. Since the release of the Western Port review, several research needs have been completed or are well underway, and many medium priority projects have been initiated (see Table 1.1, Introduction).

In this chapter, we have revised and updated the list of strategic knowledge gaps for Western Port. This has been achieved by:

- Removing the original knowledge gaps from the 2011 Western Port review that have now been addressed or are underway (i.e. as listed in Table 1.1);
- Carrying over knowledge gaps from the 2011 review that have not been addressed; and
- Identifying new research needs based on the outcomes of research from the Western Port Environment Research Program 2011-2016 and related studies.

As with the initial Western Port review process, knowledge gaps within the updated list that have not commenced have been prioritised from 1-3.

Revised Western Port Strategic Knowledge Gaps

In addition to nine knowledge gaps from the original 2011 Western Port review that are currently being addressed (Table 1.1), there are a further 14 knowledge gaps being addressed that were identified as important by various studies as the Western Port Environment Research Program progressed between 2011 and 2016 (Table 9.1). Remaining knowledge gaps identified by either the 2011 review or this synthesis document are summarized in Table 9.2, of which there are 36 outstanding and grouped into 11 themes:

1. Improved hydrodynamic models of Western Port
2. Develop a complete sediment transport model
3. Other physical environmental understanding
4. Sediment and nutrient thresholds for important plants
5. Characterise present biodiversity
6. Trends through time
7. Functional links between organisms and habitat
8. Species of particular interest
9. Toxicants
10. Harvesting
11. Climate Change and changes to habitat quality

Of the remaining knowledge gaps, five are considered a high priority and are discussed in more detail below:

9.1 Finer resolution mapping of stream bank and gully erosion in the catchment

Stream bank and gully erosion is predicted to be a major source of fine sediment to Western Port, particularly from the Bunyip and Lang Lang River systems. Mapping of stream bank and gully erosion was initially undertaken by Hughes et al. (2003). New LiDAR imagery provides the opportunity for finer resolution data that could improve catchment modelling predictions of sediment loads, including assessing the local effectiveness of stream bank vegetation at mitigating erosion. This could be targeted at major sediment sources through selection of specific streams and gullies (Chapter 2 Wilkinson et al.).

9.2 Identify options for erosion control along the Lang Lang coastline to achieve water quality outcomes

Erosion of banks around Lang Lang in the north east contributes approximately 30% of total annual sediment load into the bay. Further studies of options to control coastal bank erosion are needed. This includes the feasibility of using artificial structures to reduce wave energy in combination with ‘green infrastructure’ (i.e. mangroves and potentially saltmarsh) to provide a long-term stabilisation solution for this coastline. Erosion control trials would be informed by sediment modelling scenarios of the relative benefits of catchment vs. coastal works for water quality (Chapter 2 Wilkinson et al.; Chapter 6 Hurst).

9.3 Determine water quality targets for sediments and nutrients that support microphytobenthos, reef algae, saltmarshes, and mangroves.

Anthropogenic pressures rarely act in isolation. In Western Port, interactive effects of sediment and nutrient loads on major primary producers are highly likely, including feedbacks via sediment stabilisation and nutrient transformation. Understanding the interactive effects and feedbacks will assist the prioritisation of management actions to reduce sediment and nutrient loads. Research as part of the Western Port Environment Research Program has initially focussed on seagrass, but not microphytobenthos, reef algae, saltmarshes or mangroves (WP review Chapters 8, 10, 13, 14; Chapter 3 Manassa et al.). Further research on these primary producers is required to gain a broader understanding of the potential impacts of sediment and nutrient inputs from the catchment and coastline.

9.4 Determine capacity for *Zostera* to recover and colonise new areas. Recovery of seagrass will require colonisation of large areas that previously had seagrass, and may require assisted recovery. Largely focused on *Z. muelleri.*

Although achieving a suitable light climate for seagrass to cover substantial areas of the bay is likely once legacy fine sediments have been flushed out of the system in the coming decades, we do not fully understand the capacity for *Zostera* species to recover and colonise new areas when light conditions become favourable. Further studies of *Zostera* spp. biology, reproductive strategies, and environmental tolerances (light, temperature, salinity, and nutrients) are required, building on work by Bulthuis and Woelkerling (1983), Clough and Attiwell (1980), and more recently the Monash University led Australian Research Council Linkage project (Chapter 3). Existing work in Port Phillip on *Z. nigracaulis* is providing some information on how large areas may be recolonised, but there is less information for *Z. muelleri* – the species that has experienced the greatest reductions in cover in Western Port. This information is needed to predict resilience to variables such as light reduction, climate change, increased sedimentation and freshwater run-off, and thus enable managers to predict future environmental impacts. A significant knowledge gap is whether large scale germination and establishment of seeds can occur (Western Port review Chapter 10; Chapter 3 Manassa *et al*. this document). Seagrass restoration trials in Western Port have previously been attempted (e.g. Western Port Seagrass Partnership 2008) and will inform further trials using a broader range of planting approaches. Model predictions from the current hydrodynamic and water quality model (Chapter 4) will help determine where restoration trials are most likely to be effective.

