Understanding the Western Port Environment

A summary of current knowledge and priorities for future research

State Government Victoria

Port Phillip & Westernport CMA

Melbourne Water

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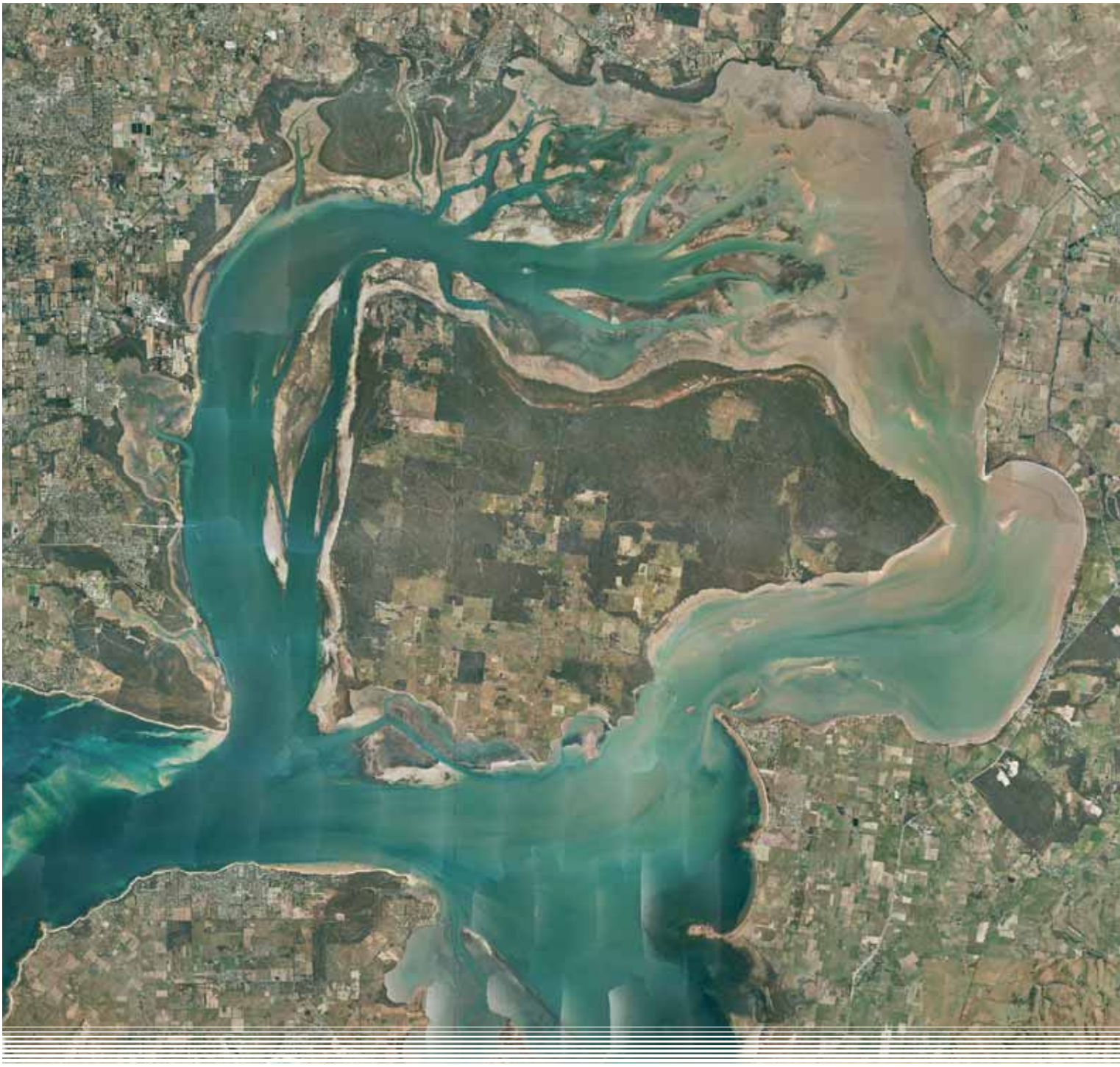
Executive Summary



Western Port is special

Western Port is a unique feature on the Victorian coast, a large, semi-enclosed embayment on an exposed coastline, formed by complex geological processes (Figure 1). Superficially similar to Port Phillip Bay, it is more complex than its western neighbor, with a greater tidal range, extensive intertidal mudflats, and two large islands (Phillip Island and French Island). The tidal flats are cut by deep channels, with several catchments draining (some artificially connected) into the northeastern and eastern parts of the bay. All of this makes for complex oceanographic circulation. Much of its coastline is fringed by mangroves and saltmarshes, and there are extensive seagrass meadows on mudflats and below the low tide level.

Scattered rocky reefs add to the diverse mix of habitats, and nearly 40 years ago (when it was last measured), Western Port had a much more diverse marine fauna than Port Phillip Bay. Western Port also provides extensive habitat for shorebirds, with much of its shoreline included in Ramsar sites that are designated for their importance for international migratory birds. The bay's waters also provide a range of ecosystem services, including nutrient cycling and support of fishing and other recreational activities, and Western Port has hosted an important commercial shipping port for many years.

Western Port has 3 of Victoria's 13 Marine National Parks within its boundaries and the Mushroom Reef Marine Sanctuary just outside its western entrance.

Western Port and its surrounds have also been recognized internationally, with UNESCO's Man and the Biosphere program designating it as one of just over 500 Biosphere Reserves around the world, which combine outstanding natural values with intense interactions with human populations (www.biosphere.org.au). This places it alongside 13 other such reserves in Australia, including Wilsons Promontory, Uluru-Kata Tjuta, and Kosciuszko National Parks.

Further detail about the Western Port environment and its importance is provided later in this executive summary and even more in the full report *Understanding the Western Port Environment*.

> *In general the invertebrate fauna of Westernport Bay contains three to four times the total number of species present in Port Phillip Bay and includes the majority of the species which occur in that Bay.*
> Shapiro (1975)

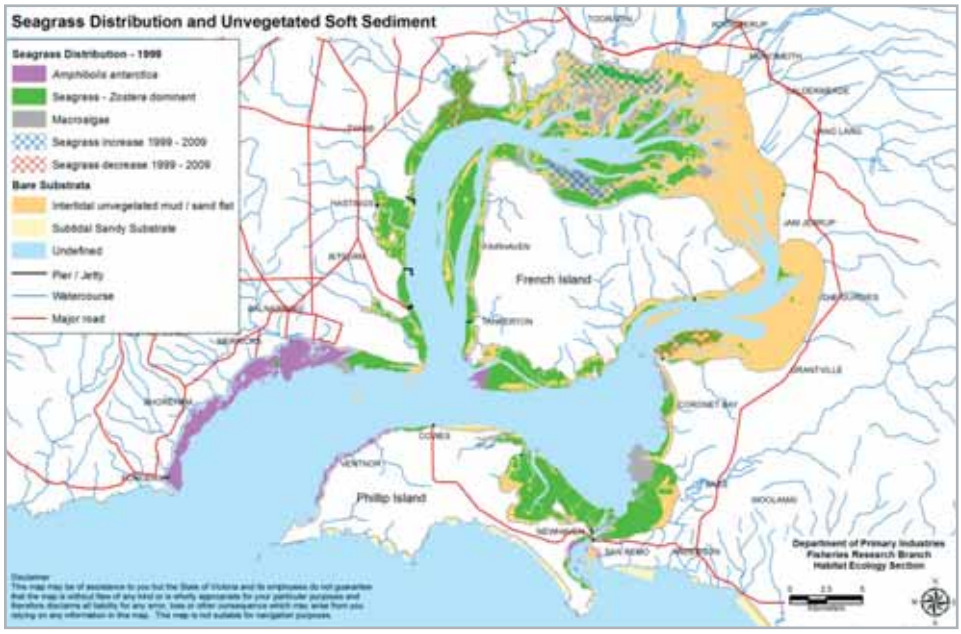


Figure 1. Western Port in 1999, showing intertidal mudflats, areas of seagrass, and subtidal unvegetated sediments. Note that much of the area shown as "undefined" is thought to be subtidal bare sediments.

Times are changing: the need for a review

Western Port is under pressure

As for every embayment near a major urban centre, human uses put pressure on Western Port. Over the past 200 years, the Western Port environment has undergone significant changes e.g. vegetation clearing within the catchment, draining of the expansive Koo Wee Rup swamp and the progressive growth of agriculture, industry and residential areas. Given the close connection between the health of the catchment and the health of the bay, dramatic changes such as these are expected to put considerable pressure on the marine and coastal environment.

The expansion of Greater Melbourne means that increasing numbers of people are moving to live along the northern shores of the bay, bringing substantial land use changes that may alter the quantity and quality of river discharges and increases in recreational use of the bay, including fishing. Expected changes to Victoria's climate will bring rising sea levels, changes to catchment discharges, altered bay water chemistry, and changes to wind and storm patterns. Predicting the effects of these changes with confidence requires re-evaluation of current scientific understanding to assess its applicability under new climatic regimes and new scientific research to understand their impacts.

Western Port has a wide range of values to Victorians, and much of its seabed and coastal area – including these environmental assets are subject to multiple uses, many of which can threaten other environmental uses and underlying ecological processes. The key is balancing the these different values and uses and principles of Ecologically Sustainable Development. With a long term strategic policy and management emphasis relating to Western Port's environmental assets, social and economic considerations also need to incorporated into the development of any future policy, strategies and plans for Western Port.

Why this review?

Nearly 40 years ago, Western Port was one of the best known parts of Victoria's marine environment.
The Westernport Bay Environment Study (the "Shapiro" report) in the 1970s involved over 100 people and was intended to be the first stage of a long-term research and management plan. It was followed by other reviews, including one by EPA Victoria about 15 years ago, and another in the past decade by the Coastal CRC, a CSIRO review of sediment processes around the same time, and a recent ecological review focused on the Western Port Ramsar site (prepared for the federal Department of the Environment, Water, Heritage and the Arts).

Since the Shapiro report, there have been big changes to the catchment of Western Port. There is remarkably fast population growth in Cardinia and Casey Shires. The rate of urban expansion in the Pakenham–Cranbourne growth area is the fastest in the state. From 1996 to 1999 this area accounted for 43% of all residential development in growth areas across Melbourne, and Casey was the fastest-growing municipality in the metropolitan area (DSE 2005). Along with this have been changes to land use in the mixed urban-agricultural land around Western Port. Management agencies have also been active, with commercial fishing being phased out, and various remedial actions taken in the catchments. We also know that there have been some big changes to the Western Port ecosystem over this time, but there has been no systematic examination of these changes or the currency of our scientific understanding. We also need to consider climate change – our current human actitivies (and past emissions) are expected to result in significant future change to marine waters, and Western Port will be vulnerable to rising sea levels with much of shoreline sloping very gently. Other climate effects, such as ocean acidification, have the potential to produce very large disruptions.

There is widespread agreement amongst stakeholders about the need for an expert review of the scientific knowledge base that underpins current management of Western Port. An updated and consolidated understanding of Western Port will better inform natural resource management, environment protection, planning, on-ground works and future research. This has been highlighted in various strategic plans including the Port Phillip and Westernport Regional Catchment Strategy and more recently, the Better Bays and Waterways water quality improvement plan.

Of the range of potential management, we need to prioritize, and for this to be effective, we need to understand which threats are most serious and we need to be able to screen potential management actions to identify those that are likely to lead to successful outcomes. Identifying and screening possible policy settings and management actions of course involves a range of social, economic, and environmental considerations, but a clear understanding of the science linking threats, environmental change, and management actions provides a strong basis to inform these considerations.

Sound science is needed to help to design future developments to minimise the risks to the bay's critical ecological processes, to identify and plan the most effective management actions in the bay's catchment, coastal and marine environments to repair past damage and strengthen its resilience to future change, and to monitor ecological condition and manage environmental risks. Where possible, the scientific knowledge should clarify causal links between human activities and ecosystem condition and identify the ecological processes that have the strongest influence on Western Port's ecosystem.

About half of the catchment (46%) is utilised for dairying, with all agriculture occupying 68%. Forests occupy 20%. Only about 1% is used for urban subdivision, with a total of 8000 urban houses (0.5 house per hectare) including about 3200 holiday houses that are seasonally occupied. The present population in the Study area is about 45000. Western Port at the time of Shapiro

A good understanding will also inform the review and setting targets for Western Port's condition. An important aim of current management is to maximize the "ecosystem health" of Western Port, but it is important to set realistic targets that reflects not only that "health", but a balance in the way that the community values different uses of Western Port (e.g. State Environment Protection Policy Schedule F8 (Waters of Western Port and Catchment) No. S192. Gazette 2/11/2001). That balance is necessary, because some of these uses are conflicting.

Ecosystems change naturally, but humans have also changed ecosystems, and some of these changes, both natural and anthropogenic, are not easily reversible. We must be realistic about what is now achievable in terms of the state of Western Port. Targets for Western Port could relate to when Western Port reached its current form 6000 years ago, or when Europeans first arrived, or the time of the Shapiro report in the 1970s, or we could base these around its current condition. Some of the changes to Western Port and its catchment are irreversible, so we will never return to some of the past states, and we know that climate change will bring changes beyond our local control, requiring adaptation, rather than mitigation. A robust scientific understanding of ecosystem processes will help us to understand what ecosystem states are possible and which ones are not.

This review is not 'reinventing the wheel', but builds on previous reviews and strategies in the context of new (and broader) research and current pressures. Outcomes and the future research directions that it recommends will help us to better predict the vulnerability of Western Port's marine ecosystems to future changes in climate, population growth and land use. Outcomes, including subsequent research, could inform:

- The tools (e.g. legislation, policy, natural resource management strategies, plans and/or targets) that guide activities to protect the health of Western Port into the future
- Sound investment in catchment, coast and marine environment protection and improvement activities
- Environmental regulation of land and coastal development
- Reporting on the state of the Western Port environment.
- Wise investment in environmental research

Scope of this review

Some threats to Western Port have been known for many years, and there is ongoing mitigation of these threats, which has been informed by our current understanding of Western Port. These actions are summarized in regional management strategies such as Better Bays and Waterways (Melbourne Water 2009a) and the Port Phillip and Westernport Regional River Health Strategy (Melbourne Water 2007). This review is broadly asking how Western Port has changed, what the current threats are to this ecosystem, and what kind of scientific knowledge we need to be able to effectively manage these threats into the future:

1. *What are the knowledge gaps?*
2. *Which knowledge gaps are critical to underpin management decisions and agency prioritization?*
3. *What research will fill these critical knowledge gaps?*

The focus is on the most important parts of the marine ecosystem within Western Port and along its coast – the "assets", and the threats to those components now, and into the future. The scope is marine waters, up to the high tide mark, including mangroves and saltmarsh. Threats to the assets can stem from activities within and beyond the marine and coastal zone, including pollution generated in the catchment. Although consideration is given to the threats that are likely to be most significant to the health of Western Port, this review is not a formal 'risk assessment'. In regards to threats, this document focuses on the scientific evidence linking these environmental assets and threats and the scientific knowledge that is needed to inform potential management actions at a large spatial scale and a strategic scale.

This review also takes an ecosystem focus to Western Port, looking at the whole bay, the connections between different parts of the ecosystem, and the significant and long-term threats to this ecosystem. This means that it does not focus around small-scale management issues in particular locations, jetties, channels, coastal infrastructure, etc. These activities and their management can be an integral part of maintaining a healthy ecosystem, but where the activity has a very localized effect, it has not been pursued in detail here.

The key environmental assets and threats at this scale to be considered by the review were initially identified from existing Western Port strategies and plans, and subsequently validated by a broad range of stakeholders, including government agencies, scientific experts and community groups. The review focuses on our scientific understanding of threats and assets, which can feed into future planning for Western Port, along with important social and economic information.

Review approach

The agencies

Many agencies contribute to managing the Western Port environment, including its catchment. It is a strength of this review that nine Victorian government agencies agreed on its importance, and collaborated in order to inform the identification of strategic natural resource management needs, as well as agree on the review process and outcomes.

The Western Port environment review was led by Melbourne Water and the Department of Sustainability and Environment, with funding also coming from the Port Phillip and Westernport Catchment Management Authority via the Victorian Investment Framework. A Victorian Government inter-agency advisory committee was also established to assist with the review, including additional representation from the Central Coastal Board, EPA Victoria, Parks Victoria, Department of Primary Industries, Department of Transport and South East Water.

The scientific team

The delivery model for the review was through a head scientist (and executive assistant) supported by an expert scientific team comprised of leaders in fields relating to Western Port's ecosystem components and threats.

The Head Scientist was Professor Michael Keough (University of Melbourne), assisted by Dr Rachael Bathgate (University of Melbourne), and the expert team was drawn from around Australia, and a mix of academic and government positions.

The experts were (alphabetically):

- Professor Paul Boon (Victoria University), an expert on coastal vegetation, particularly saltmarshes
- Dr Peter Dann (Phillip Island Nature Parks), who covered seabirds, shorebirds, marine mammals
- Associate Professor Sabine Dittmann (Flinders University), who was responsible for mangroves and soft-sediment habitats
- Professor Greg Jenkins (Department of Primary Industries), expert on coastal fish and fisheries
- Dr Randall Lee (EPA Victoria), an expert on oceanography and coastal processes.
- Professor Gerry Quinn (Deakin University), an estuarine scientist with particular expertise in catchment-coast links and ecosystem management
- Dr Jeff Ross (University of Tasmania), who contributed expertise on ecosystem processes, particularly nutrient cycling, and contributed to sections on soft-sediments
- Professor Diana Walker (University of Western Australia), Australia's leading seagrass expert
- Dr Robin Wilson (Museum Victoria), who covered soft-sediments and general biodiversity

These experts contributed chapters on their own areas of expertise, provided peer reviews of other chapters, and worked as a group to refine the lists of assets and threats, and to develop and scope the overall research priorities.

The structure

The scientific team first considered the list of Western Port assets and perceived threats that was developed by the lead agencies and other stakeholders. These lists were consolidated and used to structure the overall review. The experts then focused on components of the Western Port ecosystem, and summarized the current knowledge and the important research gaps, particularly those perceived as being important for effective Western Port management. The list of specialists' gaps was consolidated and reviewed as a group, resulting in some items being merged, others discarded, and a final list developed, taking into account the direct management benefit, the urgency of the research and the likelihood of success.

Figue 3. Approach used for this review

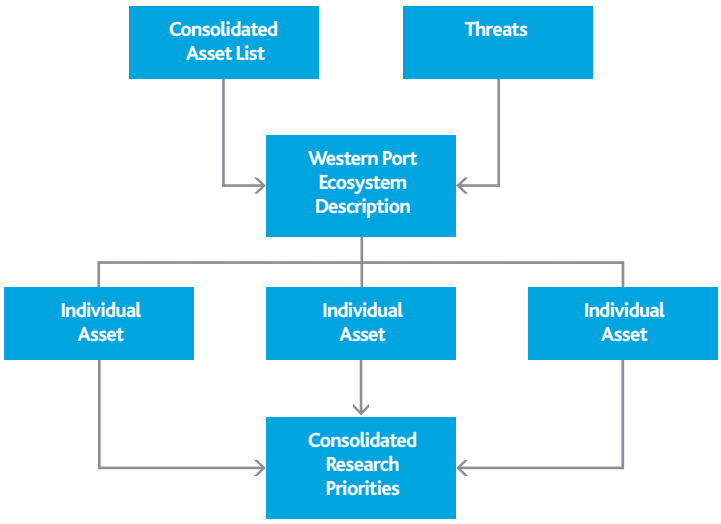


Photo courtesy Michael Keough.

Assets

As discussed above, our first step was to identify the "important" parts of the Western Port ecosystem – the assets[1]. These assets are diverse, and they range from individual species to ecosystem processes. Individual species may be valued because they are charismatic (e.g. seadragons), support important recreational activities (e.g. flathead, elephant fish) or are part of international agreements (migratory shorebirds) or important ecologically (e.g. seagrass). At the other extreme, apparently featureless areas of mud may be important for their important ecosystem service of cycling nutrients through the system. These categories are not mutually exclusive, for example, elephant fish are biologically significant and a valued fishery.

Our approach to assets focuses on three important components of Western Port:

- We recognise the role played by aquatic vegetation in supporting a wide range of species, so we separate the main habitats of Western Port into reefs and soft sediments, and we distinguish between soft sediments that are "bare", those with seagrass, and those around the edge of Western Port with mangroves or saltmarsh
- Ecosystem processes, particularly those involving nutrients, are crucial in maintaining the condition of Western Port, but are not uniquely associated with any particular habitat, so we consider them separately
- We recognise two groups of "iconic" species (in addition to seagrasses and mangroves). We consider fish separately, particularly those of conservation significance and those supporting recreational fisheries. We also consider birds and mammals as separate assets, including those subject to international agreements.

Threats

The wide range of activities in and around Western Port places pressure on the ecosystem of the bay, and these pressures are changing as Melbourne expands to the southeast, land use changes, and climate change accelerates. Some pressures can be reduced by actions in or around Western Port, others require actions further afield, and others cannot be mitigated by local management action. Some of these pressures are serious threats to Western Port's assets, while others are less serious. To identify the important threats, we need to understand the concerns of stakeholders, examine the scientific evidence about their severity, and understand the link between activities around Western Port and a threat to an ecosystem component.

We made an initial assessment of the major threats to particular components of the Western Port ecosystem, where these threats were assigned to one of three levels of priority or classified as too poorly known to categorise.

1 The term "assets" has a variety of meanings, and the way it is used in this report, to indicate important or valued ecosystem components, is different from how it is used in NRM planning in Victoria. For natural resource management planning, a map of statewide marine assets has been developed by DSE (www.dse.vic.gov.au). The study team for this review contributed to the Western Port component of this map.

What do we need to know about Western Port now?

It is remarkable what little knowledge about Western Port has been added since the 1970s. We have had several reviews, but not much new information. The number of scientific papers, which are a measure of robust new research, is tiny, particularly in comparison to neighbouring Port Phillip Bay. One important consequence is that much of our knowledge of the state of the Western Port ecosystem is now more than 35 years old.

As an ecosystem

Western Port has an extraordinary diversity of habitats, from rocky shores to deep channels with strong currents, mangroves, saltmarshes, seagrass beds, intertidal mudflats that are so important to shorebirds and subtidal soft sediments that harbour a diverse invertebrate fauna. Often these habitats are close together, resulting in areas of high diversity, such as the southeastern corner, where there is a diverse reef fauna close to rhodolith beds and important breeding areas for elephant fish. The proximity of these habitats means that they are interdependent.

The geography of Western Port also generates complex relationships within the bay, especially because its strong currents move sediments, nutrients and toxicants around, and provide a path for plants and animals to disperse. This means, for example, that nutrients entering the bay may be processed and removed in areas distant from where they entered. Some of Western Port's plants and animals also use different parts of Western Port during different stages of their life cycles, or only live part of their lives in the bay. While it is helpful to consider individual assets of Western Port or particular threats, we need to keep in mind the critical linkages within this ecosystem.

Physical processes and ecosystem function

Western Port is a large shallow embayment that is divided into five basins or segments by large islands and mudflats (~1/3 in area) and several narrow constrictions. Although generally well flushed by tides through the western entrance, wind forcing drives a prevailing clockwise circulation. The flow entrains catchment inflows and resuspended bay sediments, resulting in poorer water quality (and higher residence times) in the east. There is also short term variation in water quality over tidal cycles, most likely from an interplay between the mudflats and incoming ocean waters.

While system-wide hydrodynamics have been adequately described, our knowledge of the finer-scale hydrodynamics (at a basin scale), which is necessary to understand connectivity through the system, is much poorer. Well-calibrated hydrodynamics is a fundamental first step in building an understanding of the bay as a dynamic system.

Identifying the contribution of nutrients and sediments from the catchment, atmosphere and within bay processes is an important priority to inform management. We suggest that integrating a full range of bay models with other well-accepted catchment and airshed models would provide a more holistic picture of regional processes that would more accurately represent the present and future bay conditions and responses.

Figure 5. Water circulation in Western Port.
(Source: Hancock *et al.* 2001.)

Natural ecological processes underpin the important habitats and the diverse range of animals they support in Western Port. Understanding if and where nutrients accumulate in marine systems is an important element of any environmental management strategy, particularly where nutrients are considered a major threat. In Port Phillip Bay it is well established that the process of denitrification in subtidal sediments permanently removes much of the excess nitrogen that enters in the bay. Westernport Bay, however, is very different to Port Phillip Bay and our understanding of nutrient cycling there is inadequate. Over a third of Westernport is intertidal seagrass and bare mudflat yet we know little about nutrient transformation in these environments. Elsewhere in the world, similar mudflats have been shown to be either sources or suppliers of nutrients to the marine ecosystems.

The decline and limited recovery of seagrass in the eastern section of the bay is symptomatic of nutrient and sediment loads exceeding the system's capacity to process and assimilate them. However, our understanding of the ecological thresholds of the major habitat forming primary producers such as seagrass and the consequences of a habitat shift for nutrient and sediment dynamics is limited. In the absence of this knowledge our ability to prioritize nutrient and sediment reduction strategies is constrained.

A multi-stage research program is proposed that would develop a nutrient and sediment budget for Westernport, identifying key areas and habitats for the transformation and removal of nutrients and the settlement and resuspension of sediments. The recommended stages will provide for a rapid assessment, which will inform an assessment of the need for detailed formal measures of nutrient cycling, and the need for a formal process based model for Westernport. Such a model, coupled with improved sediment and hydrodynamic models, will allow detailed exploration of the benefits that would be expected from alterations to catchment inputs of nutrients and sediments, but also the capability to predict the response of the Westernport ecosystem to future climate scenarios.

Figure 6. *Trypaea australiensis*, one of several ghost shrimp species occurring in Western Port. (Photo: M. Marmach, Museum Victoria.)

Figure 7. *Zostera nigricaulis*. (Photo: M. Keough.)

The major habitats

The water column

The water column in Western Port is inhabited by microscopic single-celled organisms (phytoplankton), small animals that drift passively with the currents (zooplankton), and larger, passively drifting animals such as jellyfish. The phytoplankton are important indicators of environmental impacts such as elevated nutrients, and changes can be monitored by measuring the concentration of phytoplankton pigments in the water column. The limited information on phytoplankton species available indicates that diatoms tend to be much more common than dinoflagellates. The zooplankton of Western Port, unlike Port Phillip Bay and the open coast, is dominated by one species of small crustacean belonging to the copepod genus *Acartia*.

We identify several research gaps, including a better understanding of the species composition, assemblage structure and ecology of phytoplankton. Information on species dominance patterns and how they change spatially and temporally (both within and between years) within Western Port is completely lacking, as is their behaviour with respect to identified nutrient sources and circulation patterns within the bay. In the case ofzooplankton, there is relatively little information on marine invertebrate larvae and the biology of the larvae of most species is not known. A major knowledge gap also exists for jellyfish; only one brief study on a single species has been undertaken.

Mud

Soft sediments are the prevailing habitat in Western Port, covering about two thirds of the bay. The area of unvegetated sediments has increased following the loss of seagrass beds. The extensive intertidal flats are important foraging grounds for shorebirds. Several hundred species of infaunal and epifaunal organisms have been recorded, including a high diversity of ghost shrimps, brachiopods that are considered living fossils, rare rhodoliths and various species that are listed as endangered. Benthic fauna occurs in defined assemblages depending on sediment characteristics and water depth. Fewer species have been found at sites with a history of disturbance and eutrophication.

Most of the research on soft-sediments in Western Port is several decades old, and a survey to assess the current biodiversity in comparison with past records and adjacent bays is needed. Such information would inform assessments of various disturbance effects and invasive species. Other areas requiring research attention are functional roles of benthic organisms and how they contribute to the productivity, sediment dynamics and nutrient fluxes in Western Port.

Seagrasses

Seagrasses are unusual aquatic flowering plants that also have an important function as ecosystem engineers, being involved in sediment movements, nutrient and energy transfer, and the provision of habitat for a diversity of animals. This has led to a common view by a range of stakeholders of the importance of seagrass in protecting the health of Western Port. In Western Port, the most widespread and abundant seagrasses are *Zostera* species, which occur on intertidal flats and subtidally in many areas in northern and eastern Western Port. The south-western parts of Western Port support areas of *Amphibolis*, which may also be ecologically important. There was extensive loss of seagrasses in the 1970s, followed by some recovery, but large areas that lost seagrass have not recovered, and recovery has been poor in areas where water quality is a concern.

> *There is no evidence that the seagrass of Westernport Bay is infected by 'wasting disease' which has caused the destruction of large areas of seagrass in the Northern Hemisphere.*
> Shapiro (1975)

A patch of *Avicennia marina* near Tenby Point.
Photo courtesy Adam Pope

It is generally agreed that greater coverage of seagrass in Western Port is desirable, but there are several knowledge gaps preventing us from identifying the best way to achieve this. First, and most practically, we do now know which species of *Zostera* are present in Western Port, so we do not know the extent to which we can make use of earlier work or use results from Port Phillip Bay to inform management strategies for Western Port. We know that turbidity limits the areas in which seagrasses can grow in Western Port, but we are not sure whether the limits come just from suspended sediments or a combination of sediments and nutrients. Understanding the limits for seagrasses will allow us to determine how much water quality needs to improve in Western Port, and which aspects of water quality should be targeted.

Even if water quality improves, we do not know how seagrasses recolonise suitable habitat, and an important research need is to understand these processes in Western Port. Knowledge gaps around seagrass were first highlighted by Shapiro and colleagues, who identified the factors limiting seagrass recolonisation as an important gap; their attention was focused on sediments, with less attention to nutrients. Their work preceded the loss of seagrasses.

Subsequent reviews have continued to highlight seagrass recovery. Recent work has revisited the classification of seagrass, creating an important knowledge gap, and our review has also proposed a formal process to identify water quality needs for seagrass.

Mangroves

Western Port harbours some of the southernmost mangroves in the world, composed of a single species, *Avicennia marina*. Mangrove forests line much of the shore of the bay, and occur in the three Marine National Parks. Historical comparison shows some loss, especially near Hastings. Localised destruction, disturbances and changes in the sediment budget of the bay have contributed to changes in mangrove distribution.

There are no recent records of biota associated with mangroves in Western Port, apart from fish frequenting the mangrove fringe and forest, and a biodiversity survey is needed to understand links between biota and mangrove disturbance history. Further research gaps relate to the functional relevance of mangrove biota for coastal ecosystems in Western Port. Habitat loss and fragmentation are serious threats to mangroves, and landward retreats are needed to prevent mangrove reduction with sea level rise.

Field experiments, undertaken in selected plots of Heterozostera tasmanica *and associated plants, could be designed to elucidate the following points:*

(i) The limiting factors of rate and total sedimentation of which seagrasses are tolerant, and the point at which photosynthesis is inhibited or ceases.

(ii) The limiting conditions of wind disturbances and water movement which control particulate deposition among the seagrasses, and the factors, including that of time, involved in the eventual incorporation of sediments into the seagrass meadows.

(iii) The time and biological factors involved in the regenerative succession in plots from which seagrasses have been partially or entirely removed.

Shapiro (1975)

Photo courtesy Michael Keough.

Saltmarshes

Saltmarshes occur around much of the coast of Western Port, generally between the mangrove fringe on the seaward side and more terrestrial vegetation, such as Swamp Paperbarks and Manna Gum woodlands, on the landward side. There is about 1000 ha of saltmarsh in Western Port, which is about the same area as there is of mangroves and is among the largest tracts of saltmarsh in Victoria. A number of the larger saltmarshes in Western Port occur in protected areas, such as the Yaringa (980 ha), French Island (2800 ha) and Churchill Island (670 ha) Marine National Parks. Saltmarshes in Western Port are likely to be very vulnerable to sea level rise and other consequences of climate change, especially rising air and water temperatures. Saltmarshes have been progressively lost already, mostly because of development for agriculture and industry, around the western and northern shores of Western Port.

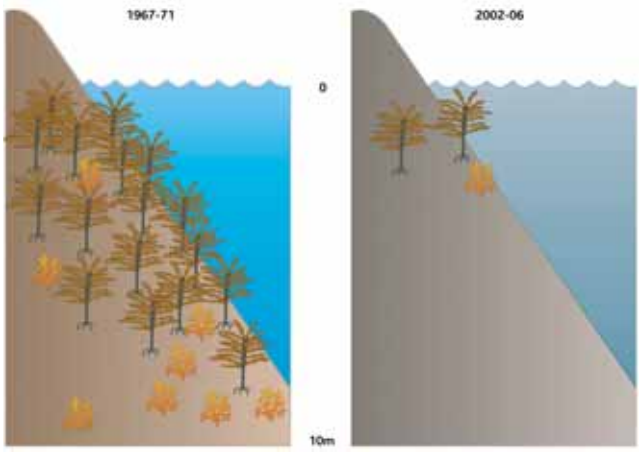
We identify several research gaps, including a better understanding of the ways tidal inundation affects waterlogging and salinity regimes in saltmarshes, and in particular how they affect the saltmarsh plant communities that provide food and habitat for terrestrial and aquatic animals. Much more research is needed on the way that terrestrial (e.g. bats and bushbirds) and aquatic (e.g. waterbirds and shorebirds) animals use saltmarshes. The susceptibility or resilience of saltmarshes to threats such as nutrient enrichment, oil pollution, weed invasion (e.g. by *Spartina*), altered salinity and hydrological regimes, and climate change is also an important research gap.

Rocky reefs

Rocky reefs occupy only a very small part of Western Port, but three areas are notable; Crawfish Rock is an unusual habitat with very high biodiversity, a small reef near San Remo is listed under the *Victorian Flora and Fauna Guarantee Act 1988* for its opisthobranchs, and intertidal reefs along the south-western coast, particularly Honeysuckle Reef, also have high diversity. Intertidal reefs in Western Port are likely to be very vulnerable to sea level rise. There is evidence that there has been some loss of diversity at Crawfish Rock, most likely a result of high turbidly in the North Arm.

We identify several research gaps, including a better understanding of the biodiversity of deep channels and understanding of the impacts of recreational activities on intertidal reefs, but the two most important gaps are an assessment of risks from sea-level rise and a better understanding of the sediment-based water quality threshold for algae on reefs in the North Arm. We suggest that these thresholds do not need to be resolved immediately, because seagrasses will be more susceptible to light reduction, and improvements to seagrass habitat will flow through to improvements to reef algae.

Figure 11. Changes to depth distribution of kelps and red algae at Crawfish Rock between the early 1970s and mid 2000s. The figure shows patterns for high current flow areas at this site, with algae now confined to very shallow depths. Redrawn from Shepherd *et al.* (2009)



Red-necked Stint.
Photo courtesy Annette Hatten.

Iconic fauna

Fish

Western Port has a high diversity and productivity of fish, especially the small fish (including juveniles of important fishing species) associated with the extensive seagrass beds. Western Port is also an important habitat for pelagic species such as Australian anchovy, for a number of species of conservation significance, and is a breeding habitat for species such as elephant fish and school shark. Western Port also supports a very significant recreational fishery.
The greatest threat to fish in Western Port is the loss of their habitat, in particular the potential loss of seagrass habitat. Other threats include water quality effects on eggs and larvae, the potential for over-fishing, and climate change impacts such as increased water temperature.

Figure 12. Elephant fish.
(Photograph © Bill Boyle/OceanwideImages.com)

We identify several research gaps, including: a better understanding of the relationships between fish and less studied habitats such as *Amphibolis* seagrass, studies of eggs and larvae to determine spawning areas and also their sensitivity to toxicants; studies of the water quality requirements of estuarine fish, and, the continuation and extension of recreational fishing monitoring to inform the sustainable management of this increasingly important activity.

Birds and marine mammals

Western Port is of international significance for aquatic birds. Its importance for birds is reflected in the abundance and diversity of species, the breeding populations of some species in the Bay or nearby (some unusually large), its importance as a drought refuge for waterbirds and its use as a non-breeding area for migrant shorebirds from the northern hemisphere and New Zealand.

The makes a significant contribution to Australia's obligations under a suite of international treaties and agreements. It is also designated as part of the global network of Birdlife International's Important bird areas. In contrast, although a variety of marine mammals have been reported in Western Port, the bay appears to have relatively little importance for that group of animals.

The greatest threats to birds in Western Port are loss of habitat, reductions in food supply through extraction (particularly fish-eating birds) and seagrass loss (most species) and high levels of disturbance from human recreational activity (shorebirds). Habitat loss includes loss of feeding areas and roosting sites through sea-level rise. The greatest threat to the commonest marine mammal in Western Port, the Australian Fur Seal, is sea-level rise, which could greatly reduce the size of the breeding colony.

We identify several research gaps, including a better understandings of the decline in fish-eating birds, the relative significance of shorebird and waterbird intertidal feeding areas, the factors involved in roost selection in shorebirds (including the role of human disturbance) and the effects of sea-level rise on shorebirds and waterbirds.

Common threads

Water quality emerged as a consistent serious threat across most ecosystem components. Suspended sediments are the most important aspect of water quality, and have been a target of management action for some time. It is not clear how much sediment comes from catchments, compared to resuspension and shoreline erosion, with the contribution of shoreline erosion an important uncertainty. Nutrients may be an issue in some parts of Western Port (around Watsons Creek and in the Corinella segment), but the evidence is somewhat equivocal. Toxicants, particularly those associated with the eastern catchments, are a potential concern, although there is generally too little information to be able to determine whether they should be a focus for actions to improve water quality.

For individual components of Western Port, there are specific threats. For example marine invertebrates are collected from mudflats for bait, from reefs for food, and fish are taken from several areas, often in substantial numbers. Our assessment is that recreational fishing is an important threat requiring attention, while bait/food collection is a less serious issue on reefs and mudflats, and not an issue in seagrasses, saltmarshes or mangroves.

Climate change is a consistent theme, with rising sea levels a clearly serious threat for saltmarsh, mangroves, intertidal rocky reefs, and for shorebirds and water birds. Acidification is also likely to be important in the future, but there is some uncertainty about its severity, and actions in Western Port could not ameliorate this threat when it becomes important later this century.

What's new in the review and implications for management?

This review places a greater emphasis on the need to have a systems understanding of Western Port (i.e. how all the aspects of the environment work together) that can drive a targeted protection, rehabilitation and management program. In addition, as well as the traditional focus on sediment, more attention should be placed on understanding the threat from nutrients, toxicants and freshwater inputs from the catchment. This will facilitate the derivation of meaningful bay-wide water quality targets and the nature of management efforts e.g. standards for urban stormwater treatment, rural land management programs. The threat from climate change to certain species and habitats, as well as opportunities for management, also needs a greater focus.

Some knowledge gaps are not new; as described earlier, Shapiro and colleagues identified some science gaps, such as recovery of seagrasses and an understanding of nutrient cycling. Nearly 40 years later, these are still important gaps, with little progress in filling them over that time. Other recommendations reflect changes in scientific understanding; for example, recent use of molecular genetics tools has created uncertainties in our understanding of seagrasses in Western Port, there are recent signs of changes to some of the valued birds of Western Port, and our concerns about toxicants have moved from primarily hydrocarbons and heavy metals to the cocktails of toxicants that are part of the runoff from agricultural lands and some urban areas.

Figure 13. Asset-threat matrix, with pooled water and sediment quality.

Asset		Water and sediment quality	Habitat loss and fragmentation	Extraction and disturbance	Sea level rise	Temperature increase	UVB	Pests	Cumulative impacts
Habitat Groups									
Rocky reefs	Intertidal	Low Threat	Low Threat	Intermediate Threat	Intermediate Threat	Intermediate Threat	Intermediate Threat	Low Threat	Unknown Threat
	Subtidal	Intermediate Threat	Low Threat	Low Threat	Low Threat	Low Threat	Low Threat	Low Threat	Unknown Threat
Vegetated sediments	Mangroves	High Threat	High Threat	Intermediate Threat	Intermediate Threat	Low Threat	Unknown Threat	High Threat	High Threat
	Saltmarshes	Intermediate Threat	High Threat	High Threat	High Threat	Intermediate Threat	Unknown Threat	High Threat	High Threat
	Seagrass	High Threat	Intermediate Threat	Low Threat	Unknown Threat	Unknown Threat	Unknown Threat	Low Threat	Unknown Threat
Unvegetated sediments	Intertidal	High Threat	Intermediate Threat	High Threat	Intermediate Threat	Intermediate Threat	Unknown Threat	Intermediate Threat	High Threat
	Subtidal	High Threat	High Threat	High Threat	Low Threat	Intermediate Threat	Unknown Threat	Intermediate Threat	High Threat
Water column		High Threat	Low Threat	Low Threat	Low Threat	High Threat	Intermediate Threat	Low Threat	High Threat
Significant Fauna									
Iconic species	Shorebirds	High Threat	High Threat	High Threat	High Threat	Intermediate Threat	Unknown Threat	Low Threat	High Threat
	Seabirds	Intermediate Threat	Intermediate Threat	High Threat	Intermediate Threat	Low Threat	Unknown Threat	Low Threat	Intermediate Threat
	Marine Mammals	Low Threat	Low Threat	Low Threat	High Threat	Low Threat	Unknown Threat	Low Threat	Low Threat
Commercial/Recreational	Fish	High Threat	High Threat	High Threat	Low Threat	High Threat	Intermediate Threat	Intermediate Threat	Intermediate Threat
Emergent Features									
	Ecosystem Processes	High Threat	High Threat	Intermediate Threat	Intermediate Threat	High Threat	Unknown Threat	Intermediate Threat	High Threat

High Threat Intermediate Threat Low Threat Unknown Threat

Executive Summary

Eroding shoreline in south-east Westernport.
Photo courtesy Adam Pope

This review continues to confirm that the strongest management "levers" are catchment works, aimed at reducing the amounts of sediments, nutrients, and toxicants entering Western Port, with sediments previously acknowledged as a major issue. We do know that some of Western Port's shoreline is eroding, and may also be contributing substantial amounts of sediment to the bay, so another important management lever is to change land use and remediate these shorelines, particularly along the eastern shore. We also know that areas previously covered with seagrasses are another important source of sediment, but we do now know whether catchment and shoreline remediation will provide the conditions necessary for these areas to be revegetated (and consequently stabilized), or if other factors will limit the recolonisation of seagrass. Although we suspect that reducing suspended sediments will be the highest priority, there are some important questions about the significance of nutrients and toxicants, particularly for important primary producers in the bay.

Outcomes from the review suggest that, broadly speaking, the direction of current environmental management activities within the catchment, coastline and bay is likely to be beneficial to health of Western Port e.g. stormwater treatment, revegetation of riparian and coastal zones, wastewater treatment, 'best practice' rural land management. Most importantly what is clear, is that we do no have a firm picture of the extent to which we need to invest in each of these various activities, or the specific locations where these activities will lead to the greatest outcome for the bay. The targeted research program outlined in this review is expected to help provide this guidance, along with the establishment of critical management objectives, e.g., similar to the nitrogen load target for Port Phillip.

> *The total pesticide loads flowing into the Bay are extremely small although pesticide usage in some areas of the catchment is intensive. The significance to Westernport of an increase in these pesticide loads has not been determined.*
> Shapiro (1975)

Shapiro's emphasis was on industrial toxicants, such as heavy metals, rather than chemicals from domestic or agricultural use. We are increasingly aware of the effects of some of these chemicals at low concentrations, and their influence in Western Port is not known.

The most important science gaps are...

We developed a list of 43 research needs, which were then assigned to three priority categories. The highest priority research needs are those that are achievable, and would make an immediate difference to Western Port management actions. The other research needs include some that are critical but do not need to be done immediately (e.g. quantifying responses to some aspects of climate change), some that might not feed directly to management actions but would lead to more informed decision-making (e.g. determining how much Western Port has changed since its previous assessment 36 years ago, to help set more realistic goals for environmental condition), and research needs where several additional steps would be needed to realise management benefits.

The 12 highest priority research tasks fall into several groups:

Improving our understanding of **physical processes**. Suspended sediments, nutrients and contaminants are important threats to the bay and its assets, and understanding how this material enters Western Port and how it moves about the bay requires a sophisticated suite of models that can describe the complex patterns of water movement around Western Port. We have a partial understanding of these processes, but we need to

1. Obtain detailed and up-to-date bathymetry for Western Port
2. Calibrate hydrodynamic models to ensure accurate representation of water movement

The continued health of Western Port depends on important groups of plants – habitat forming species such as seagrasses and mangroves and algae that are responsible for important ecosystem services, particularly nutrient cycling, and we need to know relationships between these species and **water quality**, particularly **sediments** and **nutrients**. There is uncertainty about the extent to which nutrients may be an issue in northern parts of Western Port, and uncertainty about the relative importance of different sources of suspended sediments, so we need to

3. Determine a preliminary nutrient budget
4. Measure nutrient cycling in major habitats (unvegetated soft sediments and seagrass habitat)

Seagrasses are the most important habitat-forming species in Western Port, but to understand the loss and recovery of this habitat, we need to

5. Assess the degree of nutrient and light limitation of the major primary producers, seagrass and possibly microphytobenthos
6. Determine water quality targets for sediments and nutrients that support seagrasses (and possibly microphytobenthos)
7. Determine which species of *Zostera* are present in Western Port
8. Determine capacity for *Zostera* to recover and colonise new areas

Our review also found an important knowledge gap about the extent to which **toxicants** entering Western Port pose a threat to the marine environment, so we need to

9. Make an initial estimate of the risk from toxicants beyond discharge points

The remaining research gaps relate to **iconic species**, specifically the fish that are responsible for much of the recreational value of Western Port and the shorebirds and waterbirds that use Western Port so extensively. We need to:

10. Determine linkages between fish and habitats, to better understand the significance of changes from seagrass habitat to algae-dominated habitat
11. Determine the effects of recreational fishing on fish stocks
12. Examine the trends in abundance of fish-eating birds in Western Port.

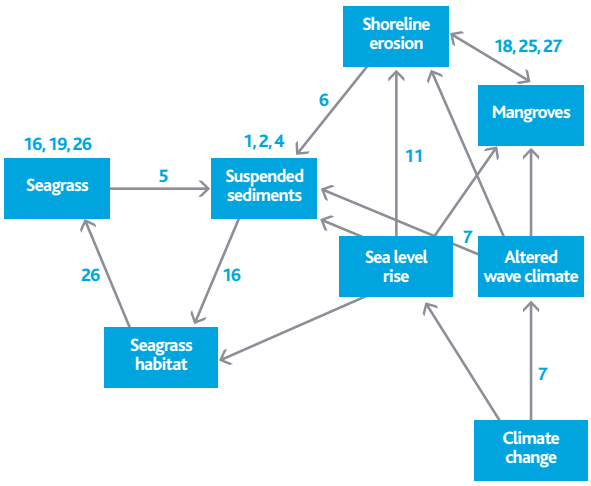
Photo courtesy Michael Keough.