9.5 Determining the locations and timing of Elephant Fish *(Callorhinchus milii)* reproduction, and better understand the population decline in Western Port.

Given the apparent decline in the Elephant Fish population in Western Port, priority should be given to determine habitat use for spawning females and early juvenile through field surveys. The status of the larger, offshore stocks of Elephant Fish should be reviewed and compared with Western Port to determine if a decline is occurring more broadly across the species range. (Chapter 7 Jenkins).

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Table 9.1 Research projects that were identified as important in the Western Port Environment Research Program between 2011-2016 and that are underway.

No.	Brief Description	Details	Justification/Benefit	Chapters
Hydrodynamics and sediment dynamics				
P.1	Determine the likely time-frame for the bay to flush itself of legacy sediments, determine sustainable catchment and coastal erosion sediment loads, and identify areas suitable for seagrass regrowth.	Detailed spatial mapping of sediment composition and depth and longer model runs >20 years are required. Combine modelling with monitoring of concentrations and volumes exiting the upper north arm eastwards (remote-sensing or in-situ).	A better understanding of the current sediment composition and depths around the bay and longer model runs will assist with determining a sediment mass balance and may allow quantification of the length of time likely for the bay to flush itself of the legacy sediments e.g. Upper North Arm. Longer model runs will also establish the level of catchment and shoreline erosion that can be naturally managed by the bay and identify potential areas suitable for seagrass regrowth. Led by MW/Hydronumerics/CSIRO.	Chapter 2 Wilkinson et al., Chapter 4 Cinque et al.
P.2	Incorporate climate change scenarios in the WP hydrodynamic model, including sea level rise, sea temperature increase and changed rainfall and streamflow patterns.	EPA have run simmulations for sea level rise affecting tidal inflows (increases in current speeds of up to 25% in channels) and also looked at effects of mitigation actions (tidal barriers and coastal protection to improve light clarity (for seagrass) and mitigate sediment movement. Have also run some simple 2050 catchment inflow scenarios.	Climate change will likely alter the distribution of seagrass beds. Initial modelling of seagrass response by CSIRO confirms that both water quality and water depth impact significantly on light availability and that one metre of sea level rise and/or an increase in water temperature would be sufficient to substantially reduce seagrass extent. Led by MW/Hydronumerics/CSIRO.	Chapter 4 Cinque et al.
P.3	Integration of seagrass algorithms that predict seagrass growth and its influence on sediment dynamics	Integration of the dynamic seagrass algorithms into the WP hydrodynamic model which include the interplay between above ground and root biomass and incorporate feedback loops to flow and sediment erosion and deposition.	Can currently use models to identify areas where light climate is amenable to seagrass growth (but where no seagrass is currently living). This research would enable identification of the sediment conditions suitable for seagrass colonisation, and potential effects on sedimentation following recolonisation.	Chapter 2 Wilkinson et al. and Chapter 4 Cinque et al.
P.4	Develop a dynamic catchment sediment loads model	Development of a catchment planning tool such as Dynamic SedNet that spatially and temporally represents the primary sources of sediment and nutrients.	A catchment planning tool will help determine priorities for erosion management, and evaluating the effect of changes in management. Led by MW/Hydronumerics/CSIRO.	Chapter 2 Wilkinson et al.
P.5	Monitoring of river sediment and nutrient concentrations - sediments a priority	In addition to ongoing routine water quality monitoring in the catchment, it is recommended that the loads monitoring program within major waterways be re-established. Further analysis of historical fine river sediment and nutrient concentration and particle size could be undertaken.	On-going routine and loads monitoring of sediment and nutrient concentrations in major waterways would help to inform management priorities, to evaluate their effects, and to constrain modelling of catchment sources. Turbidity sensors have been demonstrated to improve load estimates. MW/CSIRO.	Chapter 2 Wilkinson et al.
P.6	Determine the relative contribution of urban areas to catchment sediment and nutrient loads including during construction.	Much smaller source than streams. More chronic in effect (less episodic) and potentially better regulated. On their own, unlikely to drive enough change in water quality but could be reduced.	Quantifying the contributions of urban development relative to runoff from existing urban and agricultural areas would help to inform management priorities and inform future stormwater management targets. Initial modelling work has been commissioned by Melbourne Water and may lead to further work, including field validation of predictions.	Chapter 2 Wilkinson et al.

Table 9.1 Research projects that were identified as important in the Western Port Environment Research Program between 2011-2016 and that are underway.