Some are big, some are small tasks

These tasks vary in size. Some are small tasks that would have immediate benefits. For example, calibrating the hydrodynamic models for Western Port requires the short-term deployment of current meters, but would guarantee a high quality tool for predicting changes to water quality. Similarly, genetic studies to determine which seagrass species occur in Western Port can be done quickly and would remove an impediment that is preventing us from interpreting past work in the bay and making comparisons to Port Phillip Bay.

Other projects are more substantial, and would need longer times and greater investment. Determining water quality requirements for healthy seagrass, and determining the processes by which seagrasses recolonize suitable habitat are not trivial tasks. While it is not possible to scope this task accurately for now, comparable work in Port Phillip is requiring several years and investment of several million dollars. Generating an accurate sediment budget and determining risk from toxicants are similarly large tasks.

Figure 14. Example relationships between sediments and seagrass cover in Western Port, showing important functional relationships. The numbers in blue indicate numbered recommendations, and show the interdependency of research needs.



They are interdependent

The linkages that exist between many of these research priorities affected the prioritisation of some research needs and also made the scoping more problematic. For example, increasing the extent of seagrass requires an understanding of water quality requirements and some basic ecological information (6 and 8 above). To do this, we need to know which seagrasses are present (7), whether they are limited by sediments in the water or sediments plus nutrients (5), which in turn depends on whether nutrients are a serious issue for Western Port (3, 4). This information would also determine whether we can use seagrass data from Port Phillip Bay or need dedicated new research for Western Port.

One of our strong recommendations is the development of a coupled geochemical model, which would be capable of linking catchment processes, nutrient cycling, and sediment transport, and would allow accurate predictions of how water and sediment quality would change under various management scenarios, such as particular catchment works or shoreline remediation. This model would link directly to the understanding of water quality thresholds for seagrasses (and other primary producers) and seagrass colonisation processes, to enable us to link seagrass coverage in Western Port to these management actions.

If we knew the light and nutrient requirements for seagrass we could map the areas of Western Port for which water quality provides suitable seagrass habitat. These maps could be based on empirical measurements of light quality and nutrients, or they could be produced from the geochemical model for Western Port. A map of seagrass habitat would allow us to compare the actual distribution of seagrass with the area of suitable habitat. A substantial mismatch would indicate that processes other than water quality are important.

The other important knowledge gaps

The "other" recommendations are those that we assigned a slightly lower priority, although the fact that they survived our screening shows that, as a group, we still thought them important. They include work that we recommend is done, but not immediately. For example, it appears that rising sea levels will threaten major shorebird feeding areas and also most of the intertidal reefs. We need to document these effects and look at whether there is any scope for these habitats to migrate. We have a little time to do that, as the rate of sea level rise will not become significant in this regard until the latter parts of this century and beyond. Another important example is the need to understand just what Western Port is like now, rather than what it was like when last studied in detail 40 years ago – how have patterns of diversity changed? Have pest species become established? Are these changes reversible? The answers would not change what we do right now in Western Port, but they may affect our strategic plans and the kinds of targets that managers might set for future environmental condition in Western Port, and assessment of condition is part of an adaptive management framework.

The full list of important knowledge gaps is provided in Chapter 15 of the full report.

A research program

The 43 recommendations are not just a list; they fit into several broad themes, and they are related to each other. Some of the specific recommendations form a "chain" of research, in which an element feeds into or shapes another the next.

We see the set of recommendations as a coherent research program that focuses initially on the high priority research tasks, makes plans to do the critical, but not immediate tasks, and seeks opportunities to do the others.

This is consistent with the intentions of earlier reports, and with a goal of maintaining the Western Port ecosystem in the best state possible, while supporting a range of beneficial uses, and in the face of substantial externally driven changes.

1 Introduction

Michael Keough

Western Port is a unique feature on Victoria's coast — a large, shallow, semi-enclosed bay with a diversity of habitats. Superficially similar to neighbouring Port Phillip Bay, it differs in many respects, including oceanography, habitat diversity, biodiversity and the range of human pressures. It is surrounded by mixed urban and agricultural lands, and much of the natural landscape has been altered dramatically. Victorians use Western Port's waters for a wide range of recreational activities. There is also a long history of industrial use on its western side, commercial fishing until recently, and a long-term Defence presence along the western fringe.

Western Port's shores are fringed by sandy beaches, rocky reefs, intertidal mudflats, saltmarsh and mangroves, and these areas can be extensive because of the nearly 3 m tidal range. Out in the bay, extensive mudflats emerge at low tide. Until the 1970s many of these mudflats were covered by seagrasses. Below the low tide level, most of the seabed is soft sediments. Some areas are covered by seagrasses, with their diversity of fauna, while others are populated by a great diversity of invertebrates and fish, often different from those associated with seagrasses. Deep, steep-walled channels allow extensive tidal movements through Western Port, resulting in a variety of soft sediments: well-mixed oceanic waters at the southern end produce clean, relatively coarse sands, but towards the north-eastern parts of the bay the waters become more turbid and the sediments become finer. Rocky reefs, which are not common in Western Port, harbour a rich diversity of invertebrates and seaweeds.

The geology and geomorphology of Western Port have been comprehensively documented (Marsden and Mallet 1975, Bird 1993, Rosengren 1984). These physical attributes, along with water movement, meteorological and climatic factors, provide the underlying structural components of Western Port.

We have long been aware of the physical and biological diversity of Western Port and the diversity of uses we have for it, and this has been recognised in a variety of ways (e.g. as a UNESCO biosphere reserve). We have also been conscious that human activities pose a range of risks, which require careful management.

Effective management of marine and coastal environments must be informed by a strong scientific understanding that allows us to set targets, predict the consequences of particular actions, and make projections about future conditions. This was recognised in the 1970s with the commissioning of the first major environmental study into its waters. This comprehensive study — the Shapiro study (Shapiro 1975) — described catchment inputs to Western Port, summarised its geography and oceanography, provided detailed assessments of the diversity of its animals and plants, and gave some insights into its ecological processes. But in the following three decades there was a scientific hiatus with only scattered research on Western Port.

The need for a more comprehensive scientific approach was reaffirmed in the past decade, with a small desktop study that identified some research gaps and described a strategy for identifying science research needs.

Natural ecosystems change. Some of this change is natural, reflecting seasonal cycles, long-term weather patterns, and unusual events. Other change is linked to human activities, as the stresses that we place on natural ecosystems rise and fall when populations expand, land use changes, and our management of these environments alters. We are now beginning to plan for the climate change that seems inevitable, as the best scientific advice predicts changes in sea levels, rainfall patterns, temperature, and the acidity of oceans.

Change can be seen in Western Port. Humans have lived around and made use of Western Port since its formation 8000–9000 years ago; the Boonerwrung people have lived in this part of Victoria for many thousands of years, and two clans are particularly associated with Western Port's shores — the Mayone-Bulluk clan (top of the Mornington Peninsula and head of Western Port) and the Yallock-Bulluk clan (near the Bass River on the eastern catchment of Western Port). The Boonerwrung were semi-nomadic hunter-gatherers, moving around a well-defined tract of land according to the seasons, exploiting and managing a range of resources (DPCD 2011 and Figure 1.1). Since the first formal surveys of Western Port we have been aware of extensive changes to lands around the bay, including the draining of Koo Wee Rup swamp and loss of coastal vegetation, and also a widespread loss of seagrasses in Western Port. Changes are continuing along the coastline, and there are also major changes occurring in the Western Port catchment as Melbourne expands rapidly south-eastwards. These changes create a need for a flexible way of managing coastal environments, responding to new stresses. We must be realistic about what is now achievable. Targets for Western Port could relate to when Western Port reached its current form 6000 years ago, or when Europeans first arrived, or the time of the Shapiro report in the 1970s, or we could use its current condition. Some of the changes to Western Port and its catchment are irreversible, so we will never return to some of the past states, and we know that climate change will bring changes beyond our local control, requiring adaptation, rather than mitigation.

1 Introduction

A robust scientific understanding of ecosystem processes will help us to understand what ecosystem states are possible and which ones are not. To underpin flexible management, our science also needs to be updated periodically.

Even with well studied ecosystems, there will inevitably be knowledge gaps; scientific understanding is always incomplete and changing. Scientific understanding also inevitably generates new questions, so a list of scientific knowledge gaps would be very long, and would reflect the detailed understanding of individual scientists or scientific groups. It would need to be matched to management goals, and each item would have to be assessed against management-related criteria, such as the extent to which it would improve management, the speed with which a piece of knowledge would result in improvement, and whether the piece of scientific knowledge would provide a direct benefit or require additional steps.

This science review takes all of these things into account. We begin with an assessment of the important components of Western Port's marine ecosystem – its assets[2] and the threats posed to those assets. For each of the assets (or groups of assets), we critically assess the scientific knowledge for that asset and describe the major threats. This assessment is the basis for identifying the extent to which gaps in scientific understanding constrain mitigation. This then forms the basis for defining a set of research needs.

The study was commissioned by Melbourne Water and the Victorian Department of Sustainability and Environment, with partial funding from the Port Phillip and Western Port CMA and the Victorian Investment Framework.

Project aims

In the brief for this study, the overall project aims were outlined, as follows.

- Consolidate existing knowledge on key values and threats identified by relevant agencies and other stakeholders.
- Consider extent of alignment between existing knowledge of values and threats and current/planned management activities.
- Identify potential short term scientific research projects actions arising from the review.
- Identify critical knowledge gaps relative to agency strategic information needs.
- Scope a targeted environmental research program including methodology, costs and delivery options.

This review is broadly asking how Western Port has changed, what the current threats are to this ecosystem, and what kind of scientific knowledge we need to be able to effectively manage these threats into the future:

1. What are the knowledge gaps?
2. Which knowledge gaps are critical to underpin management decisions and agency prioritization?
3. What research will fill these critical knowledge gaps?

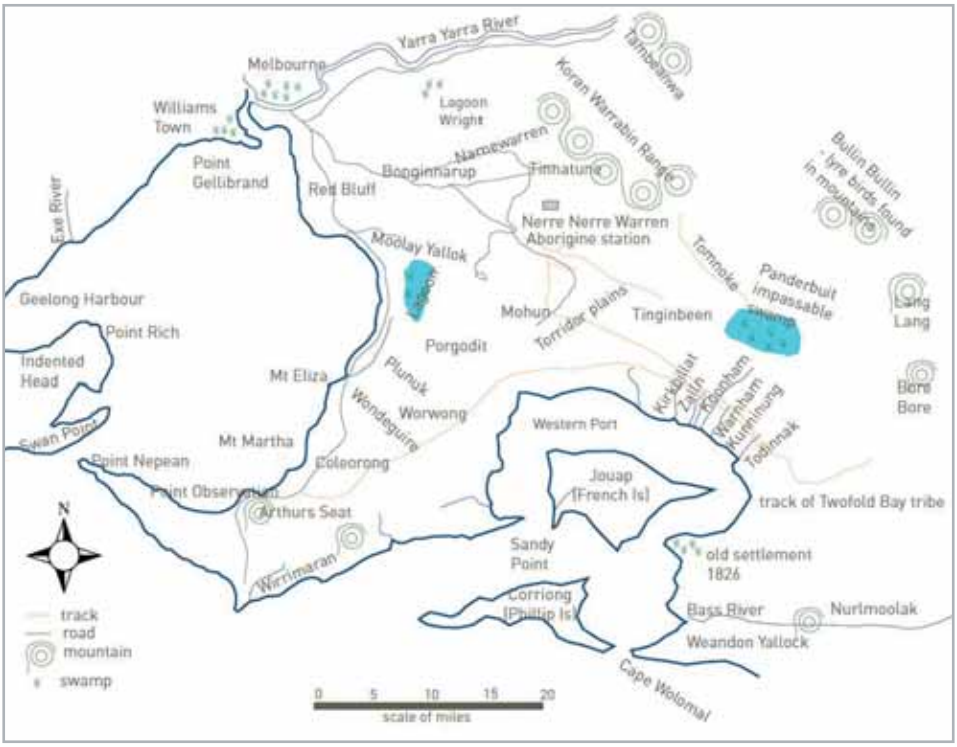


Figure 1.1. The Western Port District in 1840, from an original map by William Thomas, Assistant Protector of Aborigines (DPCD 2011).

2 The term "assets" has a variety of meanings, and the way it is used in this report, to indicate important or valued ecosystem components, is different from how it is used in NRM planning in Victoria. For NRM planning, a map of statewide marine assets has been developed by DSE (www.dse.vic.gov.au). The study team for this review contributed to the Western Port component of this map.

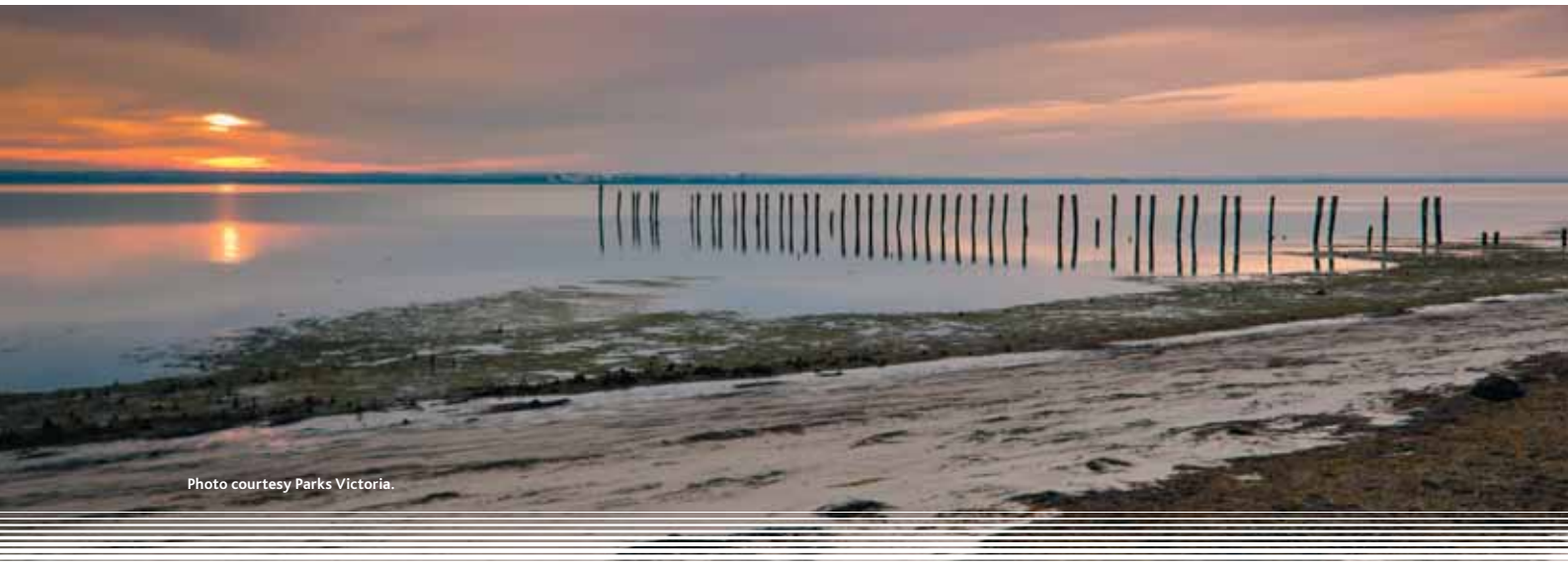


Photo courtesy Parks Victoria.

The scope was marine waters, up to the high tide mark, including mangroves and saltmarsh. Although consideration is given to the threats that are likely to be most significant to the health of Western Port, this review is not a formal 'risk assessment'. In regards to threats, this document focuses on the scientific evidence linking these environmental assets and threats and the scientific knowledge that is needed to inform potential management actions.

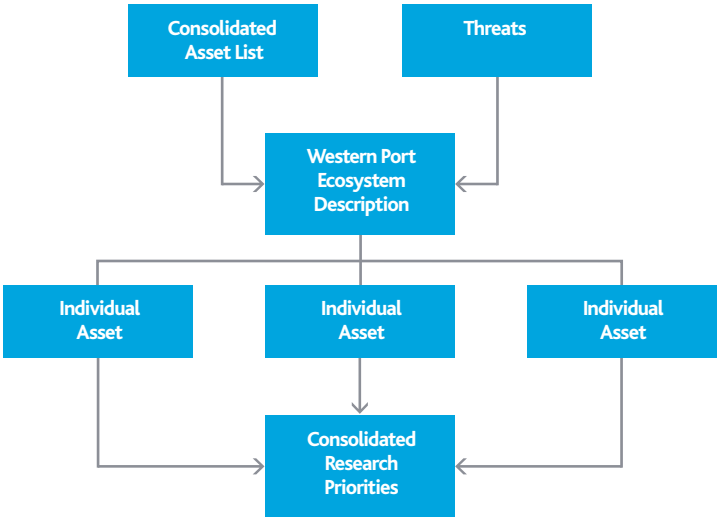
This review also takes an ecosystem focus to Western Port, looking at the whole bay, the connections between different parts of the ecosystem, and the significant and long-term threats to this ecosystem. This comes at the expense of small-scale management issues in particular locations, jetties, channels, coastal infrastructure, etc. These activities and their management are an integral part of maintaining a healthy ecosystem, but where the activity has a very localized effect, it has not been pursued in detail here.

Structure of this document

We begin with an outline of the definitions of assets and threats used in this document (Chapters 2, 3; Figure 1.2). The assets and threats reflect the collective view of the authors, incorporating stakeholder input, and serve as a broad basis for the detailed examination of Western Port in Chapters 4–15. We begin with a description of Western Port as an ecological system (Chapters 4–6). This requires a critical assessment of how well we understand the physical basis of this ecosystem (Chapter 6). We then consider the major groups of assets, from major habitats (e.g. seagrass beds) to important organisms (e.g. shorebirds) to the ecological processes that underpin healthy ecosystems.

Each chapters generates a set of research priorities that reflect the expertise and priorities of the authors. These research priorities were in many cases developed in consultation with expert colleagues. They were then reviewed as a group to identify similar priorities arising from different chapters (which might simply reflect duplication, or indicate that there are similar research needs across different components of the Western Port ecosystem), to align recommendations against current and future management needs, and to rank each research need accordingly. The result was a consolidated and prioritised list of research needs that reflects the combined view of the authors (Chapter 15).

Figure 1.2. Schematic of approach used for this review.



An estimation of the cost, expertise and materials required to deliver the highest-priority research item are also provided, and an assessment is made of the risk of failing to deliver relevant management information. We have also considered whether people and research groups with relevant expertise and the capacity to deliver these research tasks are available in Victoria.

1 Introduction

Climate change

This review identifies research priorities to inform management actions in Western Port in the coming years, so considerable attention will be paid to identifying and scoping research programs that might begin in 2011 or soon after. We also address more strategic knowledge needs for which climate change presents special problems. An ongoing problem with communicating the risks of climate change and defining mitigation and adaptation responses is the time-scale involved. Climate change is a global phenomenon that has been accelerated by the release CO_2 into the atmosphere from human activities. A rising atmopsheric CO_2 level triggers a cascade of effects because it changes atmospheric processes, leading to rising temperatures that themselves are expected to trigger a range of effects. CO_2 also moves into the world's oceans, causing an increase.

There is massive inertia in the global atmosphere and oceans, in large part because of the volume of the world's oceans. There is a large time lag between changes to atmospheric CO_2 concentrations and changes to ocean temperatures, sea levels, and acidity. Current CO_2 emissions will result in changes for decades or centuries, and, correspondingly, avoiding dangerous climate change in the latter part of this century will depend on actions taken to reduce emissions in the next few years.

We do not expect changes to occur at uniform rates. For example, current projections of sea level rise under a 'business as usual' emissions scenario have sea levels rising by close to 1 m by 2100.[3] Such a rise would occur slowly, and current sea levels are rising at approximately 3 mm/y in south-eastern Australia (Church *et al*. 2010). Changes to coastlines will vary on smaller scales, and can be influenced by local geological movements, such as subsidence.

The annual rate of increase will be much higher in the last parts of this century and beyond. The effects of rising sea levels (and other climate changes) will become increasingly important, but the initial effects on our coastal activities will be small.

This creates three relevant time-scales at which management action is appropriate:

- To avoid 'dangerous' climate change (IPCC 2007), we must reduce global emissions in the next few years. This action will require widespread change, with limited local input.
- Adapting to or mitigating climate change will become steadily more important for local managers, with the most significant responses required later this century.
- Considerable scientific uncertainty exists about the exact nature of physical changes, and how local ecosystems will be modified. Understanding these changes is essential if we are to develop effective mitigation and adaptation plans, so we know how we will be affected and can predict the success of our interventions. Gaining this knowledge is a precursor to effective local management. Developing a detailed understanding is a strategic research need over the next one or two decades, commencing now.

In this review we highlight some of the strategic research needs relevant to climate change, but we do so in the knowledge that although this information is critical to our future management it does not have immediate implications for how we manage Western Port.

3 There is some scientific uncertainty about this figure; see Church *et al.* (2010). The Victorian Coastal Strategy (Victorian Coastal Council 2008) used a value of 0.80 m as a guideline, with a recommendation that the value be reviewed frequently.

The area covered

The western boundary of Western Port extends in a straight line from West Head at Flinders to Point Grant on Phillip Island, and the eastern boundary extends from a point south-east of San Remo (just east of Griffith Point) across to Newhaven on the eastern shore of Phillip Island (Figure 1.3).

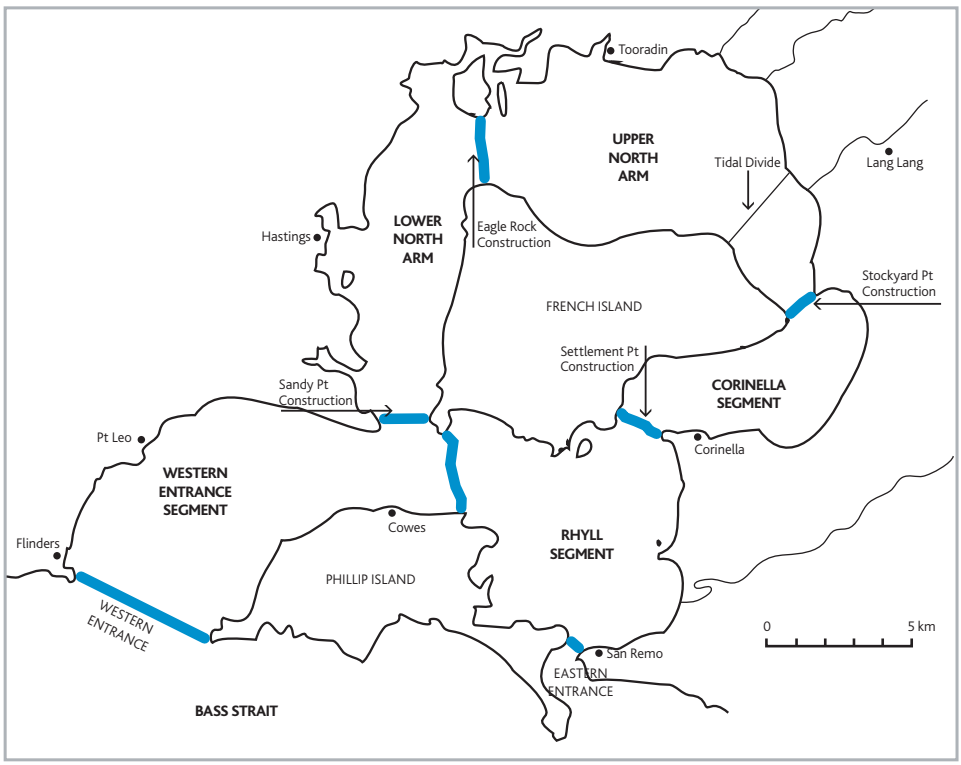


Figure 1.3. The extent of Western Port, showing its western and eastern demarcations.
(After Marsden *et al.* 1979.)

2 Overview of assets

Michael Keough and Rachael Bathgate

Photo courtesy Michael Keough.

In order to manage marine environments effectively, we need to know what aspects of marine environments are important (and to whom), and recognise the activities that might pose threats; and from that position, examine ways of managing major threats to important environmental components. Important aspects of ecosystems are commonly called 'values' or 'assets', and we adopt the term 'assets' here. Assets are components of the ecosystem. They may have a very narrow focus, such as an individual species and a specific breeding area, or they may be broad and inconspicuous, such as habitat areas that support nutrient cycling or other ecosystem services.

The Victorian Department of Sustainability and Environment (DSE) uses an assets-based approach to identify ecological features of state and bioregional significance and to prioritise natural resource management activities (DSE 2009). This approach has an inherent spatial component because it involves the ranking of different examples of a particular class of assets. For example, there are many examples of *Zostera* seagrass beds, but some may be in better condition, more resilient, or support a greater diversity of other species, and thus be weighted more heavily. Similarly, most intertidal mudflats are used by shorebirds, but some feeding areas are more important than others. Significant marine environmental assets are broadly defined as those that:

1. are of state or bioregional importance for:
   - biodiversity or endemism, or
   - their ecological role or function.
2. support and make a major contribution to the 'fitness' of a species or subset of species that:
   - are themselves of international, national, state or bioregional importance for biodiversity, or
   - play a key ecological role or function (i.e. at a state or bioregional level).

DSE uses other factors in determining significant environmental assets, including habitat representation, naturalness and resilience.

Stakeholder list of assets

Following consultation with a wide range of Western Port stakeholders (principally through the Western Port Catchment Committee), DSE and Melbourne Water provided the Strategic Knowledge Review team with a consolidated list of assets. In the first workshop these assets were organised into larger groups, based on the nature of the assets or their ecological significance.

Table 2.1. Draft environmental assets list provided by DSE and Melbourne Water.

Asset	Example
Habitats	
Iconic	marine parks and protected areas
	FFG listed communities (such as San Remo)
	sites of geological and geomorphologic significance (including 'ancient' watercourse features)
Intertidal reefs	exposed rock platforms
	sheltered rock platforms
Sediment	sandy beaches
	seagrass meadows
	intertidal mud sediments
	mangroves
Subtidal/offshore reefs	exposed subtidal reefs
	sheltered subtidal reefs
	deep reefs
Water column	pelagic zone
	demersal zone
Coast	saltmarsh
Bird roosting/breeding	
Biota	
Iconic	penguins
	dolphins
	fur-seals
	whales
Birds	migratory waders
	conservation-listed species (e.g. Orange-bellied Parrot)
	other significant species (e.g. White-bellied Sea-eagle)
Fish	conservation listed species (e.g. Elephant Fish, sharks, Pale Mangrove Goby)
	migratory fish species (marine/estuarine/freshwater)
	other significant species (e.g. King George Whiting)
Invertebrates	'living fossil' invertebrates
	invertebrates important for key ecological marine and coastal processes
Ecosystem services and processes	nutrient cycling
	carbon credits

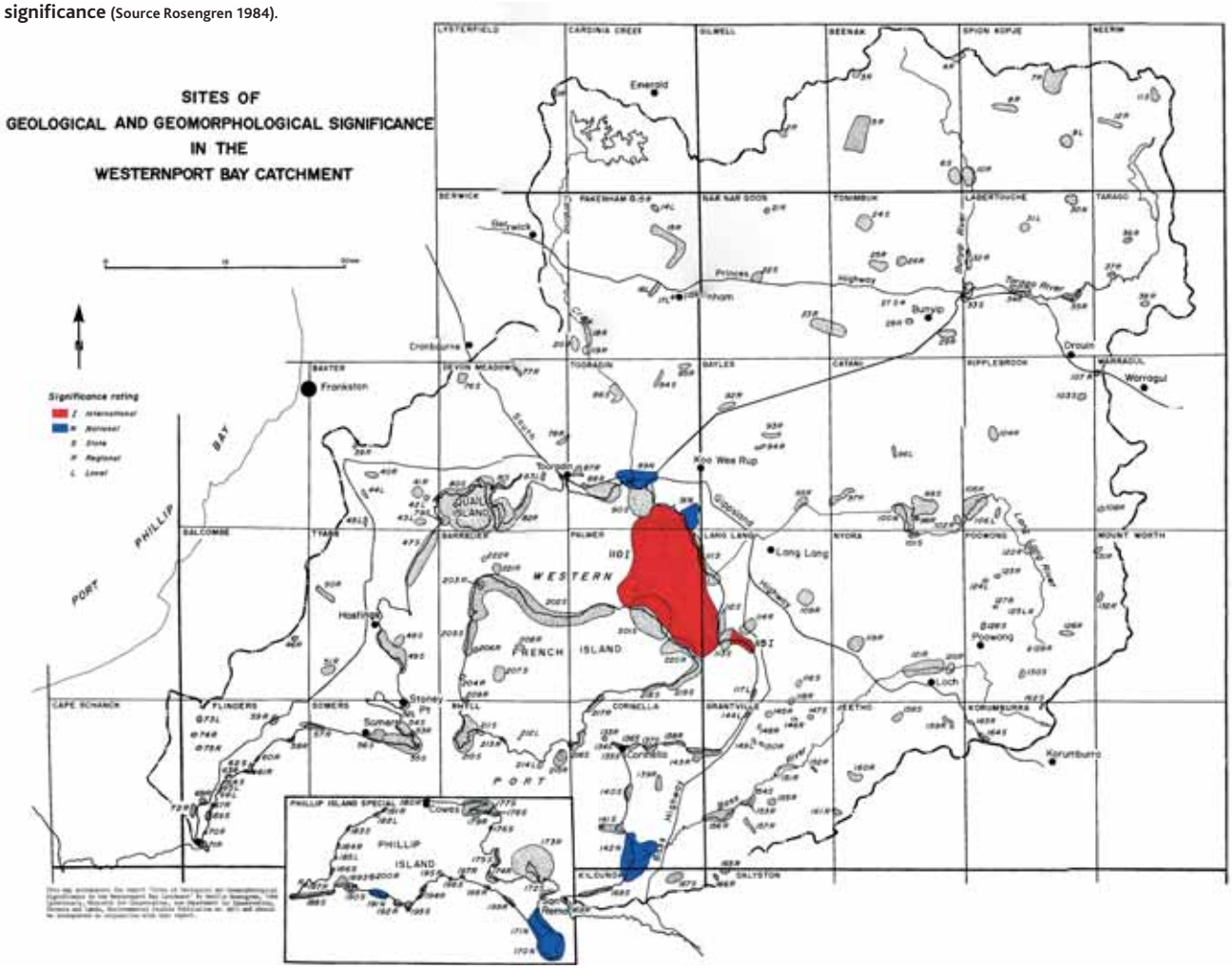
A habitat-based approach

The approach used in this study is to recognise that assets can be grouped into large categories that are often based on the physical nature of the habitat, such as rocky reefs or soft sediments.

The physical attributes of Western Port, particularly its geology and geomorphology (reviewed comprehensively by Marsden and Mallet 1975, Bird 1993, Rosengren 1984), have a profound effect on habitat distributions. Along with water movement, meteorological and climatic factors provide the underlying structural components of Western Port. Knowledge of local geology, and more specifically, lithology, is important to understanding processes such as movement of sediments and the dynamics of intertidal flats, and can influence species distributions in soft sediments and in some cases on rocky reefs. Sites of geological and geomorphological significance in Western Port have been described by Rosengren (1984) and are available from DPI (2011a). Of particular note is the Western Port Tidal Watershed, located in northeastern section of the Bay, between Palmer Point (French Island) and the Lang Lang River mouth. This tidal divide is of international significance and the dynamics of the area play a critical role in determining the nature of tidal flow in other parts of Western Port. Management actions for conservation of this area include controls on dredging and spoil disposal, and improved navigation aids to prevent boat propeller damage (DPI 2011b). The Quarternary cliffs at Pioneer Bay are also internationally recognised for their value in understanding the nature of sea level change on different continents (DPI 2011c). Other listed sites are of national significance, with numerous locations around the Bay being identified as significant at state, regional and local levels.

The physical nature of the habitat constrains the kinds of organisms present, and hence the dominant ecological processes. The physical characteristics of the environment also influences the way in which humans use or interact with particular habitats. Within broad habitat categories there are several subcategories based on the presence or absence of habitat-forming species, or 'ecosystem engineers'. The influence of these ecosystem engineers is evident, for example, in the contrast between unvegetated subtidal soft sediments and areas supporting seagrasses. Likewise, along the coastal fringe the unvegetated beaches or mudflats are again very different from those with mangroves or saltmarshes. We also recognised that intertidal habitats are very different from those below the low-tide level; for example, intertidal plants and animals subject to increased stress through desiccation, extremes in temperature, higher levels of ultraviolet radiation, and more variable salinities as a result of rainfall. Other listed sites are of national significance, with numerous locations around the bay being identified as significant at state, regional and local levels (Figure 2.1).

Figure 2.1 Western Port sites of geological and geomorphological significance (Source Rosengren 1984).



These considerations resulted in the identification of eight main habitat groups, including the water column (pelagic environment), as summarised in Table 2.2. As part of that classification, we also recognised the distinct ecology of different vegetation classes in the coastal fringe (e.g. mangroves and saltmarsh).

Table 2.2. Asset categories used in this review.

Asset	Subcategory	Example of asset or special cases of asset
Habitat groups		
Rocky reefs	Intertidal	San Remo opisthobranchs
	Subtidal	Crawfish Rock
Vegetated sediments	Mangroves	Marine national parks
	Saltmarshes	
	Seagrass	
Unvegetated Soft Sediments	Intertidal	
	Subtidal	
Water column		
Significant fauna		
Iconic species	Shorebirds	Breeding sites
	Seabirds	
	Marine mammals	
Commercial/Recreational	Fish	Chondrichthyan breeding sites
Emergent features		
	Whole of bay properties	Unusual example of mosaic of different habitats connectivity
	Ecosystem Services	Nitrogen cycling
	Collections of assets	Biodiversity hotspots

These main habitat types are heterogenous around Western Port, with some very extensive (e.g. unvegetated intertidal soft sediments) and others small and scattered (e.g. subtidal rocky reefs). The most recent known distributions of these habitat types were used as the basis for individual chapters, and also formed part of a formal asset identification for DSE.

The most extensive habitats are unvegetated soft sediments, including very extensive intertidal mudflats and areas of fringing mangroves and saltmarsh (Figure 2.2). Some of the intertidal flats are covered by seagrasses, but seagrass areas are currently more extensive below the low tide level (Figure 2.3). Rocky reefs are sparse in Western Port; they are confined largely to the south-western section (Figure 2.4), although there are small reef areas elsewhere that may be important for biodiversity. The distribution of these habitats is dealt with in detail in later chapters.

Habitat groups do not encompass the entire range of assets. For example, there may be individual species or populations within Western Port that have particular conservation significance (such as representatives of endangered species) or are ecologically important (such as species whose presence affects other species). There are also other components of Western Port that may require special attention for other reasons, e.g. fish of recreational or commercial significance or 'charismatic' species of marine mammals. 'Value' can also be conferred on species or groups of species because of national or international agreements that apply to them, and there are often regional obligations and actions associated with these species. Our second major category of assets is therefore significant fauna.

Significant fauna include iconic species, such as some of the migratory shorebirds that are common on Western Port's tidal flats. These species are the focus of existing international agreements such as the Ramsar Convention. The habitats that support these birds are considered to be assets for Western Port, and includes areas used for feeding, roosting, and nesting (Figure 2.5). We also considered fish separately because some species are valued as part of recreational fisheries in Western Port (e.g. King George Whiting, flathead, Elephant Fish), or in commercial fisheries elsewhere (e.g. Elephant Fish). Other fish highlighted in this review are those that are charismatic (e.g. seadragons) or protected by law (e.g. under the Victorian *Fisheries Act 1995*).

The recognition of individual components of the Western Port ecosystem is important, but when considered in an integrated manner these individual components reveal a further tier of assets. These 'complex' assets, which we have called emergent features, are not necessarily defined spatially in the way individual components are, but rather link or support these individual components. They may support other Western Port assets, or they may form features that are significant at a state or national scale.

Emergent features include ecosystem services such as the removal of nitrogen, which is a process that can occur across several habitats and depends on the exchange of material among them (e.g., between unvegetated soft sediments and the water column). The movement of organisms between habitats is another example. The open nature of marine environments means that habitat components are not isolated; currents transport nutrients, toxicants and plankton (including larvae of marine animals and spores or seeds of plants and algae), and these currents cross habitat boundaries. Larger organisms such as fish can also move freely between habitats. These larger animals may make occasional use of certain habitats, or occupy habitats during a specific life-history phase, or have regular and periodic patterns of use (e.g. seasonal or tidal patterns). In particular, many marine animals spend the early part of their lives in habitats quite different from those in which adults live.

2 Overview of assets

Western Port (and its companion, Port Phillip Bay) is also an unusual geological feature, formed when the lower reaches of a river system were inundated by a rising sea level during the Holocene period. This 'Western Port sunkland' now forms an extensive tidal bay. The bay as a whole, and features within and around it, are of considerable geological and geomorphological significance (Rosengren 1984). These large embayments are also unusual in the variety of habitats that are present, from relatively exposed rocky reefs to mangroves and sheltered mudflats. Each of these habitats has a unique mix of fauna and flora, and some sites in Western Port support a high species diversity. This mosaic of habitats may be considered an emergent feature.

For example, areas regarded as important for Elephant Fish breeding in south-eastern Western Port are close to small reefs noted for their high diversity of opisthobranch molluscs. Individually, each of these items might not warrant special value or consideration, but combined they may be of significant value.

Western Port also falls under several other classification systems or agreements, which reflect various perceptions of the bay's values:

- Western Port was listed as a Ramsar site in 1982, reflecting its status as a Wetland of International Importance for migratory and resident shorebirds (DSE 2003).

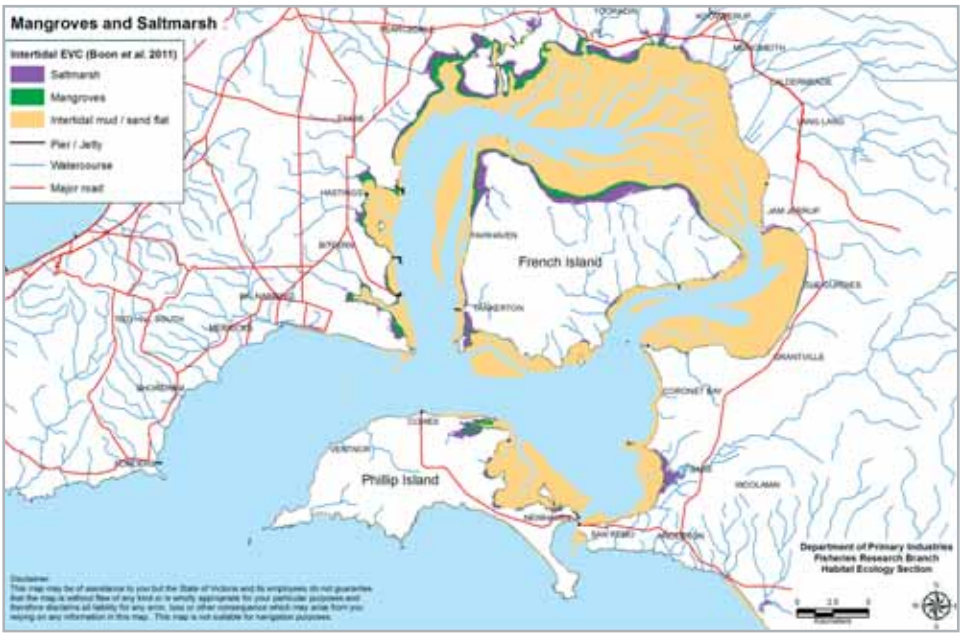


Figure 2.2. Distribution of intertidal soft sediment habitats in Western Port, showing unvegetated tidal flats and fringing mangroves and saltmarsh.

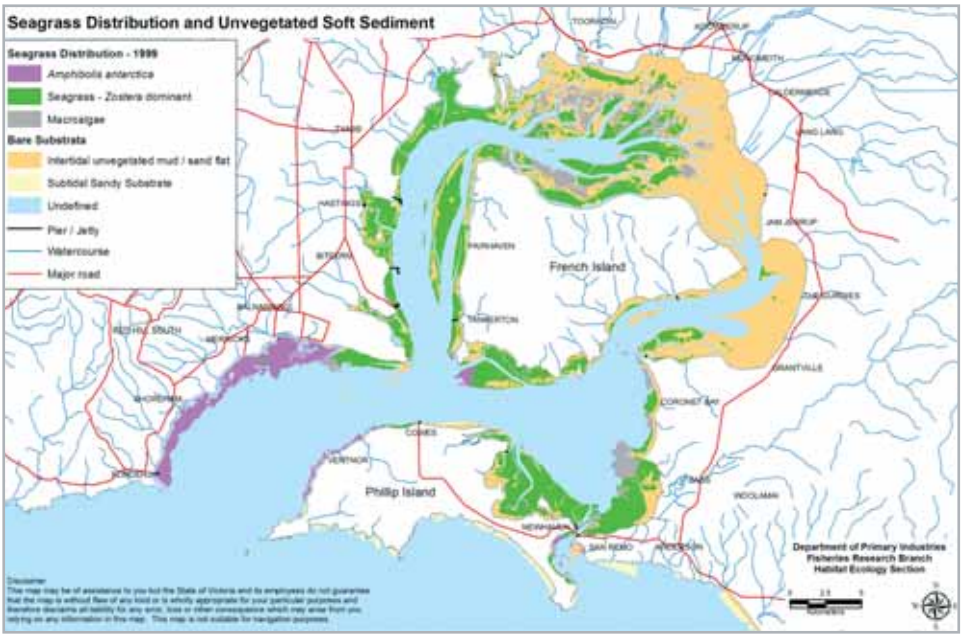


Figure 2.3. Distribution of seagrasses, intertidal and subtidal bare sediments in Western Port in 1999. Note that much of the area shown as 'undefined' is thought to be subtidal bare sediments.

- Western Port is also within the UNESCO Mornington Peninsula and Western Port Biosphere Reserve that was designated in 2002, although this reserve designation does not affect the legal status of the land and imposes no binding international legal requirements regarding its management (DSE 2003).
- Marine National Parks (MNPs) were established in Western Port in November 2002 as part of a state-wide representative system of marine reserves. Churchill Island MNP on the north-eastern shore of Phillip Island, and French Island and Yaringa MNPs in the northern section of the Bay, contain a wide range of habitat types representative of Western Port.
- A number of species, as well as the San Remo marine community, are listed under the Victorian *Flora and Fauna Guarantee Act 1998* (FFG Act).

Various other conservation strategies and management actions apply to Western Port, and these are listed in recent management plans (Parks Victoria 2003, 2007a). Where relevant, more detail on these sites and species within Western Port is provided in individual assets chapters.

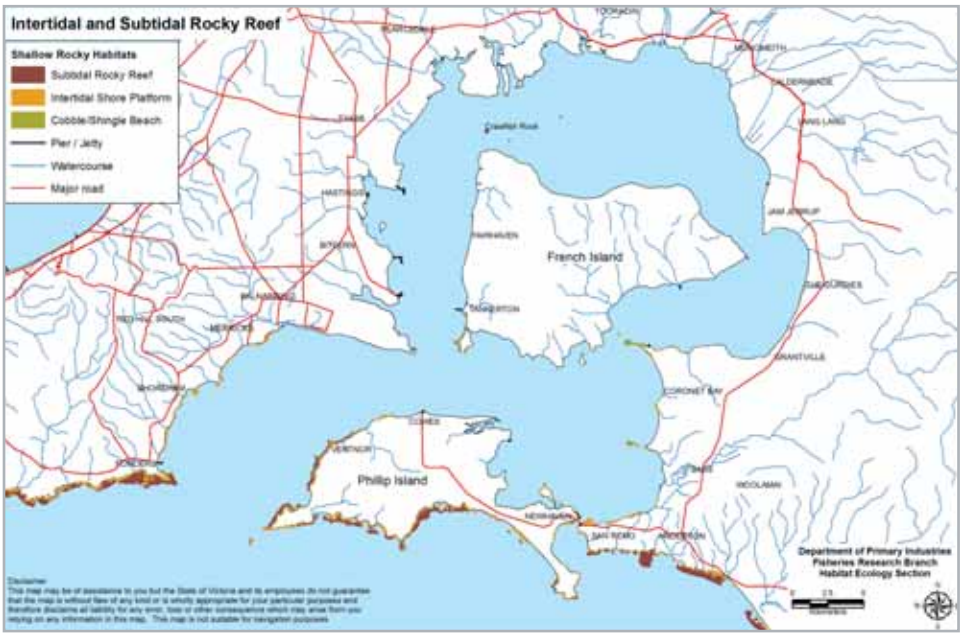


Figure 2.4. Rocky reef areas in Western Port. Note that most of the subtidal reefs shown are outside Western Port, on the ocean side of the coastline.

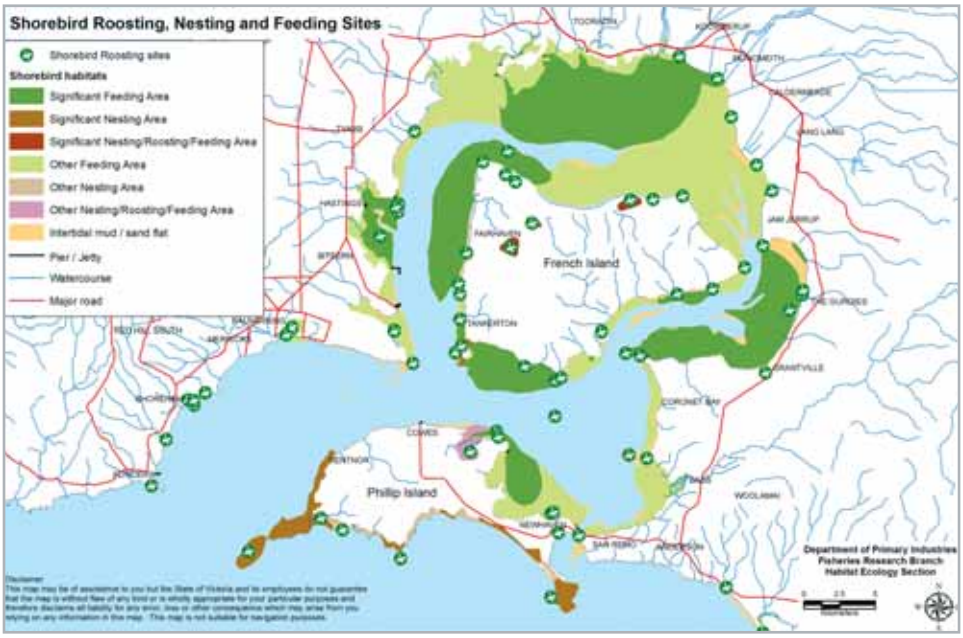


Figure 2.5. Areas of Western Port used by shorebirds.