No.	Brief Description	Details	Justification/Benefit	Chapters
Ecosystem processes				
P.7	Determine contribution of seagrass nitrogen fixation to food webs.	Nitrogen fixation well known, has been characterised, but not links to food webs.	Nitrogen fixation by seagrass has been shown to be an important process. Further studies are required to determine the importance of this to food webs within WP. MW/Monash.	Chapter 5 Manassa et al.
P.8	Identify site-specific tolerances of intertidal seagrass to water quality, particularly for areas where losses of seagrass cover have been the greatest. Relates to R.15 and R.16 (Table 1.1)	Site-specific studies on the genetics, physiology and morphology of intertidal seagrasses around the Bay and how this relates to water quality.	Research as part of the WP Environment Research Program has determined substantial genetic structure and variable water clarity responses of intertidal seagrasses for different parts of the bay. It is particularly important to understand the tolerance of intertidal seagrass in areas where losses of seagrass cover have been the greatest. Initial genetics work done, but not site-specific tolerances. MW/Monash.	Chapter 3 Manassa et al.
P.9	Explore the potential use of black swan data as an indicator of seagrass cover and use as citizen-science monitoring tool.	Complement remote sensing imagery or more labour intensive seagrass mapping undertaken through field surveys.	Swans virtually disappeared from the Corinella segment in the early 1980s, following a major documented loss of seagrass from that area. It may be possible to correlate changes in swan counts with satellite data on seagrass distribution. Counts of Black Swan may provide a useful citizen-science tool for detecting changes in seagrass distribution. MW/ EcoInsights.	Chapter 8 Loyn et al.
P.10	Identify drivers of long-term biological change in WP.	Analysis of trends and change point data for 3 fish species with environmental parameters.	For many coastal and estuarine systems we currently lack the knowledge to link particular environmental drivers or events to observed biological changes. Long-term data sets, when appropriately analysed, can provide this information. This understanding is essential if we are to sustainably manage coastal and estuarine environments and protect their valuable ecosystem services in a changing world. MW/University of Melbourne.	Chapter 7 Jenkins.
Toxicants				
P.11	Assess occurrence of pesticides in surface waters and sediments within additional sub-catchments	In order to understand the extent of pesticide contamination in WP, chemical analysis of surface waters more broadly across WP is recommended. While current monitoring and research has indicated that intensive agricultural land uses (such as market gardens) are likely to be the main contributor to pesticides in the north-western waterways of WP, land use in these catchments is significantly changing. Reassessment of pesticide risks is recommended in regions where significant land use changes have occurred or are occurring.	A project led by CAPIM and Melbourne Water investigated the temporal occurrence of pesticides in waterways flowing into the north east of WP commencing 2017. This will provide a greater understanding of pesticide risks across the broader WP catchment. MW/University of Melbourne.	Chapter 5 Myers et al.

Table 9.1 Research projects that were identified as important in the Western Port Environment Research Program between 2011-2016 and that are underway.

No.	Brief Description	Details	Justification/Benefit	Chapters
P.12	Determining the major catchment sources of toxicants.	Expansion of initial survey to address recommendation in WP review to determine whether toxicants likely to be a threat in WP. Toxicants in sediments and surface waters measured across the bay and select waterways. Current work is focused on a pesticide sourcing program (PSP) - to isolate and identify the sources of pesticides in key catchments in WP - and to assess the health of resident fish.	Determine to what extent toxicants are affecting marine, estuarine and freshwater biota. Identify and manage toxicant sources to minimise environmental impacts. See Myers et al. (2016). MW/University of Melbourne.	Chapter 5 Myers et al.
P.13	Additional investigation of toxicant effects on freshwater and estuarine fish. Work on climate impacts not yet commenced.	Further investigation of the tolerances of fish species to reduced water quality in north of the bay. Suitable indicator species are Blue Spot Goby (estuaries) and Flat Headed Gudgeon (freshwater). Expand fish toxicant surveys throughout WP and additional external reference sites. A 2017 project led by CAPIM and Melbourne Water using Smooth Toadfish as an indicator species is currently underway.	Chemical analyses conducted to date indicate that the majority of pollution in WP is derived from upstream inflows and tributaries and an assessment of the health of fish living within these catchments is recommended. Additional sites will assist with interpretation of previous toadfish findings and more generally our understanding of the effect of toxicants within WP on fish health. MW/University of Melbourne.	WP Review Chapter 11; Chapter 5 Myers et al.
P.14	Understand the connectivity of individuals and population structure of Smooth Toadfish throughout the bay, and implications for toxicant research.	An investigation of the genetic structure of toadfish sampled from different locations across WP and more broadly. This project is underway as part of the 2017 fish health study.	It is presumed that toadfish sampled from different estuaries within WP represent discrete populations and therefore reflect local conditions. An investigation of the genetic structure of fish sampled from different locations would identify how much dispersal and mixing may be occurring. If there is a high degree of mixing amongst toadfish from different sites (considered unlikely), then it may be more appropriate to consider the health of WP toadfish across the whole bay, rather than within individual waterways. MW/University of Melbourne/Deakin University.	Chapter 5 Myers et al.

Table 9.1 Research projects that were identified as important in the Western Port Environment Research Program between 2011-2016 and that are underway.