3 Threats and exposure pathways

Michael Keough and Rachael Bathgate

The many different activities around Western Port place pressures on its ecosystem, and these pressures are increasing as Melbourne's urban growth expands south-eastwards, agricultural and industrial activities intensify, and climate continues to change. Some impacts can be reduced by actions in or around Western Port, while others require actions farther afield or are beyond the scope of local action. Some pressures are more serious threats than others to Western Port's assets. To identify the important threats, we need to understand the concerns of stakeholders, examine the scientific evidence about the severity of the threats, and understand the link between a particular activity around Western Port and a threat to an ecosystem component.

The purpose if this chapter is to describe the understanding of the relationship between important components of the Western Port ecosystem described in the previous chapter and threats. Threats were assigned to one of three categories or designated as too poorly known to categorise. The rationale behind these classifications is expanded in the following chapters, but poor water quality emerges consistently as a serious threat across most ecosystem components. Suspended sediments are the most important aspect of water quality, although nutrients may be an issue in some parts of Western Port. Toxicants, particularly those associated with the eastern catchments, are also a potential concern, although there is generally insufficient information to be able to prioritize their relative importance.

For individual components of Western Port there are specific threats. For example, marine invertebrates are collected from mudflats for bait and from reefs for food, and fish are taken from several areas. Our assessment is that recreational fishing is an important threat requiring attention, while bait and food collection is a less serious issue on reefs and mudflats and is not an issue in seagrasses, saltmarshes or mangroves.

Climate change is a consistent theme, with rising sea levels a serious threat for saltmarsh, mangroves, intertidal rocky reefs, shorebirds and waterbirds. Acidification is also likely to be important in future, but there is some uncertainty about its severity, and no practical actions in Western Port could ameliorate this threat when it becomes more important later this century.

In addition to recognising the important parts of Western Port's ecosystem, we also need to understand the pressures operating on this ecosystem, particularly those that are derived, directly or indirectly, from human activities, and to think about how different Western Port components respond to those pressures. It is easy to identify a large number of potential threats, but the challenge is to identify those most likely to result in undesirable change and, within that group, to highlight threats that can be mitigated by management actions. Effective mitigation will also require an understanding of the causal pathways, linking a particular action to a change in Western Port, so we can identify points at which intervention might be most effective.

This chapter provides an overview of the major threats for Western Port, reflecting stakeholder and expert experience and knowledge. We identified the major threat pathways and summarised the major threats for the each of the major habitat types in Chapter 2. Threats have been classified broadly as high, medium, low, or unknown if there are grounds for concern but insufficient information to classify the level of threat. The details are expanded in subsequent chapters.

In examining the threats, we have considered the risk to different components of the Western Port ecosystem. Risk is used in its general sense, rather than a more formal, semi-quantitative Risk Assessment (e.g. compliant with ISO 31000:2009). Our intention here was to screen out threats that were of low priority.

Water quality is a concern across many habitats, and an elevated level of suspended sediments is a major threat. Nutrients and toxicants are a possible concern, particularly for seagrasses, mangroves, saltmarshes, and unvegetated soft sediments, but we lack detailed information to assess the severity of this risk. Other aspects of water quality, such as temperature and acidity, are also important, but there is limited capacity to mitigate the risks at present.

Some threats are asset-specific. For example, invasive species are probably more widespread than expected within saltmarsh, less of an issue in mangroves and seagrass habitat, and potentially important but not well documented in unvegetated soft sediments. Sea-level rise is likely to be very important for intertidal reefs and shorebirds but relatively unimportant for fish.

Approach

Stakeholder perceptions

Melbourne Water and DSE consulted widely with Western Port stakeholders to produce a consolidated list of perceived threats (Table 3.1). These threats were worked through by the study team.

Grouping of threats

The list of perceived threats was reorganised and augmented during the workshop, linking threats that cause changes to water quality, sediment quality, habitat, etc. After this reorganisation there remained some threats that did not easily fit into large groupings, and were treated separately. This final threat list (Table 3.2) was the basis for individual chapters.

Exposure pathways

A direct threat to a Western Port asset is a concern, but managing that threat requires an understanding of the proximate and ultimate causes behind this threat. For each threat we have illustrated the pathways by which a particular asset may be threatened, illustrating points at which some intervention may be possible, but identifying the precise source of threats is beyond this review.

3 Threats and exposure pathways

Table 3.1 Perceived threats identified in stakeholder meetings.

Threat	Examples
Diffuse and licensed point-source pollution (surface water, groundwater and atmospheric)	• nutrients • sediment/turbidity (including catchment and waterway erosion) • heavy metals • organics (pesticides, herbicides, petroleum hydrocarbons) • freshwater • saltwater • acid sulfate soils • specific land use practices (including agricultural aerial spraying) • fuel reduction burns (not ecological burns)
Marine and coastal pollutant generation and re-suspension	• coastal erosion • dredging • oil spills
Land use change	• urban growth • agricultural change (including moves to intensive agriculture in areas such as Mornington Peninsula and Koo Wee Rup, and recycled water schemes) • industrial development • coastal development (including Port of Hastings expansion, Stony Point ferry)
Climate change	• sea-level rise • flooding • freshwater and saltwater inputs • increased air and water temperatures
Fragmentation of marine and coastal processes and habitats	
Historical and future coastal modifications	• draining of Koo Wee Rup swamp and potential for recovery • flood/high flow velocity mitigation structures • sea walls, rock groynes (including coastal fortification with urban growth) • existing and building of new levee banks
Marine and coastal pest plants and animals	
Current and future recreational, commercial and industrial activities	• increased recreational pressure (disturbance, structures) anticipated given extent of planned urban growth • increased car park areas needed beside launching ramps (taken from public land) • need to understand cumulative impact as well as individual impact of increased activity
Coastal and catchment vegetation clearing/grazing	
Surface water and groundwater extraction	
Management constraints	• ecosystems are spatially dynamic (rivers, coastline, vegetation zones move) but management boundaries are often fixed • land tenure

Table 3.2 Consolidated threat list used for this study.

System Category		Sources	Pathways	Driver of Change
Water and sediment quality	**Nutrients**	agriculture	creeks, rivers, drains, discharge pipes	population growth, land use change
		aquaculture	direct run-off	
		burning	atmospheric deposition	
		effluent discharges from WWTPs*	discharge pipes, septic tanks	
	Sediment Input	agriculture	coastal erosion and run-off	population growth, land use change
		draining of Koo Wee Rup swamp		
		coastal development		
		port and marina development		
		unsealed roads		
	Sediment resuspension	port and marina development	in-situ resuspension	population growth, climate change
		new structures		
		anchors, engines, propellers		
		dredging		
		freshwater/saltwater inputs		
		altered wind-wave patterns		
		increased storm events		
		loss of seagrass		
	Heavy Metals	Port of Hastings (existing activities)	sediment resuspension	population growth
		industrial development	in-situ leaching	
		effluent discharges from WWTPs	discharge pipes, septic tanks	
	Non-Metal Toxicants			population growth, land use change
	TBT and anti-fouling substances	port development, marinas and jetties	sediment resuspension in-situ leaching	
	Pesticides and herbicides	agriculture	creeks, rivers, drains, direct runoff	
	Pharmaceuticals and estrogens	agriculture (animal husbandry) effluent discharges from WWTPs	creeks, rivers, drains, direct runoff discharge pipes, septic tanks	
	industrial chemicals	industrial and coastal development	creeks, rivers, drains, direct runoff	
	Petroleum Hydrocarbons	oil spills motorised watercraft	Port of Hastings creeks, rivers, drains, discharge pipes in-situ discharge	population growth
	Pathogens	aquaculture	in-situ (including natural and human-mediated vectors e.g. boating)	
	Salinity	agriculture	marine and freshwater inputs	population growth, climate change
		urban growth		
	Acidity	acid sulphate soils	creeks, rivers, drains, leaching, runoff	population growth, climate change
		ocean acidity	climate change	
Hydrodynamic and atmospheric variables	**Sea-level Rise**	flooding		climate change
	Alteration of Coastal Processes	altered wind-wave patterns		
	Increased UV-B			
	Increased Temperature			
Pest plants and animals		port activities		population growth
		shipping and boating		
		aquaculture		
		agriculture		
		urban growth		
Habitat and species	**Haitat Loss**	coastal development		population growth
	Fragmentation	industrial development		
	Habitat Quality	urban growth		
	Debis	new structures		
	Disturbance	human and automated traffic		
	Extraction	changes to water movement		
Cumulative impacts	N/A	N/A	N/A	N/A

* Waste Water Treatment Plant effluent may contain particulate organic matter, nitrogen, phosphorus, carbon, human pathogens, metals, and endocrine-disrupting chemicals (EDCs).

We do this for individual threats, and include a brief description of the nature of the threat, the consequences of this particular threat, and a consideration of the pathway by which the threat is realised. Where possible, we also indicate whether the particular threat is likely to increase or decrease in future, based on changing population and land use around Western Port and also through climate change.

We also summarise, where possible, knowledge of how particular threats function around Western Port, and expand on this knowledge in the following chapters.

Changes to water and sediment quality

Nutrients

Excess nutrients are a problem in many coastal areas. Elevated levels of nutrients are of particular concern along urbanised coastlines, in areas where there is intense agricultural activity, and where there are large estuarine discharges or sewage release points. Increased availability of nitrogen and phosphorus from anthropogenic sources can stimulate the growth of phytoplankton and macroalgae, potentially resulting in algal blooms.

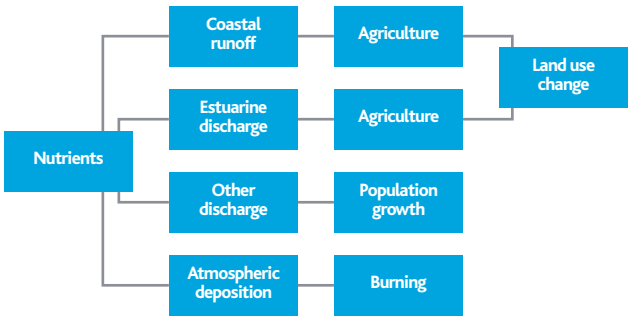
High nutrient levels coupled with other physical factors (e.g. increased temperature and light, reduced current flow) can cause excessive phytoplankton growth, which can have negative effects on ecosystem condition. Phytoplankton blooms can increase turbidity as well as the magnitude of diurnal oxygen changes. In some cases, toxic cyanobacteria (e.g. *Anaebena*, *Nodularia* and *Microcystis*) proliferate in these blooms which then become harmful to aquatic species, wildlife and humans (Arundel *et al.* 2009). While nitrogen is most often the nutrient of concern in marine waters, phosphorus can be a problem in some situations. Carbon and silicates may also be important in the Western Port system, but there is little information about these nutrients (Counihan and Molloy 2003). Nutrients enter the bay via waterways draining urban and agricultural areas around Western Port, unsewered townships, industrial effluent, erosion of coastal sediments and the atmosphere.

Western Port is generally considered to have low nutrient inputs relative to other bays such as Port Phillip Bay; there is no direct sewage discharge, and the catchment inputs are comparatively small. The most significant catchment inputs are in the Upper North Arm (e.g., Watsons Creek) and Corinella segments. The Bass River is also an important source but it discharges into well-flushed parts of Western Port. Groundwater intrusion, with its associated nutrients, has not been quantified for Western Port, but may be a source or sink of freshwater and dissolved nutrients (Counihan *et al.* 2003). Nutrients and toxicants may arrive from Boags Rocks outfall via the Western Entrance (see Chapter 4), but this has yet to be quantified.

Little is known of nutrient cycling in Western Port, and with only a limited study of nutrient fluxes between marine sediments, water and biota.

Pathways

Figure 3.1 Exposure pathways that may result in changes to nutrient levels



Likelihood

EPA water quality data for nutrients and other physico-chemical indicators at three fixed sites suggest that concentrations remained constant from 1984 to 1996 and were largely unchanged from those reported by Shapiro in 1975 (EPA 1996). Corinella routinely fails to meet environmental quality objectives, having consistently higher nutrients, turbidity, chlorophyll-a (phytoplankton biomass estimates) and metals than Hastings and Barrallier Island (EPA 1996, 2011).

Nutrient inputs from catchments are determined by more distant activities, particularly agricultural activities and urban development. Melbourne Water has modelled these inputs with the PortsE2 catchment model (Chapter 4). Watsons Creek is also a hotspot of concern (discussed in other chapters). Careful management of urban expansion and agricultural intensification over the next few decades is very important.

Catchment inputs have been reduced over the past decade or more, with rainfall below long-term averages and relatively low discharges during this period. Water quality conditions in 2009 reflect drought that began in 1998 (EPA 2011). A comparison of nutrient levels in the bay during drought and 'normal' rainfall years suggests that levels of nutrients in the bay are likely to decline with the expected reduction in mean annual rainfall through climate change (Chapter 4). Discharges are also predicted to become more variable in time and magnitude.

Septic tank seepage from Merricks and other locations is being addressed through a transition to reticulated sewerage.

Sediment input

Sediments in the water column in Western Port are the result of new material arriving and the resuspension of existing material. Increased turbidity from sources such as dredging and rivers is thought to have been a factor in the decline of seagrass in the Upper North Arm (EPA 1996). Turbidity is caused by suspended matter such as sediment, debris and phytoplankton, and dissolved organic matter such as humic substances (Arundel et al. 2009). In Western Port, resuspended sediments are thought to be the major cause of turbidity.

Sediment input and resuspension has been comprehensively reviewed and modelled in a series of reports under the Western Port sediment study conducted by CSIRO,

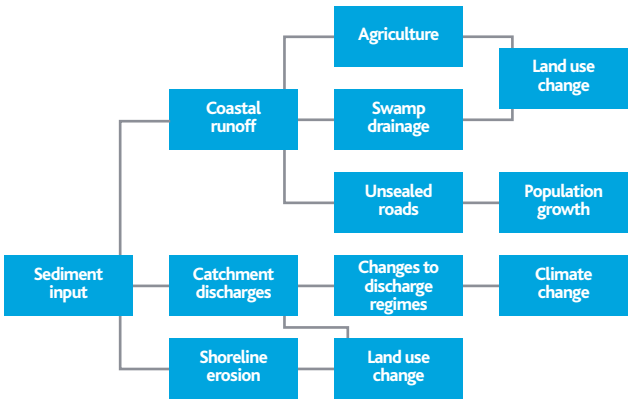
Melbourne Water and the EPA (Wallbrink et al. 2003). Early studies identified the major sources of sediment to Western Port as in-stream bank erosion, catchment erosion and coastal erosion of the Bay shoreline (Sargeant 1977). Large-scale dredging programs in the 1960s and 1970s — including sea and land disposal of dredge spoil — have also contributed significantly to turbidity levels in the past (EPA 1996).

Rates of erosion have increased since European settlement, primarily due to land-clearing which destabilises land surfaces and increases catchment runoff; and through modifications to the natural catchment drainage system, in particular the pathways through the low-lying areas of the catchment, such as the Koo Wee Rup swamp, that were previously mostly disconnected from the bay (Wallbrink and Hancock 2003). Much of the sediment entering the North Arm is transported clockwise around the bay, thus affecting the water quality and seagrass habitat of eastern and southern regions and increasing the extent of mud deposition in the Corinella and Rhyll segments (Wallbrink *et al*. 2003).

Wallbrink and Hancock (2003) provided an overview of the tonnage of sediment input that resulted from the draining of Koo Wee Rup swamp, coastal erosion and dredging, as estimated in various reports (e.g. EPA 1996, Sargeant 1977). These estimates are fairly coarse and need to be revaluated if an accurate sediment budget for Western Port is to be developed.

Pathways

Figure 3.2 Pathways by which sediments are added to the water column



Likelihood

Historical sources of sediments into Western Port are associated with the draining of Koo Wee Rup Swamp, agricultural developments, urban development, and large-scale dredging projects (EPA 1996). These kinds of inputs are unlikely to occur in the future, partly as a result of improved regulatory controls. The Port of Hastings is responsible for maintaining the prescribed depths of the shipping channels and berths requiring occasional dredging (Toll Western Port 2001). Maintenance dredging is also undertaken at smaller boating facilities throughout Western Port (EPA 1996, Parks Victoria 2005).

The dominant catchment source for fine sediment is channel and gully erosion of Lang Lang River and, to a lesser extent, Bunyip River. Erosion from the clay banks north-west of the Lang Lang jetty also appears to be an important local source of fine sediment. Increases in shoreline erosion, particularly the loss of coastal vegetation, would increase inputs.

Sediments from catchment erosion will continue to be a problem for Western Port if further rehabilitation and stabilisation programs are not undertaken (Wallbrink *et al*. 2003). Climate change is expected to increase coastal inundation, reduce overall riverine flows and increase the frequency of storms and flash floods, which together may increase sedimentation in river deltas and estuaries (DCC 2009).

Monitoring by EPA Victoria indicated some decrease in turbidity and increase in clarity between mid-late 1980s and 1996, but an investigation of suspended solids from 1990 to 2009 did not find evidence of a continued improvement in water clarity (EPA 2011). EPA's three fixed sites are also a long way from near-shore assets such as seagrass.

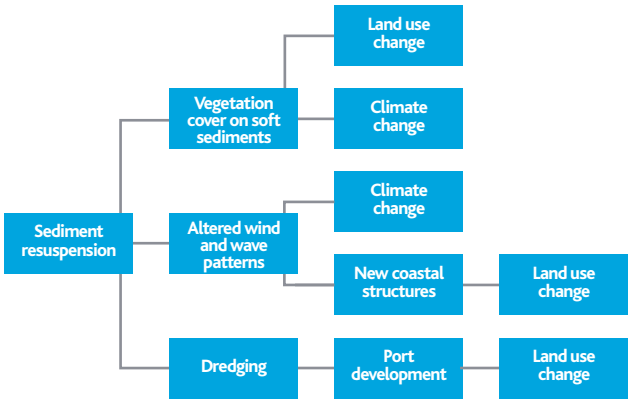
Sediment resuspension

Resuspension is the major source of high sediment loads in Western Port waters, particularly in northern sections (Wallbrink *et al*. 2003). Sediment entrainment and transport processes create high and persistent turbidities within the bay, particularly in the shallow northern and eastern zones. These persistent high turbidities in Western Port arise from the daily reworking and resuspension of fine sediment by tidal, wind and wave action (Wallbrink *et al*. 2003). The average resuspension rate for the bay is estimated to be 6 000 $kt.y^{-1}$ (based upon differences between suspended sediment concentrations and input loads). Suspended particle residence times are short – generally less than a day (Hancock *et al*. 2001). There is a migration of sediment from the north, above French Island, to the east (Corinella) and south (Rhyll) segments, a process that appears to have created significant changes to the composition of North Arm sediments over the last 25 years (Hancock *et al*. 2001).

Resuspended sediments are significant because they change the light climate for algae, seagrasses and benthic microalgae, and when they settle from the water column they can cover the photosynthetic surfaces of algae and clog the feeding structures of animals.

Pathways

Figure 3.3 Mechanisms for altering rates of sediment resuspension.



Likelihood

The rate of sediment movement within the embayment and the rate of material export to the ocean are unknown. Modelling based on 2030 global climate change predictions shows there will be significant increases in suspended material throughout the system, most likely with heightened concentrations in the Eastern Arm (Lee *et al.* 2009).

All resuspension is ultimately determined by wind and wave patterns and interactions between the resulting water movement and the seabed. This interaction is mediated by aquatic vegetation. Turbidity generated by resuspension may have been exacerbated by the recent decline of the seagrass beds, and may continue to retard their recolonisation. Any additional changes to seagrass cover may also feed back to resuspension rates.

EPA monitoring has shown a strong positive correlation between suspended solids and chlorophyll-*a*. It is suggested that high concentrations of suspended solids may make more sediment-bound nutrients available for plankton to grow, and the shading effect may be limited by the shallowness of the bay (EPA 2011).

Toxicants

Toxicants are chemical contaminants that can have toxic effects on biota, and include metals, aromatic hydrocarbons and biocides (ANZECC and ARMCANZ 2000). Toxicants generally have localised impacts in marine and estuarine environments, usually close to the mouths of rivers and creeks, or point-source discharges (EPA 2001), although this assessment does not reflect more recent land-use changes, particularly in the northern parts of the catchment.
The types of chemicals thought to be of most concern for Western Port are pesticides from agricultural run off (McKinlay et al. 2008), veterinary pharmaceuticals and estrogens from dairying (Fisher and Scott 2008), and hormones from sewage effluent and septic tanks (K. Hassell, CAPIM, pers. comm.). Heavy metals associated with industry and boating may also be a problem. Toxicant monitoring in Western Port has historically been very limited. While a significant number of overseas studies have investigated the growing number of human-derived toxicants in marine and estuarine waters, there is little comparable research being undertaken in Victoria.

Here we provide a broad overview of toxicant sampling within Western Port and, because these different groups of toxicants enter through different pathways and have different environmental fates, we consider metals, hydrocarbons and pesticides/biocides separately.

CAPIM

Monitoring of toxicants in the sediments of several rivers and creeks entering Western Port has been conducted by the Centre for Aquatic Pollution Identification and Monitoring (CAPIM). Sampling was undertaken near the mouths of Watsons Creek, Merricks Creek, Cardinia Creek and Bunyip River in January and June 2010, and in December–January 2010–2011, and an additional site at Warringine Creek was surveyed in the last sampling round. These data include an extensive range of heavy metals, nutrients, total petroleum hydrocarbons, synthetic pyrethroids, organophosphorus pesticides, organochlorine pesticides and triazines (CAPIM, unpublished data).
As part of CAPIM's research it is anticipated that levels of endocrine-disrupting chemicals (EDCs), specifically natural and synthetic estrogens, will also be measured in waters of Western Port using appropriate biological indicators (most likely fish and macroinvertebrates).