No.	Priority	Status	Brief Description	Details	Justification/Benefit	Chapters
Hydrodynamics and Sediment Dynamics						
Theme: Improving hydrodynamic models of Western Port						
S.1	2	Carry over from WP review (R.3)	Incorporate contributions of heating and cooling of intertidal mudflats into oceanographic model.	The shallow WP system is highly sensitive to conditions on the mud flats that transfer to the water column during tidal exchange. More sophisticated models are required to represent this process. Targeted continuous data collection of key physical parameters (temperature, salinity, solar radiation).	Part of refining bay circulation model. Will be needed for accurate predictions of climate effects.	WP review Chapter 4
S.2	2	New	Further mapping of WP bathymetry.	Bathymetry data is patchy across the north and northeast of the bay, particularly in the area between Corinella and Stockyard Point. EPA has undertaken some preliminary depth-sounding work but more systematic data collection is required.	Updating the bathymetry in the areas where there is still some uncertainty is important to ensure flow velocities and subsequent settling are more accurately predicted, and will be important for numerous future activities outside of modelling. In addition to improving model predictions for this region, measures of existing bathymetry should serve as reference points to monitor future change.	Chapter 4 Cinque et al.
S.3	2-3	New	Testing the sensitivity of a spatially variable wind field in the model.	Undertake further monitoring, and validate against Lang Lang data. This could include more monitoring stations or a validated wind field model based on current stations and periods at Lang Lang.	There are currently only two WP wind-monitoring stations - Cerberus and Rhyll - but the wind field varies across the bay, and these measures are not indicative of the north and northeast. This would enable further accuracy of the model, including wave-field and erosion depiction.	Chapter 4 Cinque et al.
S.4	2	New	Improve the ability of the WP hydrodynamic model to predict water clarity and seagrass cover using remote sensing data.	Develop an understanding of the of the time-extent-duration of resuspension by using the full set of Landsat images covering a large number of tide stage and wind conditions. Use Landsat-based predictions, including new Landsat 8 and the Sentinel series, of particulate concentrations, water clarity and seagrass/macro-algae extent.	This would provide data to constrain modelling of seagrass extent and turbidity/water clarity and provide a monitoring program covering seagrass extent and turbidity that is more comprehensive spatially and temporally than can be achieved by field survey. Further improve the accuracy of the hydrodynamic model and to monitor future changes in water clarity and seagrass cover. Field measurements of the spectral characteristics of WP would improve remote sensing analysis of seagrass extent and particulate concentrations, and digital data from additional historical seagrass surveys will improve validation of remote sensing. Monitoring changes in the spatial extent of macrophytes over time using remote sensing would require consideration of exposed at low tide as well as that submerged.	Chapter 2 Wilkinson et al.

Table 9.2 Recommended strategic knowledge gaps for the management of the Western Port environment. These include new research projects identified in this synthesis, as well as some projects carried over from the original 2011 review.

No.	Priority	Status	Brief Description	Details	Justification/Benefit	Chapters
S.5	2	New	Determine the relative impact of tidal resuspension and river plumes on seagrass shading based on a detailed historical archive of water quality.	Particle size variations in turbid parts of WP could be modelled using remote sensing imagery to help distinguish between new sediment inputs and sediment resuspension by tidal currents and wind-induced waves. Would be a side investigation using the same imagery and field validation as S.7.	Developing a more detailed historical archive of water clarity would assist further investigation of the effect on particulate concentrations of wind and tidal resuspension relative to river inputs. The effect of river loading on seagrass shading events can be tested by simulating river plume development.	Chapter 2 Wilkinson et al.
Theme: Develop a complete sediment transport model						
S.6	1	New	Finer resolution mapping of stream bank and gully erosion in the catchment.	Mapping of stream bank and gully erosion was initially undertaken by Hughes et al. (2003). New LiDAR imagery provides the opportunity for finer resolution data that could improve catchment modelling predictions.	To improve catchment model predictions of sediment loads, including assessing the local effectiveness of stream bank vegetation at mitigating erosion. This could be targeted at major sediment sources - selection of specific streams and gullies.	Chapter 2 Wilkinson et al.
S.7	1	New	Identify options for erosion control along the Lang Lang coastline to achieve water quality outcomes.	Conduct a feasibility study into coastal erosion control structures, and possible trial, with a view to preventing coastal erosion in the short-term and allowing mangrove (and potentially saltmarsh) establishment to provide a long-term stabilisation solution for this coastline. Would be informed by sediment modelling scenarios of the relative benefits of catchment vs. coastal works for water quality.	Erosion of banks around Lang Lang in the north east contributes approximately 30% of total annual sediment load into the bay. Control of coastal bank erosion requires further study, to identify and prove suitable options.	Chapter 2 Wilkinson et al.; Chapter 6 Hurst
Theme: Other physical environmental understanding						
S.8	3	Carry over from WP review (R.7)	Incorporate contributions from groundwater and in-stream processes to provide more robust modelling.		Identifying the origin of nutrients from the catchment, atmosphere and with-in bay processes is important to prioritise management of water quality in WP.	WP review Chapters 4, 14