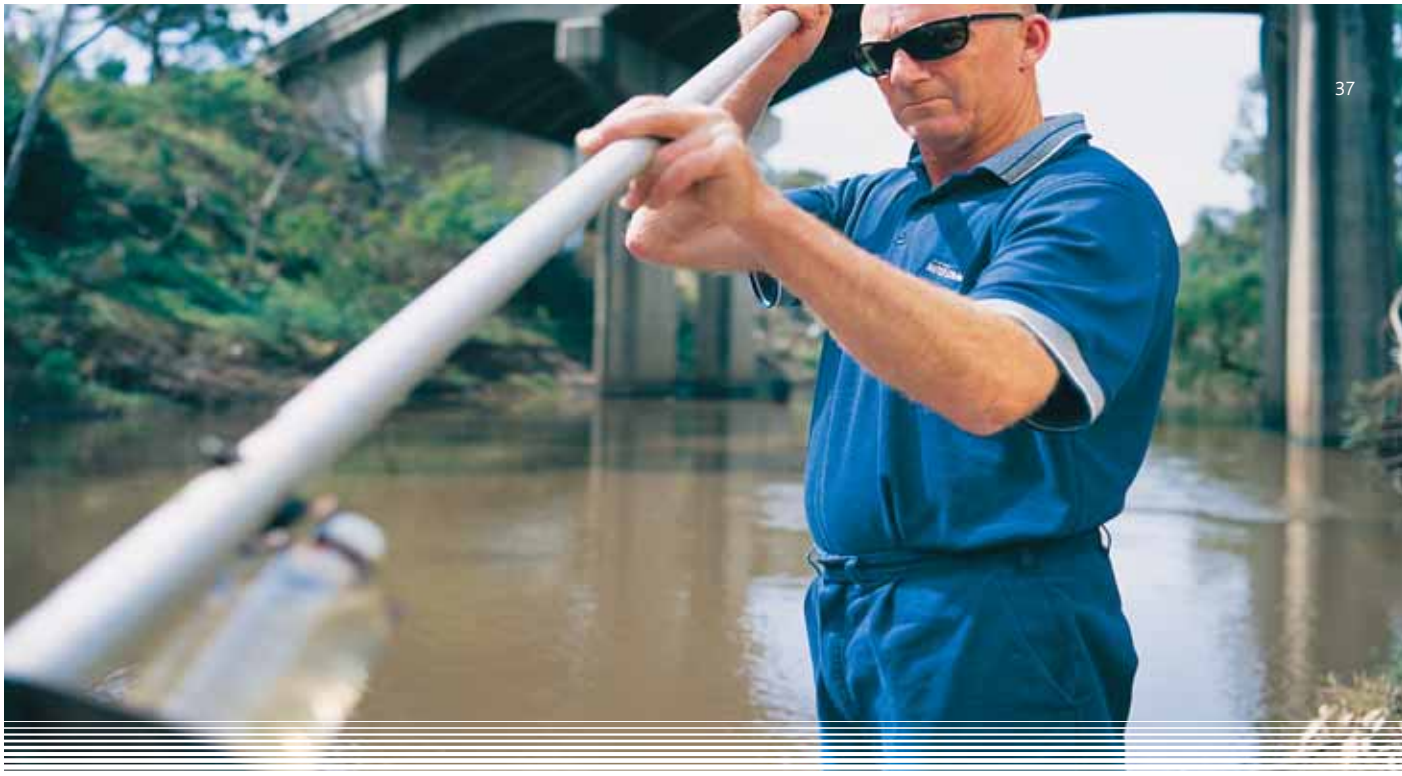
Sediment monitoring to date shows elevated levels of heavy metals at some of the sites listed above, but no detectable levels of any of the remaining toxicants tested except at Watsons Creek, which had a very low level of the fungicide Boscalid. Of those heavy metals with ANZECC trigger levels, elevated concentrations of arsenic, mercury, lead and zinc were found in some areas (ANZECC and ARMCANZ 2000). A number of other metals without ANZECC identified trigger values (e.g. aluminium, selenium and cobalt) were higher at some of the above sites, particularly Merricks Creek and the Churchill Island wetlands, relative to other sites in Western Port and Port Phillip Bay. These data are to be published in 2011.

EPA

EPA measures toxicants in the water column as part of its ongoing fixed-site monitoring program, but testing is limited to a suite of heavy metals (EPA 1996). EPA monitoring and related modelling show that the highest concentration of toxicants occurs in the north-east of the bay, with levels closely linked to catchment inputs along the northern shores of Western Port. The Bunyip, Lang Lang and Bass rivers are particularly significant sources of toxicants (see Chapter 4).

Melbourne Water

Melbourne Water has an extensive monitoring network in the Western Port catchment, including 32 sampling sites along waterways that drain into Western Port. These data include a range of heavy metals, including cadmium, copper, lead, and chromium.



Heavy metals

There have been no large-scale surveys of metals in Western Port waters, but fixed-site monitoring (at three sites) indicates that metal concentrations in surface waters have remained low and unchanged for the period 1984–1996 and are similar to concentrations measured in the 1970s (EPA 1996). Cadmium, lead, zinc and mercury are often below their limit of detection. Lead concentrations in 1998 were comparable to those in Port Phillip Bay, but maximum values were higher than those reported for coastal waters elsewhere (EPA 1998b). In 2009 the mercury level at Hastings and zinc level at Corinella did not meet the SEPP objectives, although they were still below ANZECC trigger levels (EPA 2011). Elevated concentrations of metals have been found in biota from some areas, particularly in the lower North Arm adjacent to industrial sites, and to a lesser extent in the Upper North Arm (EPA 1996).

Copper is the most common biocide used in antifouling paints and is released through leaching and through hull maintenance (e.g. hull scraping). Most commercial copper-based antifouling paints also contain cobiocides, or booster biocides, to increase their efficacy against microfouling and algal slime (Srinivasan and Swain 2007). Copper and cadmium levels at Hastings jetty were significantly greater in 1997–98 than in the 1970s, with zinc and lead unchanged (Phillips 1976, Webb and Keough 2002).

Data for metal concentrations in sediments are patchy, but studies show higher concentrations near input sources, industrial areas in North Arm, particularly around Hastings, and in mudflats in the Upper North Arm (Shapiro 1975a). Limited data collected by EPA at three fixed sites since 1990 (not at sites of likely high deposition) show low metal concentrations in sediments (EPA 1996, 2011). A more recent study of toxicant concentration in sediments showed most metals, organics and organometallics were below ANZECC (2000) guidelines for sediment quality (Rees *et al.* 1998). Elevated arsenic concentrations (thought to be of geological origin) shown in the Rhyll segment, around Crib Point, and in streams draining the Koo Wee Rup swamp in the north (Chapter 4). EPA recommended a bay-wide survey for heavy metals (including copper, lead and zinc) and tributyl tin in sediments and biota, particularly near industries and stream mouths (EPA 1996). We further discuss the need for this in Chapter 15, with a focus on a broader range of toxicants. Metals have high affinities for fine-grained sediments. The concentration of metals may therefore be influenced to some extent by processes governing sediment transport and deposition (OzCoasts 2010a).

Pathways

Heavy metals often originate in the catchment and are transported to Western Port via rivers and streams. Melbourne Water data show generally low levels in most Western Port drainages, with some elevation of individual metals in the north-west (e.g., copper and arsenic in Watsons Creek and Wylies Drain). Other significant pathways are industry effluent, resuspension from dredging activity and dumping of spoil, and anti-fouling paint on boats, ships and submerged infrastructure through leaching, scraping or paint spill.

Likelihood

Toxicity effects in Victorian marine and estuarine ecosystems are usually restricted to the proximity of point discharges (EPA 2001). Increases in boating activities might lead to localised increases in copper, but no other major changes are projected. Point source discharges of industry effluent will decrease because the volume of wastewater from several sources is to be reduced by 280 million litres per year as part of South East Water's Somers Recycled Water Project.

Hydrocarbons

Western Port is subject to much smaller and less complex discharges of petroleum-containing wastes than Port Phillip Bay. Sources of petroleum hydrocarbons are discrete ship and shore-based inputs and diffuse urban and industrial inputs (Burns and Smith 1982), although again, most of the data collection is now very dated.

Testing for hydrocarbon levels has been undertaken mainly in the Lower North Arm around the port and industry nodes. Heavy industries along this shoreline include a gas fractionating plant, a steel mill and a site of a previous oil refinery decommissioned in 1985 (M. Richards Department of Transport, pers. comm). The Port of Hastings jetties are used for exporting crude oil and LPG products and importing petroleum. BHP (BlueScope) utilise a wharf for the import/export of steel products (Meyrick and Associates 2007). The bay receives discrete, chronic low-level inputs of petroleum hydrocarbons from tanker activities, industrial effluents and boats (outboard motors), mainly in the North Arm (Burns and Smith 1982). Naturally occurring sources of biogenic hydrocarbons are also found in seagrasses, macroalgae and other ecosystem components (Burns and Smith 1982).

Monitoring in the 1970s demonstrated discrete chronic low-level inputs of hydrocarbons and identified the major source of oil contamination as refinery effluent entering the North Arm. Analyses of mussels from remote areas generally showed no detectable petroleum residues. A detailed survey of sediments confirmed that Western Port was free of oil pollution except in areas of chronic industrial discharges and some boating areas (Burns and Smith 1977). Repeat monitoring after five years showed oil in both mussels and sediments in the North Arm, but these were at relatively low levels and originated from specific point sources.

Areas of the bay remote from most human activities showed no detectable petroleum hydrocarbons, as in earlier surveys (Burns and Smith 1982). More recent but limited studies indicated very low levels of hydrocarbons around Long Island Point (CEE 1993, cited in EPA 1996). Oil spills have negative impacts on marine biota from the toxicity of oil (particularly products such as diesel fuel), smothering of birds (particularly crude, lubricating, and heavy fuel oils) and by killing vegetation (Toll Western Port 2001). Shellfish are vulnerable to tainting by oil and are particularly vulnerable to dispersed oil when dispersants have been used to protect other resources (Toll Western Port 2001). Recent incidences near Western Port include the discharge from a ship of 30 000 to 40 000 litres of waste oil sludge from the ballast water tanks into Bass Strait, about nine nautical miles off Phillip Island. Another 900 litres of hydraulic oil was spilt from a channel dredging vessel at Point Lonsdale (Port Phillip Bay).

The frequency of small oil spills in Western Port is likely to be significantly below that recorded in Port Phillip Bay, where spills of 5 L or less occur almost daily and spills of more than 100 L occur less than once a month (Melbourne Water 2009). Small boats with two-stroke engines and personalised water craft are high polluters relative to their engine size and usage (Environment Link and Vehicle Research and Design 2007). These engines emit 10–20% of the fuel–oil mix into the water. Petrol discharged into the surface water evaporates and contributes to air pollution, while heavier oils and greases remain on the surface for longer periods and some is assimilated into sediments. Unlike countries overseas, Australian outboard motors are not subject to emissions regulations (Environment Link and Vehicle Research and Design 2007).

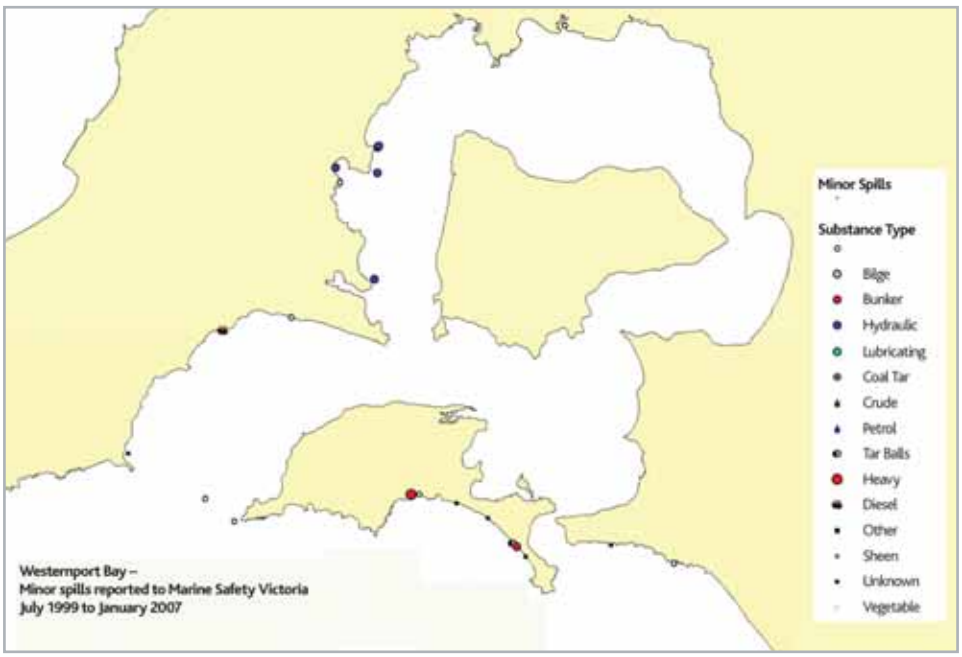


Figure 3.4 Minor spills of oils and other substances reported to Marine Safety Victoria, July 1999 – January 2007. (Source: EPA (2008)).

Pathways

Petroleum hydrocarbons may enter Western Port through oil spills from tanker accidents, pipeline accidents or malfunctions, or spills of fuels or lubricants from shipping and boating activities such as refuelling and bilge pumping (EPA 1998a, Toll Western Port 2001). Oil spills may originate from commercial ships other than those involved in petroleum transport. Other pathways are stormwater and diffuse urban run-off, and *in situ* discharges from boat engines, particularly from two-stroke engines (EPA 1998a).

Likelihood

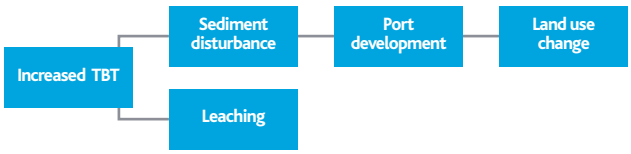
The strong tides and shallow waters of Western Port render this environment vulnerable to large oil spills (Toll Western Port 2001). Vulnerable resources include aquaculture facilities, penguin rookeries (particularly on Phillip Island), mangrove swamps and recreational beaches (Toll Western Port 2001), although other bay assets may also be vulnerable. Minor spills may occur from time to time (Figure 3.4), and equipment and a vessel for dealing with oil spills are located at Stony Point jetty. Large inputs of hydrocarbons would be associated with major spills or leaks, either from within Western Port or from vessels in Bass Strait. The likelihood of major oil spills from large tankers has declined since the Gippsland pipelines were built (M. Richards Department of Transport, pers. comm.) A major increase in commercial shipping may increase the likelihood of minor oil spills, but managing this risk is not a scientific issue.

TBT

Organotin compounds often receive special attention in marine environments. Historically, they were widely used in anti-fouling paints on a wide range of commercial and recreational vessels, docks and marinas. The main compound, tributyl tin (TBT), is an endocrine disruptor known to stimulate the development of male reproductive structures in female gastropods ('imposex'). TBT acts by stimulating the production of testosterone, causing females to grow a penis and/or vas deferens, reducing female reproductive capacity and causing premature death. These effects can have serious consequences for populations of species that depend on local reproduction for their persistence. TBT is known to cause thickening of bivalve shells and is highly toxic for crustaceans. Rates of denitrification driven by sediment microbes are also sensitive to TBT at bioavailable concentrations (Dahllof *et al*. 1999a,b).

Pathways

Figure 3.5 Exposure pathways for changes in tributyltin levels.



Likelihood

Organotin antifouling compounds have been largely phased out. Since 1989 their sale and use in Victoria has been restricted, and since 2008 an International Maritime Organisation convention has prohibited the application of organotin compounds on ships and associated structures, or required surfaces to be sealed to prevent leaching (IMO 2010). TBT has not been applied on any Australian Navy vessel for some time and has either been removed or fully encapsulated on navy vessels since 2006 (A. Scardino, Department of Defence, pers. comm.). TBT paints are not applied at any facilities within the Port of Hastings (Toll Western Port 2001).

In water TBT decays to less toxic forms in days, but it is more persistent in sediments, where its half-life is months to years (EPA 2004). Although TBT is relatively stable in anaerobic sediments, it probably decomposes in aerobic sediments (OzCoasts 2010a). Although TBT use is now limited world-wide, pollution by TBT is still considered a problem in parts of the world subjected to heavy shipping traffic, particularly around ports used by large commercial vessels (OzCoasts 2010). It is unlikely that TBT poses a threat to the health of marine habitats in areas outside commercial ports.

In Western Port, levels resulting from leaching from existing antifouling coatings are likely to be very low but could rise slightly with increasing numbers of large vessels. There are likely to be levels in sediments in the North Arm where there have been activities such as vessel maintenance. These areas are likely to be very localised, and the contaminants bound within sediments. In 1993 TBT was been found at elevated levels in water and biota in the vicinity of boat marinas and moorings (Newhaven and Hastings) but not at sites outside these areas (Daly and Fabris 1993). A 2004 EPA review found that that the majority of water and sediment samples in Western Port (and Port Phillip Bay) still exceeded the recommended ANZECC (2000) guidelines, thus presenting a risk of lethal and sublethal effects on marine biota (EPA 2004).

There is still some doubt about the magnitude and extent of the risk of persistent TBT for the reduction of denitrification rates in sediments (Dahllof *et al*. 1999a,b; EPA 2004). Significant effects of TBT on nutrient fluxes have been recorded at concentrations 100 million times less than the maximum concentrations recorded in Western Port (Environment Australia 2004).

Management

The release of sediment-bound TBT through sediment disturbance would only come from substantial dredging projects. The consequences of TBT exposure are such that the release of these compounds would automatically be considered in the statutory approval processes associated with such dredging. This would be expected to include a risk assessment that would incorporate sampling to measure current levels of contamination, risks of release, and actions required to minimise those risks.

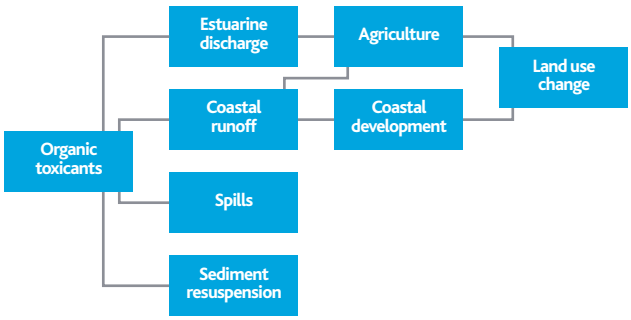
Organic toxicants

Organic toxicants include a variety of synthetic (and some natural) compounds such as polychlorinated biphenyls (PCB) and dioxin, which are considered to be severely damaging to human health, wildlife, and aquatic species. EPA Victoria does not include synthetic organic compounds in its toxicant monitoring in marine waters or sediments. There is limited information on the concentrations at which synthetic organic compounds would have deleterious effects in estuarine and marine ecosystems.

Endocrine disrupting chemicals (EDCs) are a group of organic toxicants that may interfere with the endocrine system of animals and have gained increasing attention for their potentially adverse effects in aquatic organisms (Depledge and Billinghurst 1999). A lack of research makes it difficult to characterise the risk associated with EDCs (Kookana *et al.* 2007). While there are no published reports on the effects of EDCs (other than TBT) in the Victorian marine environment, several studies are underway through the Centre for Aquatic Pollution Identification and Management (CAPIM). Proposed research includes an investigation of the potential effects of EDCs on Black Bream and the development of a fish bioindicator.

Pathways

Figure 3.6 Exposure pathway for organic toxicants.



Likelihood

There are no expectations of dramatic increases in contaminant inputs to Western Port, although changing land use patterns do result in alterations of the major chemicals used e.g. agriculture within catchments.

Biocides

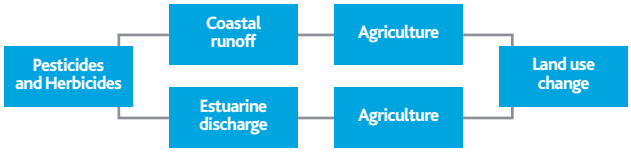
Biocides include insecticides, herbicides and fungicides. Biocides are likely to be present in waterways draining intensive agricultural areas, particularly from horticultural sources such as viticulture, orchards and vegetable growing (EPA 1996). A mid 1970s study of water and sediments revealed very low concentrations, in small numbers of samples, of organochlorine and organophosphate pesticides (Bryant *et al.* 1975). Low concentrations were also found in a small number of fish. More recently, elevated levels of pesticides have been found in streams, such as Watsons Creek, that pass through market gardens (Melbourne Water and EPA 2009). Measurements in streams discharging into Western Port have also been undertaken by CAPIM in 2010–2011, sampling that includes estuarine sediments (see above).

EPA Victoria recommended measuring the concentrations of organic herbicides near input streams and marinas, because some herbicides are being used as replacements for traditional metal-based anti-fouling agents (EPA 1996).

One major uncertainty about biocides is a clear understanding of the risks posed by individual chemicals alone and their effects when in mixtures. An improved capacity to detect the presence of these toxicants and new techniques for assaying their effects may well increase the list of chemicals of concern.

Pathways

Figure 3.7 Exposure pathways for pesticides and herbicides.



Likelihood

Concentrations of biocides are likely to remain high in catchments draining intensive horticultural and agricultural areas. Although systematic data are lacking for the entire bay, CAPIM data will be useful in establishing biocide levels at specific inputs, and may help to determine the direction of future monitoring.

Pathogens

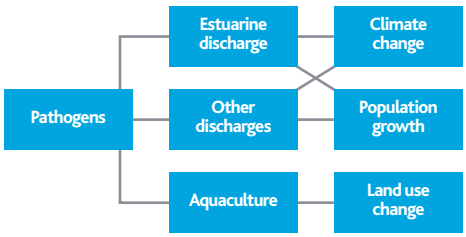
Pathogens are usually a concern when they increase risks to human health, but outbreaks can result in high mortality in other species, and this is a concern if these species are ecologically important or valued for other reasons. A few well-documented cases have seen outbreaks cause cascading ecological change (e.g. substantial loss of coral and replacement by algae across much of the Caribbean following loss of the urchin *Diadema*) or important economic loss (e.g. recent abalone deaths in Victoria). In such cases an important issue is the possible vectors for pathogens, and often the likely paths by which pathogens may spread.

Dispersion modelling conducted by EPA/ASR for 2004–05 found concentrations of pathogens (*Enterococci* – bacterial indicator for faecal contamination) around Merricks on the Mornington Peninsula coast and Cowes on the north-eastern end of Phillip Island (Lee *et al.* 2009). This modelling approach was used for pathogens of concern for human health, but it could be used for pathogens of concern for ecosystem health.

Diseases in the marine environment are potentially important but greatly understudied (Harvell *et al.* 2002, Lafferty *et al.* 2004). We include them as a threat because they are potentially important, and because the risk may increase increase in future with increased temperatures from climate change.

Pathways

Figure 3.8 Mechanisms that may alter the risk from pathogens.



Likelihood

This risk is likely to increase in future, but it is not possible to quantify the increase. We discuss this threat in slightly more detail in Chapter 13.

Salinity

Because it is a semi-enclosed bay, Western Port is subject to alterations in salinity at a range of scales. Long-term records in Western Port show strong seasonal and inter-annual variations in salinity of greater magnitude than Bass Strait. There also is a potential trend for increasing salinity as a result of decreased rainfall, increased evaporation and reduced inflows – due to diversion of water by re-cycling and other human uses, even allowing for possible increases in stormwater runoff from urbanisation (Lee *et al.* 2009). It would be interesting to contrast these projected changes, and those of the recent drought period, with the salinities that would have occurred prior to the drainage of Koo Wee Rup swamp, when there were reduced inflows.

Any future differences in salinity are likely to further affect the exchange efficiency with Bass Strait because of the altered density-driven circulation. This would thereby change the flushing capacity of Western Port Bay (Lee *et al.* 2009).

Salinity variations can have ecological consequences. Extreme low salinity events can cause mortality of organisms that rarely encounter salinities greatly different from oceanic waters. Western Port does have variable salinities in northern sections, so it might be expected that organisms there can deal with some variation, but extreme rainfall events can cause very sharp declines in salinity and still cause stress. Increased salinity can also have deleterious effects, but in oceanic waters, small changes in salinity have little impact (Roberts *et al.* 2010). Changes to salinity are perhaps a greater concern if they result in altered hydrodynamic patterns, which may alter flushing times and critical processes such as ecosystem connectivity.

Pathways

Reduced rainfall and increased evaporation resulting from climate change are likely to increase the salinity in some areas of the bay, with a possible reduction in the magnitude of flushing with Bass Strait. Increased pulses of freshwater inputs caused by predicted increase in heavy rainfall events (and urbanisation if left untreated) may also alter the long-term variation in salinity. There are no hypersaline inputs into Western Port at present, but a desalination plant is under construction not far from the Eastern entrance to Western Port. Modelling done as part of the EES for that project and results from elsewhere in Australia suggest that there will be little input into Western Port.

Likelihood

Climate projections would suggest changes to patterns of salinity fluctuation, with a bigger focus on more episodic events.

Acidity

Change to seawater acidity has potentially serious consequences for a range of marine organisms. Seawater acidity has two sources — increased atmospheric CO_2, primarily from fossil fuel utilisation, and the leaching and run-off of naturally occurring coastal acid sulfate soils (CASS). CO_2 reacts with water to release positive hydrogen ions into seawater which lowers the pH. This uptake of CO_2 has led to a reduction of the pH of global surface seawater of 0.1 units, equivalent to a 30% increase in the concentration of hydrogen ions (Raven 2005).

CASS contain metal sulfides, principally pyrite (FeS_2) and exposure of them to oxygen and water can generate sulfuric acid, which can have negative effects on aquatic organisms (EPA 2009). Acidic leachates can also mobilise toxic levels of iron, aluminium and manganese from soil and sediment, with the potential to create further deleterious impacts on organisms in the receiving environment (DPI 2003).

The biological effects of increased acidity are variable. The effects of increasing CO_2 are manifest at a global scale, and it has been suggested that reversal of ocean chemistry to pre-industrial levels would take thousands of years (Raven 2005). CASS has more localised effects. At very small scales, sharp falls in pH (i.e. increases in acidity) can be lethal, but changes smaller than this can affect the physiology of organisms.

The most widely discussed consequence is the possible impairment of skeleton formation by organisms such as molluscs, crustaceans, echinoderms and calcifying algae that use calcium carbonate ($CaCO_3$) as part of their skeletons (Orr *et al.* 2005). Calcifying marine organisms generally precipitate dissolved ions into solid $CaCO_3$ in one of two forms — aragonite and calcite. The $CaCO_3$ ion is usually at supersaturated concentrations in surface waters, but when carbonate becomes undersaturated any structures made of $CaCO_3$ are vulnerable to dissolution (aragonite being the more soluble form) (Langdon and Atkinson 2005). The effect of increased CO_2 is to raise the depth at which seawater is saturated with respect to aragonite and calcite, thereby decreasing calcification rates and increasing dissolution rates.

Small changes in pH make the construction of skeletal material more 'expensive', requiring more of an organism's energy budget to extract Ca from seawater and incorporate it into skeletons. As pH falls, not only does skeleton formation become more difficult but existing skeletons may start to dissolve. The extent of growth restriction or loss of skeleton is variable from species to species, depending in part on the form of $CaCO_3$ that is used in the skeleton (Raven 2005). It is also temperature-dependent, and in the waters of south-eastern Australia the conditions are close to aragonite thresholds (McNeil and Matear 2008).

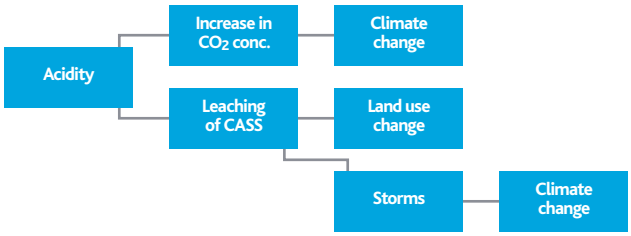
Acidity can impair other physiological processes such as fertilisation of eggs by sperm (e.g. Havenhand *et al.* 2008) and acid-base metabolism (Miles *et al.* 2007). Changes in pH can also affect the bioavailability of nutrients and toxicants (EPA 2001).

The consequences of increasing acidity are acknowledged as an area of considerable uncertainty (National Academy of Science 2010), with consequences ranging from relatively minor (if pH changes little and organisms can adapt) to catastrophic with major ecosystem changes and alterations to important fisheries (Fabry *et al.* 2008, Richardson and Gibbons 2008).

It has been suggested that organisms in shallow water coastal habitats are less likely to be affected by increasing acidity than organisms in the open ocean as they are able to cope with naturally variations in pH and CO_2 concentration. To date, studies on south east Australian species have been laboratory-based, with responses in test animals ranging from deleterious (e.g. adult mortality, impaired fertilisation and larval development in urchins (Havenhand 2008)) to undetectable (e.g. fertilisation in echinoderms and abalone (Byrne *et al.* 2010)).

Pathways

Figure 3.9 Exposure pathways for changes in acidity.



Likelihood

Climate change is expected to cause oceans to acidify, with slight changes over the coming decades but at an increasingly accelerated rate through time. The second half of this century will see much higher rates than the first half. If global emissions of CO_2 continue to rise as currently projected then the average pH of the oceans could fall by 0.5 (equivalent to a threefold increase in the concentration of hydrogen ions) by the year 2100 (Raven 2005). This would be a change that could not be ameliorated by local actions.

The disturbance of acid sulfate soils in Victoria appears to be low compared to other states, perhaps because of the relatively small area of acid sulfate soils in the state or because of under-reporting (DPI 2003). The shores of Western Port have extensive areas of potential CASS (Figure 3.10).

CASS may not be acidic if the soil that contains unoxidised metal sulfides exists in oxygen-free or waterlogged conditions (EPA 2009). Increased exposure of CASS can occur when these soils are exposed to air as a result of disturbances such as excavation, drilling, lowering of the watertable and inshore dredging (EPA 2009). Increased exposure of CASS can also occur as a result of activities associated with changes in land use, when these soils are exposed to air. The risks associated with CASS are well documented, and statutory approval processes generally require consideration of these risks and the development of appropriate management plans. CASS may also be exposed as a consequence of climate change, but the likelihood is harder to assess; rising sea levels may submerge low-lying CASS, reducing the risk of exposure, but the combined effects of sea level rise and storm surge may erode coastal sediments, exposing CASS. This increased risk is reflected in the development of the Victorian government's Coastal Acid Sulfate Soils Strategy (www.dse.vic.gov.au/coasts-and-marine/coastal-acid-sulfate-soils-strategy).

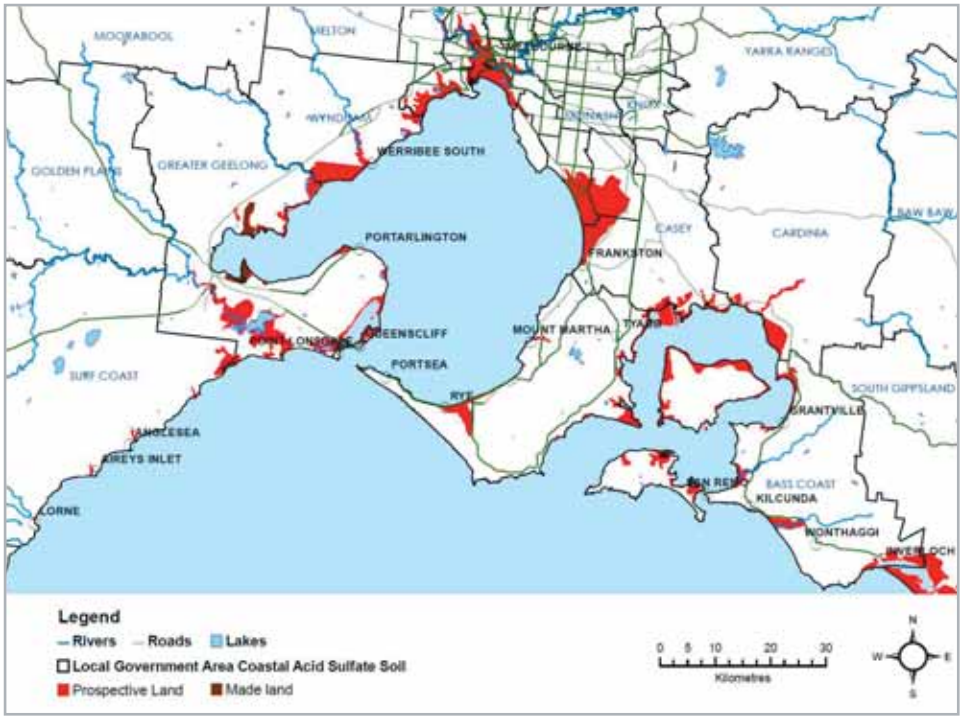


Figure 3.10 Potential CASS along the central Victorian coast, based on geomorphology. (Source: Victorian Resources Online: www.dpi.vic.gov.au.)

Hydrodynamic and atmospheric variables

Sea level rise and variability

The current scientific consensus suggests that, under a 'business as usual' scenario, mean sea levels are likely to increase by 0.8 m by 2100 CSIRO, 2009. There is, however, considerable uncertainty about this figure, with rises well beyond this being plausible (Church *et al.* 2010). Current rates of increase are of the order of 3 mm annually, but there is considerable local variation and some coastlines may experience rates quite different from this global mean. Sea levels will also be more variable because of changes in storm frequency and intensity (Church *et al.* 2010). Most coastal areas in the Western Port region are vulnerable to sea-level rise and increased intensity of storm surges. These effects can be exacerbated if storm surges coincide with floods. Potential impacts include accelerated erosion, temporary inundation and degradation of foreshore areas, and recession of the coastline (Kinrade and Justus 2008).

Sea-level rise maps for 2100 have been developed by the Australian Government under three sea-level rise scenarios — 0.5 m, 0.8 m, 1.1 m increase from 1990 levels by 2100 — for various coastal areas around Western Port (OzCoasts 2010c). Under these scenarios, low-lying areas, particularly those towards the north of Western Port (e.g. Warneet), are likely to be inundated at least once a year by 2100 (Figure 3.11). Detailed predictions of local sea rise require additional fine-scale data, and the Victorian Government's Future Coasts program will develop these fine scale assessments, particularly through its Third Pass (www.climatechange.vic.gov.au).

Pathways

Sea-level rise is directly linked to climate change, which in turn is linked to human activities. The three major pathways are thermal expansion of the oceans as they warm, increased sea water volume as glaciers and ice caps melt, and melting of the polar ice sheets. These reflect the same underlying cause, and their relative contributions are still the subject of some discussion.

Likelihood

Sea levels are currently rising, with the annual rate increasing over the past decade. From 1993 to 2003, global sea level rose by about 3.1 mm per year, compared to 1.8 mm per year for 1961–2003 (Church *et al.* 2008, DCC 2009). Recent research indicates that a 1.1 m scenario by 2100 may not reflect the upper end of potential risk (Church *et al.* 2010).

For Stony Point the predicted 1-in-100 year storm tide height in 2070 (yellow dashed line, Figure 3.12) is around 0.7 m higher than the present value. Of that increase, around 75 to 80% is due to sea-level rise, while the remainder is due to wind speed increase. The frequency of extreme events also increases in the scenarios shown: the control 1-in-100 year event would be exceeded on average once every 30 years in 2030 and once every 5 years in 2070 (McInnes *et al.* 2009b). Melbourne Water uses slightly higher values for its planning, to take more account of wave action (Melbourne Water 2010 and see Chapter 4).

Figure 3.11 Digital elevation model for Western Port, showing low-lying areas. Source: DSE (2010.)

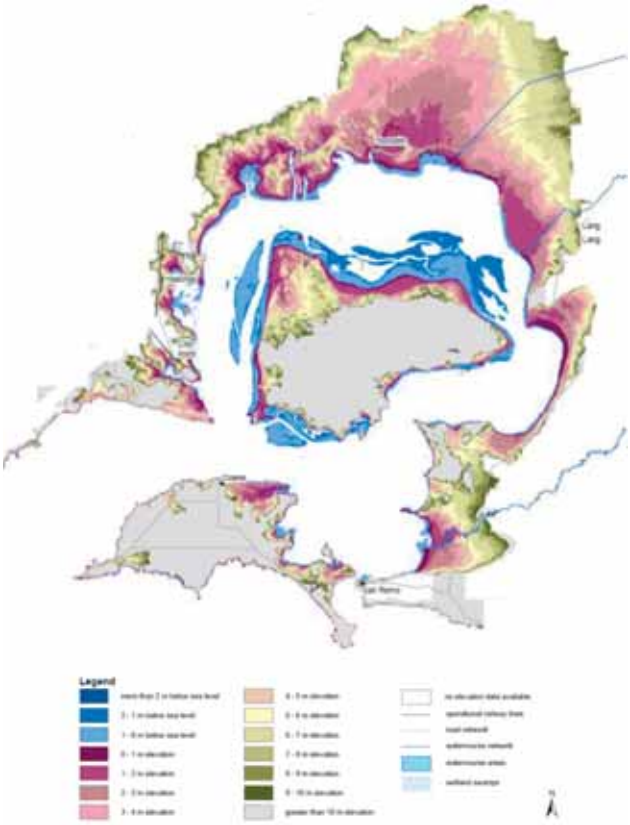
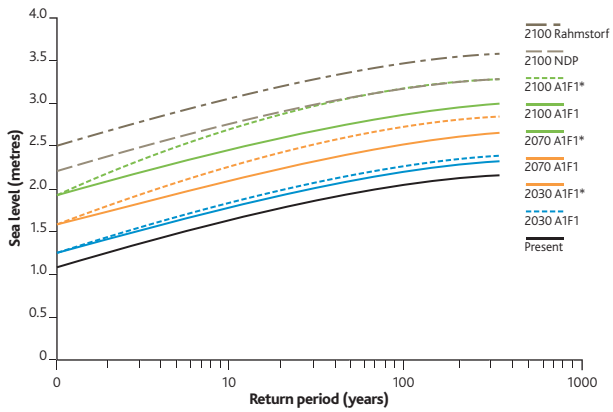


Figure 3.12 Storm-tide return period curves for Stony Point for 2030, 2070 and 2100 under the high climate change scenario. The A1FI scenarios are from AR4, and those with an asterisk include an allowance for wind speed. The NDP scenario is based on the upper sea-level rise developed for the Netherlands Delta Committee. Rahmstorf is the upper estimate from Rahmstorf (2007). (Sources: McInnes et al. 2009 and K. McInnes pers. comm., cited in DCC 2009.)



Increased UVB

Ultraviolet radiation (UV) comprises UVA and UVB. Recent stratospheric ozone depletion caused by human activities has increased the UVB flux at the Earth's surface. UVB is particularly harmful to reproduction, development and behaviour in many organisms, including marine invertebrates (Häder *et al.* 2011). Furthermore, UV has been shown to significantly interact with temperature and salinity to increase the severity of responses (Przeslawski 2005).

Some of the deleterious effects of UVB on aquatic organisms are DNA damage, developmental abnormalities, behavioural changes, and increased embryonic mortality in corals, sea urchins, algae and encapsulated molluscs (Przeslawski 2005). Biologically harmful UV can penetrate more than 20 m into water, so that submerged and intertidal plants and animals can all be affected. It is anticipated that there will be a disappearance of UV-intolerant seagrass species from shallow waters as UV radiation levels rise (Hobday *et al.* 2006).

Pathways

UV exposure may increase as a function of changes to the ozone layer, and from increased numbers of hot, sunny days resulting from climate change. Current predictions for a range of climate change scenarios indicate this has a high likelihood of occurring (see Chapter 4).

Increased temperature

The global average surface temperature has risen by 0.74°C over the past century while in Australia, there has been a 0.9°C warming since 1950 (CSIRO 2009). While ocean temperatures rise more slowly than air temperatures, CSIRO and the Bureau of Meteorology predict significant sea surface temperature increases of 0.3–0.6°C by 2030 for most of Australia, with faster warming (0.6–0.9°C) in the southern Tasman Sea. By the end of the century sea surface temperatures could have risen by up to 2°C along the south coast and 2.5°C elsewhere. Increased sea temperatures contribute to rising sea levels and may have negative impacts on marine ecosystems.

At a more local level, predictions for annual land temperature increases are for 0.2 to 1.4°C by 2030, and 0.7 to 4.3°C by 2070 (Brooke and Kinrade 2006). Other projections are for a significant increase in the number of hot days (> 35°C), a decrease in the annual rainfall but an increase in the frequency of heavy rains, more frequent and severe droughts, and increased evaporation. Along the coast there is likely to be an intensification of winds associated with more intense and frequent low-pressure systems, particularly in winter (Brooke and Kinrade 2006).

Intertidal organisms are likely to suffer the effects of increasing air temperatures through heat stress and desiccation at low tide. Sessile species of algae and animals are particularly at risk, and early life-history stages such as eggs and larvae are more vulnerable to stressors such as temperature and desiccation.

Changes in sea surface temperatures may cause tropical and temperate species to expand their ranges southward, potentially changing the structure of marine and coastal ecosystems, with risks to cool-water species such as kelps. Temperature-induced changes to ocean currents are likely and would have significant implications for marine and coastal ecosystems and processes (e.g. productivity and dispersal), but these changes are difficult to predict (Hobday *et al.* 2006).

Increased seawater temperatures have already been shown to lead to range expansions of some more northerly (warm water) species into temperate southern waters (see under 'Pest organisms').

In addition to range expansion and increased marine pest incursions, higher seawater temperatures may cause metabolic and respiratory stress for organisms and render them more susceptible to disease and disturbance (i.e. decreased resilience). Cool-water species may also suffer impaired embryonic and larval development and dispersal.

Pathways

Climate change.

Likelihood

Temperature increases have already been documented and the main questions are about the rate of increase.

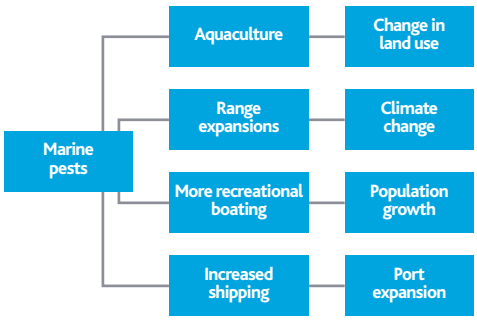
Pest organisms

Species that are not indigenous to an area can cause severe problems, which have been document widely elsewhere in Australia (e.g. Hewitt *et al.* 2004, Ross *et al.* 2007) and overseas (e.g. review by Williams & Grosholz 2008), and their effects can include displacement of natural species by a range of mechanisms and habitat modification, and they can have considerable economic costs.

Pathways

Non-indigenous species can arrive by several pathways (Figure 3.13). A primary vector is international and local commercial shipping, and there is a long history of introductions via this pathway. A substantial number of non-indigenous species were already present in Port Phillip Bay when the first formal surveys were done there in the late 19th century (Hewitt *et al.* 2004). Early introductions are presumed to have occurred primarily through hull fouling by algae and sessile invertebrates such as barnacles. In more recent times, ballast water has been recognised as an important source of pest organisms. In Australia, hull fouling is estimated to contribute to 60% of invasive species translocations, and ballast water 24% (Hewitt and Campbell 2008). At a local and regional scale, the movement of fishing equipment and activities associated with aquaculture are potential pathways for the introduction of pest organisms. Recreational activities are also potential pathways, through transport on recreational vessels and possibly recreational equipment.

Figure 3.13 Exposure pathway for non-indigenous organisms.



Non-indigenous species can also spread by non-human mediated mechanisms. For example, the cordgrass *Spartina* has been established in Gippsland and has encroached into the eastern end of Western Port through natural dispersal. Similarly, the seastar *Asterias amurensis* is well established in Port Phillip Bay, and an outbreak was recorded in Gippsland at Andersons Inlet, probably a result of larval dispersal from Port Phillip Bay.

Most of the focus to date has been on the introduction of species from outside Australia, but climate change provides another pathway for ecological change. Warmer water may result in the waters of Western Port becoming suitable for species from more northern latitudes, so the important question is whether there is a dispersal pathway for such species. This pathway may be possible under existing oceanographic conditions, although Wilsons Promontory and areas to the east provide some barrier to this dispersal. Climate change may also result in altered ocean currents, so dispersal pathways may change. Although not yet recorded in Western Port, some New South Wales species have already extended their range to Tasmania and Victoria and are capable of breeding there, notably the urchin *Centrostephanus* (Ling 2008) and various fish (Last *et al.* 2011). These temperature-tolerant invasive species could out-compete and exclude native species from coastal waters (Sorte *et al.* 2010).

Likelihood

Non-indigenous species are already well established in Western Port, with Hastings having a substantial number of such species (Currie and Crookes 1997, Parry and Cohen 2001, Webb and Keough 2002). These introductions presumably occurred as the result of a range of shipping and boating activities. The potential for more such introductions may rise if ballast water risks are not managed effectively and as population growth places more demands on boating facilities in Western Port.

One species of concern, the fanworm *Sabella spallanzanii*, has been recorded on mussel ropes in an aquaculture zone in the south-west of Western Port (Cohen *et al.* 2000). Improved treatment of mussel ropes is expected to prevent further translocations of this species to Western Port, and the aquaculture zone is used less than in previous years.

It is also possible for non-indigenous species to spread from established populations in Port Phillip Bay without human intervention. Modelling of Bass Strait water circulation indicates that larvae can be transported from Port Phillip Heads to the entrance of Western Port in less than four days under favourable wind conditions (Jenkins *et al.* 2000). Many species of concern have dispersive stages that are longer than four days, and could survive such a trip.

There are at least two pest species of concern that are already established in Western Port or may spread to there in the future. *Spartina anglica* and *Spartina* × *townsendii* have invaded the northern shoreline around The Inlets and Bass River, and could become more widespread (Chapter 10). The New Zealand screw shell *Maoricolpus roseus* is already present in Bass Strait. M. roseus has been found in high densities in Point Hicks Marine National Park, and there is anecdotal evidence that it has reduced the diversity of aquatic fauna at this site (Heislers and Parry 2007). Although it is a deeper-water species, its range has also extended to the Derwent Estuary in Tasmania, where it has affected important ecosystem services. This suggests that there are no barriers to it extending into shallower waters, and an extension in Bass Strait westwards past Wilsons Promontory would increase the risk to Western Port. The green macroalga *Caulerpa taxifolia* is an invasive marine species which has caused considerable problems in the Mediterranean, and since 2000 has also established in 13 estuaries or coastal lakes in New South Wales (Taylor *et al.* 2010). It is a declared noxious aquatic species in Victoria with strict controls on its movement and disposal (DPI 2004). The likelihood of this alga reaching Western Port, and the consequences for local ecosystems, are unknown.

None of the large pests found in Port Phillip Bay (e.g. *Undaria pinnatifida, A. amurensis, S. spallanzanii*) were found during the BlueScope marine biological monitoring program (Marine Science and Ecology 2009). It has been suggested that the absence of pest species around the Port of Hastings, particularly those now common in Port Phillip Bay, may be due to strong tidal currents in Western Port (Currie and Crookes 1997).

Habitat loss and fragmentation

Habitat for marine organisms may be physical features such as rocky reefs or sediments, but also seagrasses, mangroves, large brown algae, saltmarsh plants and other organisms. These habitat-forming organisms may be lost as a result of factors such as increased suspended sediments, and when they are lost a suite of other species that depend on them may also disappear.

Habitat may be lost through physical changes to land use, as particular habitats are altered or destroyed for other purposes. Habitat may also be created by the spread of invasive species, and this too may have substantial effects on biodiversity and some ecosystem processes. In Western Port, there have already been considerable changes to mangrove, seagrass and saltmarsh habitats.

Fragmentation poses challenges for species that depend on a particular habitat. Habitat patches may vary in quality, and small habitat patches may support fewer individuals. Individual members of the population may need to move more frequently between habitat patches, particularly at breeding time, and when the intervening habitat is less suitable (e.g. bare sand rather than seagrass) moving between patches may increase the mortality rate.

Fragmented habitats have a greater proportion that is 'edge' and interface with other habitats. More edge habitat may increase encounter rates with predators and increase the risk of invasion by pest organisms. As habitats become more fragmented their physical environment changes, which may lead to even more fragmentation. For example, small seagrass patches may be more prone to scouring and erosion, and more prone to loss, than large meadows.