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No.	Priority	Status	Brief Description	Details	Justification/Benefit	Chapters
S.9	2	Relates to R.15 (Carry over, commenced Table 1.1)	Assess the degree of nutrient and light limitation for primary producers other than seagrass.	Work for seagrass largely complete, and awaiting final model completion to scale likely light limitation to the system. Needs to be done for other primary producers. Assessment of nutrient (N vs P) and light limitation in the major primary producers (benthic microalgae, macroalgae, phytoplankton). Microphytobenthos is fairly resilient to light availability and can photosynthesise at low tide. Macro algae need less light than seagrass, so light levels for seagrass will be broadly be indicative.	The composition and biomass of the major primary producers in WP has major implications for associated food webs. Understanding the degree of nutrient and light limitation would significantly enhance our ability to predict consequences of changes in nutrient and sediment inputs given their direct and indirect effects on nutrient availability and light levels.	WP review Chapters 4, 10, Chapter 3 Manassa et al.
S.10	1	Relates to R.16 (Carry over, commenced Table 1.1)	Determine water quality targets for sediments and nutrients that support microphytobenthos, reef algae, saltmarshes, and mangroves. Related work on seagrass commenced.	Interactive effects of nutrients and sedimentation on major primary producers, including feedbacks via sediment stabilisation and nutrient transformation. Linked projects, microphytobenthos, seagrass habitats, coastal saltmarsh and mangroves, reefs, water column. Seagrasses are the highest priority (underway) and reef algae a lesser priority.	Anthropogenic pressures rarely act in isolation. In WP, interactive effects of sediment and nutrient loads are highly likely. Understanding the interactive effects and feedbacks will assist the prioritisation of management actions to reduce loads. Research as part of the WP Environment Research Program has initially focussed on seagrass, but not microphytobenthos, reef algae, saltmarshes or mangroves.	WP review Chapters 8, 10, 13, 14; Chapter 3 Manassa et al.
Ecosystem Processes						
Theme: Sediment and nutrient thresholds for important plants						
S.11	3	Carry over from WP review (R.17)	Determine cause of elevated water-column chlorophyll in Corinella segment.	Determine species composition of phytoplankton; including temporal and spatial patterns, to determine whether algae in water column are planktonic or benthic species.	Important measure of water quality under current (e.g. turbidity) and predicted (i.e. climate change-associated) stressors. Species contributing to chlorophyll-a measures. Will determine whether elevated levels in Corinella segment are related to excess nutrients and uptake in the water column or reflect resuspension of benthic microalgae.	WP review Chapters 5, 14

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No.	Priority	Status	Brief Description	Details	Justification/Benefit	Chapters
S.12	2	Carry over from WP review (R.18)	Determine the role played by dead plant material from the dominant vascular plants in the availability, transport and tranformations of nutrients, including for higher trophic levels.	Need better understanding of the effects of vascular plant detritus (seagrasses, mangroves and coastal saltmarsh) on nutrient budgets, productivity and mangrove survival.	Vascular plants produce a significant biomass of detritus (from the plants themselves and from and associated epiphytes) that can be deposited locally in the bed or transported to other habitats. The consequences for detrital based food webs and nutrient production will depend on transport processes and how labile the detritus is. This has the potential to have major implications for lateral energy flow between habitats, energy transfer to higher trophic levels and nutrient availability.	WP review Chapter 8
S.13	2-3	New	Refine indicators of seagrass stress.	Further replication and examination of the temporal dynamics of metabolites is needed to develop the use of metabolite measurements as a reliable indicator of seagrass stress. Field studies that link light climate with indicators including carbohydrates, chlorophyll a and metabolomics to find indicator thresholds for light stress.	Reliable indicators of seagrass are required for ongoing monitoring of changes in seagrass health within WP, and improved understanding of conditions that impact seagrass health.	Chapter 3 Manassa et al.
How different is the Western Port ecosystem from when it was described in 1975?						
Theme: Characterise present biodiversity						
S.14	3	Carry over from WP review (R.20)	Determine whether deep channels harbour reef fauna and use improved fish survey techniques around sedentary invertebrate isolates in these channels.	Examination of walls and floor of deep channels, to determine if they act as de facto reefs, and if this information alters our picture of overall WP biodiversity or identification of areas of particular interest for biodiversity.	Most of the reef areas have not been surveyed extensively, nor is the fauna of channel walls known, leading to possible incomplete knowledge of current biodiversity. Information may be needed in the event of any major construction that involves modification to channels. Sampling of fish communities in these habitats not is not practical using typical sampling techniques as nets will become snagged while high turbidity and low light reduce the effectiveness of underwater video. Acoustic sonar camera techniques may be one option to survey fish in these habitats in the future.	WP review Chapter 13; Chapter 7 Jenkins

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No.	Priority	Status	Brief Description	Details	Justification/Benefit	Chapters
S.15	3	Carry over from WP review (R.21 & R.30)	Identify differences between current state of WP soft sediment faunal assemblages and earlier descriptions. Produce updated spatial description of subtidal soft sediment areas.	Soft sediments a priority to assess degree of potential change, lower priority for other habitats. Invertebrate fauna in mangroves largely done. Spatial description would enable identification of areas of functional importance e.g. ecosystem engineering and biogenic structures. Determine the features that support high benthic biodiversity in WP and obtain biomass estimates of macrofauna, particularly those that are important for nutrient cycling. Evrard et al. (2013) have looked at nutrient processing on tidal flats in WP.	Would indicate if there are differences in the fauna that would affect important ecosystem processes, particularly nutrient cycling. Description of current spatial patterns would provide a basis for assessing threats to biodiversity assets within WP and may provide information on potential changes in soft sediment communities.	WP review Chapters 6, 13; Chapter 6 Hurst
S.16	2-3	Carry over from WP review (R.22) and new	Estimate extent of invasion of key marine habitats and better understand the threat of weeds to coastal vegetation.	Introduced species - extent of invasions and species present in various habitats. Only Tall Wheat grass in saltmarsh done. Linked to next recommendation for coastal vegetation specifically. Review high threat saltmarsh weeds identified in the Victorian Saltmarsh Study and determine extent and impact of these across WP saltmarshes. Collate and analyse data from WP Spartina eradication program for publication in broader scientific literature.	No monitoring since 2000. Invasive species can alter ecosystem processes, and can degrade individual assets. The information would be used to inform other management, including nutrient model.	WP review Chapters 7, 13; Chapter 6 Hurst
S.17	3	Carry over from WP review (R.23 & R.24)	Through biodiversity surveys, determine affinities of WP biota WP including biodiversity associated with saltmarshes and mangroves.	The geological history of WP suggests a stronger link with the East coast of Australia than with Port Phillip. Possibility of some immigration from Port Phillip (native and invasive species) resulting in some 'homogenisation' of the fauna of the two bays. Determine whether WP saltmarsh and mangroves harbour a fauna that differs from that occurring elsewhere in SE Australia. Clarify taxonomic and structural diversity of coastal saltmarsh, with reference to the purported lack of species diversity. Describe bird fauna of mangroves and saltmarsh, and highlight known differences between mainland and French Island.	Used in refining the identification of individual marine assets. In respect to birds, may also provide insights into possible management needs and opportunities such as addressing predation pressure from foxes and cats.	WP review Chapter 7, 8; Chapter 6 Hurst, Chapter 8 Loyn et al.

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No.	Priority	Status	Brief Description	Details	Justification/Benefit	Chapters
Theme: Trends through time						
S.18	1	New. Relates to R.26 (Table 1.1) (Carry over from WP review, commenced).	Determine capacity for *Zostera* to recover and colonise new areas. Recovery of seagrass will require colonisation of large areas that previously had seagrass, and may require assisted recovery. Largely focused on *Z. muelleri*.	Some of this work is currently underway as part of a Monash University led ARC Linkage project. Studies of *Zostera* spp. biology, reproductive strategies, and environmental tolerances (light, temperature. salinity, and nutrients). Build on work by Bulthuis and Woelkerling (1983) and Clough and Attiwell (1980). Existing work in Port Phillip on *Z. nigracaulis* is providing some information on how large areas may be recolonised, but there is less information for *Z. muelleri*.	Needed to predict resilience to variables such as light reduction, climate change, increased sedimentation and freshwater run-off, to allow managers to predict future environmental impacts. A significant knowledge gap is whether large scale germination and establishment of seeds can occur.	WP review Chapter 10; Chapter 3 Manassa et al.
S.19	3	New. Extension of P.12 (Table 9.1)	Develop a more holistic view of the drivers of long-term biological change in WP. Priority to include the turbidity information from Wilkinson et al. Chapter 2 to refine Stage 1.	Expand existing analysis of trends and change point data for 3 fish species to include biological data from long-term bird counts and catch data for other fish species, along with additional variables such as seagrass cover and turbidity.	For many coastal and estuarine systems we currently lack the knowledge to link particular environmental drivers or events to observed biological changes. Long-term data sets, when appropriately analysed, can provide this information. This understanding is essential if we are to sustainably manage coastal and estuarine environments and protect their valuable ecosystem services in a changing world	Chapter 7 Jenkins.
Theme: Functional links between organisms and habitat						
S.20	3	Carry over from WP review (R.30)	Updated and finely scaled habitat mapping and description. Identify suitable scale for future habitat mapping.	Surveys to identify areas of functional importance, e.g. ecosystem engineering and biogenic structures. Determine the features that support high benthic biodiversity in WP.	Biomass estimates of macrofauna, particularly those that are important for nutrient cycling. This information feeds into geochemical models.	WP review Chapter 7
S.21	2	Carry over from WP review (R.31)	Mangroves and saltmarsh as habitat for animals and plants.	Role played by coastal saltmarsh and mangroves in providing habitat and food (i.e. organic carbon) for saltmarsh fauna, including invertebrates. Understand links between these habitats and adjacent soft sediment habitats. Needs investigations into the dependence of habitats for certain life history stages of both invertebrate and vertebrate species. Should include assessments of exchange processes of particulate and dissolved organic matter.	Essential to obtain an understanding of the ecosystem structure and functions provided by species in particular habitats. Such knowledge is essential to evaluate how any environmental changes would affect the functioning of ecosystems or parts thereof, even if they are not in the direct path of any disturbance event. Will increase our knowledge in the relevance of this habitat heterogeneity for the biodiversity and ecosystem scale processes, and allow more coherent network design of protected areas. Some assessment already made for fish and invertebrates in mangroves, but not done for birds in either habitat, nor invertebrates in saltmarsh.	WP review Chapters 8,9; Chapter 6 Hurst

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No.	Priority	Status	Brief Description	Details	Justification/Benefit	Chapters
Theme: Species of particular interest						
S.22	2	Carry over from WP review (R.33)	Determining the locations and timing of fish spawning.	Fish egg and larvae sampling has only occurred in a limited way in the southern part of WP. Bay-wide sampling at monthly intervals over 1-3 years is needed to identify species, location and time of fish spawning (e.g. snapper).	There is little information at present on the importance of WP as a spawning area for fish species, and also the key localities and timing of spawning. This information is crucial to the management of important fish and fisheries, and would indicate when and where sensitive egg and larval stages may be exposed to poor water quality. This work would also contribute to the biodiversity assessment of WP.	WP review Chapter 11; Chapter 7 Jenkins
S.23	2	New	Broad survey of birds using aquatic and saltmarsh habitats around the coasts of WP.	Existing surveys focused strongly on birds that use intertidal areas, and generally gather at defined roosts at high tide. An additional suite of birds inhabits a range of other wetland habitats around the coasts, and little systematic information has been collected on most of those species e.g. ocean beaches, mangroves, creeks and their estuaries. Species include some listed as threatened nationally (e.g. Lewin's Rail, Orange-bellied Parrot while other species are of interest because of their specialised habitats and restricted local distributions (e.g. Common Sandpiper in narrow creeks and beaches.	Marked differences have been noted anecdotally between the bird faunas of saltmarsh on French Island and the mainland, and it has been speculated that some species may have been lost from French Island (e.g. Superb Fairy-wren and Striated Fieldwren) because of the abundance of feral cats, while others may have benefited from the absence of foxes on French Island. More systematic research and documentation is needed to explore these relationships further. This could form part of the coastal biodiversity study proposed in Chapter 6 (R6.3).	Chapter 8, Loyn et al.
S.24	2	New	Review approach for long-term monitoring of waterbirds in WP. Explore viability of reinstating 5 vs 3 seasonal counts, and novel methods including drone surveys of low-tide feeding habitats for swans, shorebirds and other waterbirds.	Continue long-term monitoring of waterbirds in WP, and explore the viability of recommencing spring and autumn counts to give more information about seasonal variation (5 vs 3 seasons). Drones provide a new opportunity to getting a systematic handle on the distribution of waterbirds at low tide.	The BirdLife Australia survey has provided important insights into waterbird population longitudinal trends (increased, decreased or remained stable over time), and allows us to focus on possible reasons for the changes or stability. Monitoring can detect species shifts in response to habitat change - for instance showing change of species in the Corinella segment with seagrass loss.	Chapter 8, Loyn et al.
S.25	2	New	Ecological research on locally breeding shorebirds and terns.	Locally breeding shorebirds and terns are generally present in low numbers and rely on limited numbers of breeding sites, mainly on or near beaches. Some have prospered in recent years, notably Pied Oystercatcher (breeding mainly on fox-free French Island) and Hooded Plover (breeding on ocean beaches of Phillip Island: this species has benefited from fox control and targeted management). Other species have declined (e.g. Fairy Tern) or have precarious localised populations.	Further research is needed to determine whether they would benefit from targeted management interventions.	Chapter 8, Loyn et al.

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No.	Priority	Status	Brief Description	Details	Justification/Benefit	Chapters
S.26	2	New	Understand the trajectory of non-harvested fish communities of particular interest.	Synthesise fish biodiversity surveys conducted in WP Marine National Parks and Marine Sanctuaries.	Sygnathids including pipe fish, sea dragons and seahorses are associated with seagrass species, and these fish are currently listed as threatened under Victoria's Flora and Fauna Guarantee Act 1988 (FFG Act).	Chapter 7 Jenkins.
Threats						
Theme: Toxicants						
S.27	2-3	New. Relates to R.37 (Carry over, commenced, Table 1.1)	Investigate pesticide effects on key fauna and flora of WP in addition to seagrass and mangroves, with a view to developing locally relevant guidelines.	Further ecotoxicological testing on locally relevant species is needed for the common pesticides that have been detected throughout this research program. Furthermore, an understanding of mixture effects is also needed to more fully evaluate risks. Seagrass and mangroves commenced.	There are currently no guideline values for many of the commonly-used pesticides that have been detected. This makes it difficult to understand the risks posed to local flora and fauna by the elevated concentrations and complex mixtures of pesticides. This project will contribute to the development of WP specific toxicant guidelines.	Chapter 5 Myers et al.
S.28	3	New	Assessment of risks from new and emerging contaminants.	An initial screening of waterways for Pharmaceuticals and personal care products (PPCPs) is recommended, with a focus on areas in the vicinity of any substantial wastewater discharges or reuse.	There is increasing evidence of PPCPs in Victoria's waterways but a lack of data to help in understanding risks posed by these toxicants in WP.	Chapter 5 Myers et al.
S.29	3	New	Investigate the role of farming practices on the transport of pesticides to WP.	A study assessing the role of application methods on pesticide movement into WP catchments and the bay is recommended.	Understanding chemical transport pathways is needed to develop management strategies that reduce the concentrations and occurrence of pesticides in WP.	Chapter 5 Myers et al.
Theme: Harvesting						
S.30	3	Carry over from WP review (R.40)	Effect of shoreline harvesting on invertebrates.	Determine degree and impact of recreational harvesting on intertidal reefs and mudflats (i.e. bait pumping) around WP.	This information might link to changes in enforcement in future. This item is considered a low priority for substantial investment, but might be appropriate as a student project through one of the tertiary institutions.	WP review Chapter 13
S.31	1	New	Determining the locations and timing of Elephant fish reproduction, and better understand the decline in the Elephant Fish fishery.	Given the apparent decline in Elephant Fish, priority should be given to determine habitat use for spawning females and early juvenile through field surveys. Review the status of the larger, offshore stocks of Elephant Fish as a comparison with that in WP to further investigate the apparent decline in the WP fishery over recent years.	In terms of the managing the sustainability of Elephant Fish population it is important to know whether the recent decline observed in WP is also reflected in the larger offshore stock or whether this is a localised trend.	Chapter 7 Jenkins

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No.	Priority	Status	Brief Description	Details	Justification/Benefit	Chapters
S.32	2	New	Increased monitoring of WP fisheries. Improved information of the status of fish stocks within WP to guide fisheries management.	In species such as Gummy Shark and Elephant Fish with very low bag limits there is a difficulty in interpreting catch rates, and collection of detailed data on discards is recommended so that catch rate can be estimated more accurately. Confusion over the use of 'partial length' to measure Gummy Sharks is also a cause for concern as it can lead to the retention of undersize sharks. It would also be desirable to have pre-recruit surveys for King George Whiting in WP rather than relying only on results from Port Phillip surveys.	Improved information on status of fish stocks will enhance management of fisheries in WP.	Chapter 7 Jenkins
Theme: Climate Change and changes to habitat quality						
S.33	3	Carry over from WP review (R.41)	Vulnerability of intertidal reefs to sea level rise.	Determine the vulnerability of intertidal rocky reefs to sea-level rise and the capacity for migration of intertidal fauna and flora.	While intertidal reefs comprise a small area of the WP intertidal zone, they are a feature of the highly visited Mushroom Reef Marine Sanctuary and support a range of biota, adding to the diversity of marine life in the bay. Permanent inundation as a result of sea level rise poses a threat, but is less immediate than trampling and illegal collection.	WP review Chapter 13
S.34	2	New	Effects of sea level rise on seagrass	Modelling the colonization of seagrass under sea level rise.	Can give new insights into adaptation capability of seagrass under future climate change scenarios. Some preliminary scenarios will be run through current seagrass ARC but further work will be required.	Chapter 2 Wilkinson et al.
S.35	2	Carry over from WP review (R.38)	Investigate climate change effects on fish, including eggs and larvae. Toxicant work already underway.	Investigation of the tolerances of fish species to increased temperature associated with climate change (and the interaction with reduced water quality from toxicants in north of the bay). Suitable indicator species: blue spot goby (estuaries) and flat headed gudgeon (freshwater).	The early life stages are the key to sustaining healthy fish populations, but are also the most vulnerable to changes in water quality through climate change (or toxicant input).	WP review Chapter 3 & 11; Chapter 5 Myers et al
S.36	2	Carry over from WP review (R.43) and new	Factors determining roost selection in shorebirds, including the role of human disturbance. Explore options for maintaining and restoring roosting sites in light of sea level rise.	Identify primary factors determining roost site selection, threats and appropriate management action. The artificial sediment mound at Long Island (near Hastings) was a valuable high-tide roost for many years but appears to have become unsuitable. Fairy terns are known to have used artificial sediment mounds elsewhere and their status as a breeding species in Victoria has become precarious, mainly reliant on one or two sites on French Island.	Human activity is increasing in the bay, and disturbance at high-tide roosts an important issue. Sea-level rise and possible port developments are likely to exacerbate the situation, but also provide opportunities to restore or create new habitat by judicious and informed use of dredge spoil or other material.	WP review Chapter 12; Chapter 8 Loyn et al.

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