Pathways

Loss of habitat occurs through urbanisation, vegetation clearing, road building, sediment input and resuspension. Habitat cover may be lost through physical changes associated with climate change, particularly by increased temperature, rising sea levels and physical destruction from increased storm frequency and severity.

Historical and contemporary agricultural use and land clearing have fragmented habitats in and around Western Port. Current and projected urban growth, and coastal engineering such as port developments may all change the coastal fringe of the bay. Away from the shorelines, the break-up of large seagrass areas has probably been caused by changes in water quality, particularly suspended sediments (Chapter 10). The other major subtidal habitats (unvegetated sediments and rocky reefs) are not likely to be fragmented.

Likelihood

Climate change, combined with the rapid urbanisation and development of the Western Port catchment, will increase the potential for habitat loss (e.g. see Chapters 8 & 9).

The rate of urban expansion in the Pakenham–Cranbourne growth area is the fastest in the state. From 1996 to 1999 this area accounted for 43% of all residential development in growth areas across Melbourne, and Casey was the fastest-growing municipality in the metropolitan area (DSE 2005). The additional habitat fragmentation that will result in Western Port from this urban expansion will be determined largely by a range of planning mitigation activities.

Seagrass losses have already occurred in Western Port, and, while not quantified, it is likely that these losses have resulted in more fragmented seagrass landscapes. Fragmentation is the likely consequence of any threat to seagrass that does not result in complete loss of meadows. It is also possible that sea level changes will cause some intertidal habitats to become fragmented.

Habitat quality

It is possible for the total amount of habitat to remain the same but its suitability for species that depend on it to diminish. This is most evident for species whose reproduction is easily disrupted, and this threat is of concern mainly for birds. For example, the area of potential nesting habitat for breeding birds might be stable, but human activities that interfere with breeding or other behaviour might reducing the suitability for nesting (Chapter 12).

Pathways

Catchment inputs, impacts associated with human visitation including trampling and other physical damage to vegetation, litter, oil spills, toxicants, disturbance of native fauna by humans and their pets, motor noise (especially boats and jet skis) (see Chapters 9 & 12).

Likelihood

Changes to habitat quality are likely to alter with land use changes, particularly urbanisation, which may bring urban environments closer to natural ones; for example, increased recreational use of Western Port could disrupt feeding and breeding patterns of birds further.

Extraction

Various materials are removed from marine environments, particularly for resource extraction, channel dredging and beach renourishment. Living material is also extracted for food or bait. In Western Port the extraction of living material is the most significant. Although commercial fishing in the bay has been phased out, recreational fishing is still very popular, and food and bait collecting in intertidal areas is common, molluscs being the main target.

Photo courtesy Julian Finn.

Pathways

Shore and boat-based recreational fishing, and collection of invertebrate animals from reef and soft sediment habitats.

Likelihood

Overall extraction rates may have fallen as a result of reductions in commercial fishing in Western Port, but estimates of recreational catches are limited (see Chapters 11 and 15). Increased population growth will place additional pressure on Western Port's waters, and the trend in extraction rates will reflect the combination of increased recreational activities and fisheries management. Trends in recreational fishing are described in Chapter 11, and intertidal harvesting is discussed in Chapters 7 and 13.

Alteration of physical coastal processes

Water movement in Western Port is driven predominantly by Bass Strait tides. Meteorological factors (winds and barometric pressure), freshwater inputs, coastal topography and seabed morphology influence the nature of currents in the bay, and the currents influence sediment transport, water quality and the biota present. Hydrodynamic processes in the bay have been reviewed by EPA Victoria (EPA 1996), and a fully integrated receiving water quality model has been developed by Lee *et al*. (2009) to investigate the effects of global climate change.

Current velocity can be critical for the dispersal of the early life stages of algae, fish and invertebrates, and thus the connectivity and persistence of marine populations. Freshwater inputs are critical for diadromous species, and salinity gradients (haloclines) may be important for larval fish entering Western Port.

Changes to wave energy and current velocity may affect the soft-sediment vegetation and epibenthic invertebrate fauna, including seagrass meadows, algal beds, and Pyura assemblages. Shallow subtidal and intertidal areas are likely to experience greater disturbance than deeper habitats.

Pathways

Climate change may slightly increase mean wave climates (wave height and energy), in line with slight increases projected in mean wind speed. Variability may also increase with greater variations in air pressure (DCC 2009). Local activities such as dredging, infilling (land claim), rock wall construction (Bird 1993) and wharf and marina construction may also alter current profiles. More variable freshwater inputs with more frequent floods could alter the bathymetry and topography of the bay.

Likelihood

An increase in the intensity of storms and floods is likely (see Chapter 4). Dredging and wharf expansion at the Port of Hastings are possible, and engineering solutions to counteract climate change impacts (e.g. the construction of retaining walls) may be explored in the future.

Cumulative impacts

These threats are often considered in isolation, possibly because this makes them more tractable scientifically. Most current management, however, involves a simultaneous consideration of multiple threatening processes, and there is a need to rank threats as a step to prioritising actions. This is particularly the case with the trend to replace sectoral management by spatial management (e.g. ecosystem-based management, integrated coastal zone management).

Even when multiple stressors are considered, it is common for them to be treated independently. It is likely that there are important interactions between stressors that make assessing risks more uncertain. In the simplest case, the water entering Western Port from its catchments contains a suite of chemicals derived from a variety of sources. With a larger number of components, the likelihood of at least one important synergy rises quickly. Toxicants enter Western Port along with nutrients and sediments, and the fate of many of those toxicants is influenced by the sediments and nutrients. In some cases the synergies may lead to more positive outcomes, such as when heavy metals are bound to organic material and become biologically unavailable. In other cases the outcomes could be negative, such as when sediment-bound metals become more available under conditions of greater acidity or reduced oxygen levels.

The potential importance of these synergies is becoming widely acknowledged in the scientific literature (Ormerod *et al.* 2010), but there remains a large gap between acknowledging the issue and having enough data to assess its seriousness.

Even without synergistic effects, a series of individual threats that are not serious individually could in combination pose a serious risk if they act simultaneously or in sequence.

There is considerable uncertainty about some of the important individual threats to Western Port. Although in the following chapters we discuss some important synergistic effects, the role of synergies and cumulative effects in general is poorly known for Western Port.

Table 3.3. Asset–threat matrix, showing the overall assessment for each asset–threat combination. The table shows classifications for threats associated with changes to water and sediment quality. Threats are classified as high (red), intermediate (orange), low (blue), or unknown (green).

Asset		Nutrients	Sediment Input	Sediment resuspension	Heavy Metals	TBT	Organics	Pesticides Herbicides	Other toxicants	Pathogens	Salinity	Acidity
Habitat Groups												
Rocky reefs	Intertidal	Low Threat	Intermediate Threat	Low Threat	Low Threat	Low Threat	Low Threat	Low Threat	Low Threat	Unknown Threat	Unknown Threat	Unknown Threat
	Subtidal	Intermediate Threat	Low Threat	Intermediate Threat	Low Threat	Low Threat	Low Threat	Low Threat	Low Threat	Intermediate Threat	Unknown Threat	Unknown Threat
Vegetated sediments	Mangroves	Intermediate Threat	High Threat	High Threat	Intermediate Threat	Intermediate Threat	High Threat	High Threat	Intermediate Threat	Intermediate Threat	Intermediate Threat	Intermediate Threat
	Saltmarshes	Intermediate Threat	High Threat	Intermediate Threat	Intermediate Threat	Intermediate Threat	High Threat	Intermediate Threat	Intermediate Threat	Unknown Threat	Unknown Threat	High Threat
	Seagrass	Intermediate Threat	High Threat	High Threat	Intermediate Threat	Intermediate Threat	Unknown Threat	High Threat	Intermediate Threat	Intermediate Threat	Low Threat	Intermediate Threat
Unvegetated sediments	Intertidal	High Threat	High Threat	High Threat	High Threat	High Threat	High Threat	Intermediate Threat	High Threat	Intermediate Threat	Intermediate Threat	High Threat
	Subtidal	High Threat	High Threat	High Threat	High Threat	High Threat	High Threat	Intermediate Threat	High Threat	Unknown Threat	Intermediate Threat	Intermediate Threat
Water column		High Threat	High Threat	High Threat	Intermediate Threat	Unknown Threat	Low Threat	Low Threat	High Threat	Unknown Threat	Intermediate Threat	High Threat
Significant Fauna												
Iconic species	Shorebirds	Low Threat	Intermediate Threat	Intermediate Threat	Unknown Threat	Unknown Threat	Unknown Threat	Intermediate Threat	Intermediate Threat	Unknown Threat	Intermediate Threat	Unknown Threat
	Seabirds	Low Threat	Intermediate Threat	Intermediate Threat	Unknown Threat	Unknown Threat	Unknown Threat	Intermediate Threat	Intermediate Threat	Unknown Threat	Low Threat	Unknown Threat
	Marine Mammals	Low Threat	Low Threat	Low Threat	Unknown Threat	Unknown Threat	Unknown Threat	Low Threat	Intermediate Threat	Unknown Threat	Low Threat	Unknown Threat
Commercial/Recreational	Fish	Intermediate Threat	High Threat	High Threat	Intermediate Threat	Intermediate Threat	Intermediate Threat	Intermediate Threat	Intermediate Threat	Unknown Threat	Intermediate Threat	Unknown Threat
Emergent Features												
	Ecosystem Processes	High Threat	High Threat	High Threat	Intermediate Threat	Low Threat	Unknown Threat	High Threat	Intermediate Threat	Unknown Threat	Intermediate Threat	Intermediate Threat
	Collections of Assets	Intermediate Threat	High Threat	High Threat	Intermediate Threat	Intermediate Threat	Intermediate Threat	Intermediate Threat	Intermediate Threat	Unknown Threat	Intermediate Threat	Unknown Threat

High Threat Intermediate Threat Low Threat Unknown Threat

Table 3.4 Asset-threat matrix, with pooled water and sediment quality.

Asset		Water and sediment quality	Habitat loss and fragmentation	Extraction and disturbance	Sea level rise	Temperature increase	UVB	Pests	Cumulative impacts
Habitat Groups									
Rocky reefs	Intertidal	Low Threat	Low Threat	Intermediate Threat	Intermediate Threat	Intermediate Threat	Intermediate Threat	Low Threat	Unknown Threat
	Subtidal	Intermediate Threat	Low Threat	Low Threat	Low Threat	Low Threat	Low Threat	Low Threat	Unknown Threat
Vegetated sediments	Mangroves	High Threat	High Threat	Intermediate Threat	Intermediate Threat	Low Threat	Unknown Threat	High Threat	High Threat
	Saltmarshes	Intermediate Threat	High Threat	High Threat	High Threat	Intermediate Threat	Unknown Threat	High Threat	High Threat
	Seagrass	High Threat	Intermediate Threat	Low Threat	Unknown Threat	Unknown Threat	Unknown Threat	Low Threat	Unknown Threat
Unvegetated sediments	Intertidal	High Threat	Intermediate Threat	High Threat	Intermediate Threat	Intermediate Threat	Unknown Threat	Intermediate Threat	High Threat
	Subtidal	High Threat	High Threat	High Threat	Low Threat	Intermediate Threat	Unknown Threat	Intermediate Threat	High Threat
Water column		High Threat	Low Threat	Low Threat	Low Threat	High Threat	Intermediate Threat	Low Threat	High Threat
Significant Fauna									
Iconic species	Shorebirds	High Threat	High Threat	High Threat	High Threat	Intermediate Threat	Unknown Threat	Low Threat	High Threat
	Seabirds	Intermediate Threat	Intermediate Threat	High Threat	Intermediate Threat	Low Threat	Unknown Threat	Low Threat	Intermediate Threat
	Marine Mammals	Low Threat	Low Threat	Low Threat	High Threat	Low Threat	Unknown Threat	Low Threat	Low Threat
Commercial/Recreational	Fish	High Threat	High Threat	High Threat	Low Threat	High Threat	Intermediate Threat	Intermediate Threat	Intermediate Threat
Emergent Features									
	Ecosystem Processes	High Threat	High Threat	Intermediate Threat	Intermediate Threat	High Threat	Unknown Threat	Intermediate Threat	High Threat

High Threat Intermediate Threat Low Threat Unknown Threat

4 Physical and chemical setting

Randall Lee

Photo courtesy Adam Pope.

Western Port is a large shallow embayment that is segmented into five basins by large islands and mudflats. Although generally well flushed by tides through the Western Entrance, wind forcing drives a prevailing clockwise circulation. The prevailing flow entrains catchment inflows and resuspended bay sediments, which results in poorer water quality (and higher residence times) in the east. The water quality can be significantly altered at much shorter time-scales than are currently monitored. This is likely to be the result of an interplay between the mudflats and incoming ocean waters.

While system-wide hydrodynamics have been adequately described, less is known of the finer-scale hydrodynamics (at a basin scale), so that we cannot properly understand connectivity through the system. Well-calibrated hydrodynamics is a fundamental first step in building a dynamic understanding of the bay.

Identifying the contribution of nutrients and sediments from the catchment, atmosphere and with-in bay processes is an important priority for management. We suggest that integrating the bay models with other well-accepted catchment and airshedding models will provide a more holistic picture of regional processes which will more accurately represent current and predict future bay conditions and responses.

Introduction

Western Port has an area of 680 km^2, of which two-fifths (270 km^2) is intertidal mudflats. The movement of water is dominated by semi-diurnal tides, and neither evaporation nor freshwater inputs are sufficient to affect flushing or to generate strong or persistent salinity wedges or estuarine circulation patterns. Water movement is principally driven by the tides, although winds over about 35 km/h can affect circulation (Hinwood and Jones 1979). The net movement of water in the bay is clockwise around French Island (see Figure 4.1), and most water enters and leaves Western Port through the Western Entrance.

The Western Entrance is the largest of the two entrances and the dominant connection with Bass Strait. Combined with the relatively small surface area, this means that the tidal wave at the Western Entrance is able to travel with very little attenuation throughout the whole of Western Port. The bay has a tidal range of between 2.3 and 3.1 metres, which comprises about 30% of the total volume of the bay at high tide.

The volume of water from Bass Strait entering the Western Arm each day is equivalent to the total volume of the bay (0.8 km^3). Water residence times range from about three months in the north and east to a few days in the south-west (Longmore 1997, Harris and Robinson 1979)

The waters of the bay are often categorised into five segments or basins (Figure 4.2), and their regional definitions are commonly referred to in the literature (Harris and Robinson 1979, Marsden *et al.* 1979, EPA 1996, Hancock *et al.* 2001). These divisions are predominantly based upon physical characteristics, particularly topography and flow, and are referred to throughout this chapter. The segments are:

- Lower North Arm, bounded by the Sandy Point constriction in the south and the Eagle Rock constriction in the north-east
- Upper North Arm, bounded by Eagle Rock constriction in the west and Stockyard Point constriction in the east
- Corinella, bounded by the Stockyard Point constriction in the north and the Settlement Point constriction in the south
- Rhyll, bounded by the Settlement Point constriction in the north and the Cowes confluence zone in the west
- Western Entrance Zone, bounded by Flinders Point on the Bass Strait entrance and the Sandy Point constriction in the north-east.

Further information on the physical setting of the bay has been previously documented by Shapiro (1975), Marsden and Mallett (1974), Marsden *et al.* (1979), Bird and Barson (1975); Bird (1993) and EPA (1996).

The Western Port catchment has an area of 3433 km^2 and is drained by 2232 km of rivers and creeks. Average annual rainfall ranges from 750 mm along the coast to 1200 mm in the northern highlands. Approximately 70% of the catchment is rural land, 20% is Crown land and 5% is urban (Counihan *et al.* 2003).

Figure 4.1 Water circulation in Western Port.
(Source: Hancock *et al.* 2001.)

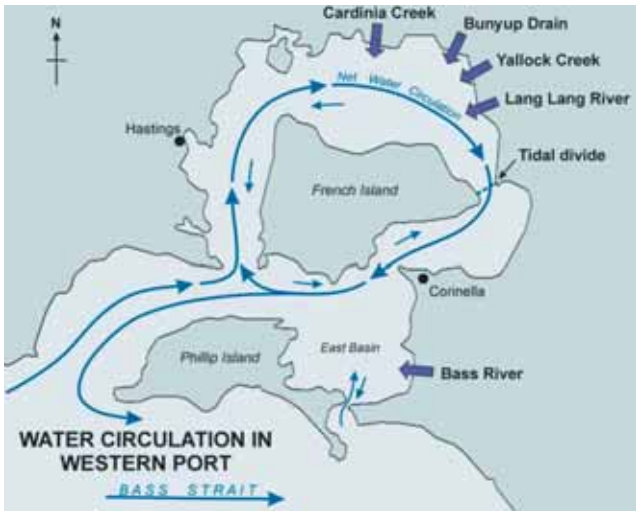
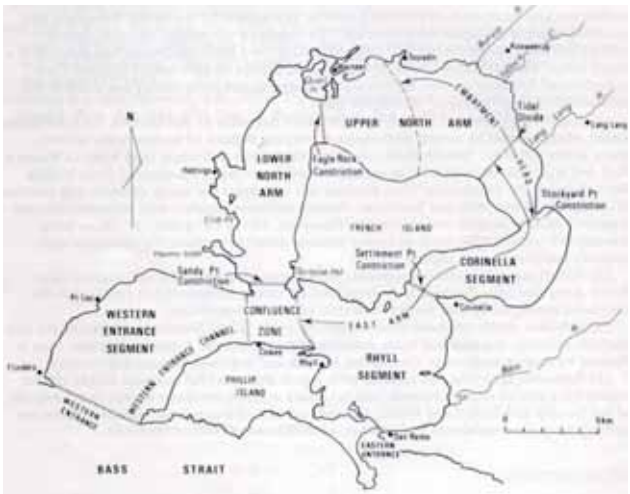


Figure 4.2 Common regional distinctions of Western Port.
(Source: Marsden *et al.* 1979.)



4 Physical and chemical setting

The major streams draining the catchment are the Bunyip, Bass and Lang Lang Rivers, which together contribute approximately 75% of the total freshwater flow into the bay (Counihan *et al.* 2003). They also deliver a significant sediment load originating from the erosion of gullies and stream banks in the catchment. Erosion of the shoreline in the Upper North Arm basin also contributes a significant sediment load (mostly fine sediment) to the bay (Counihan *et al.* 2003). Because of the net clockwise direction of water flow within the bay, much of the sediment delivered into the north-east of the bay is transported into the Corinella and Rhyll basins, where much of it is deposited (Hancock *et al.* 2001).

The geology of the catchment is mixed. The principal components are Quaternary sediments, sedimentaries from the Tertiary, Cretaceous and Paaleozoic, and much older volcanics (Marsden and Mallett 1975, Spencer-Jones *et al.* 1975).

A major lowland area, the original area of the Koo Wee Rup swamp, occupies a major portion of the catchment on the north-eastern shores of the bay, approximately 400 km^2. With settlement, the swamp was drained for agricultural use and, as a result, many of the waterways in the lower catchment are mostly modified as drains. Water quality is generally good in the upper catchments but tends to decline downstream of the Princes Highway. Clearing and draining of the catchment for agriculture has resulted in significant erosion of the waterways, particularly in the smaller, upper sections, greatly increasing the sediment run-off to the bays (EPA 1996). Figure 4.3 identifies the contribution of sediment from the catchment and associated major drainage channels.

A major issue for Western Port is the deposition of mud from upstream. Many studies have been done to identify the sources of sedimentation within the bay. Before European settlement there were no natural drainage systems from the Koo Wee Rup swamp, and in effect no direct water or sediment movement into Western Port (Butcher 1979). Today major mud sources are the Lang Lang River in the east and the eastern mud cliffs that are often eroded by wind and waves (Marsden *et al.* 1979). Increased mud deposition fills and alters the massive dynamic intertidal environment and was probably involved in historical sea-grass dieback (Chapter 10).

The unusually wide range of habitats shown in Figure 4.4, ranging from deep channels to seagrass beds, mangroves and salt marsh, support a large diversity of aquatic life. The Westernport Bay Environmental Study (Shapiro 1975) concluded that seagrass is the major source of primary production and plays an important role in Western Port's ecology. In addition to the fisheries, bird habitats and biodiversity that depend on the productive seagrass ecosystem, the seagrass maintains the water quality by stabilising the shallow and intertidal mud banks (EPA 1996).

Approximately 150 000 people live in the 25 towns around the Western Port coast. The areas in between are used for agricultural or rural residential purposes (WPRPCC 1992). As part of the Melbourne 2030 vision (DSE 2005) the catchments to the north and north-west are part of a growth corridor, and this is likely to result in increased loads of pollutants to the bay via rivers. The Port of Hastings, in the Northern Arm, is one of Victoria's deepwater ports (AGC Woodward-Clyde 1993). It was established as an oil port in the 1960s and now supports some industrial development. The Victorian government is considering increasing this development on the western side of the bay.

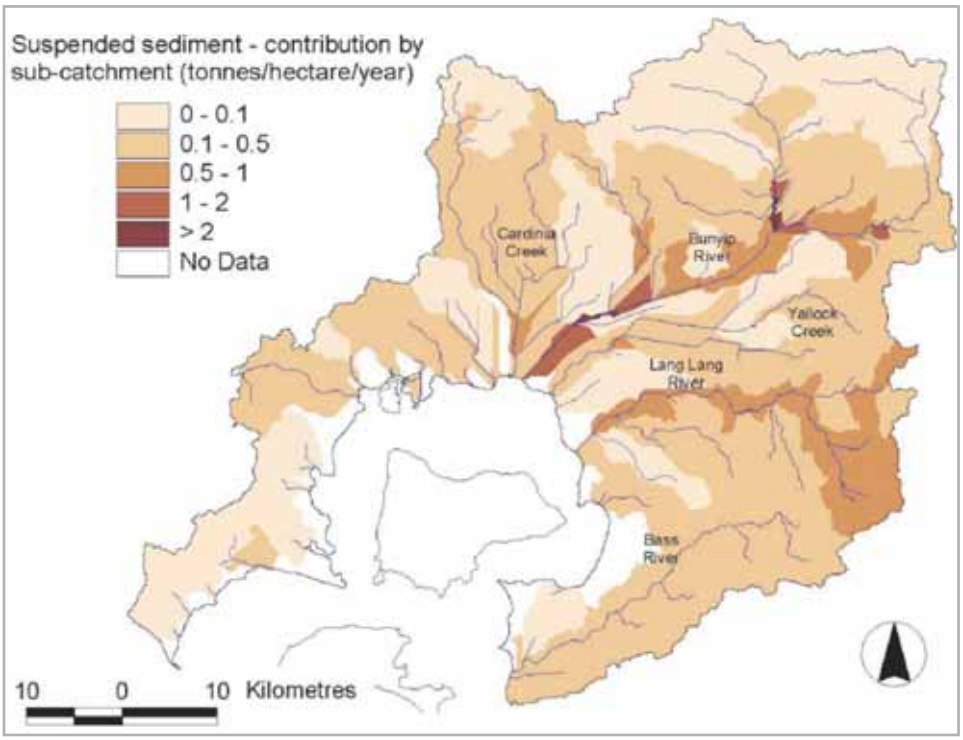


Figure 4.3 The contribution of run-off from various parts of the catchment to suspended sediments in Western Port.
(Source: Hughes *et al.* 2003.)

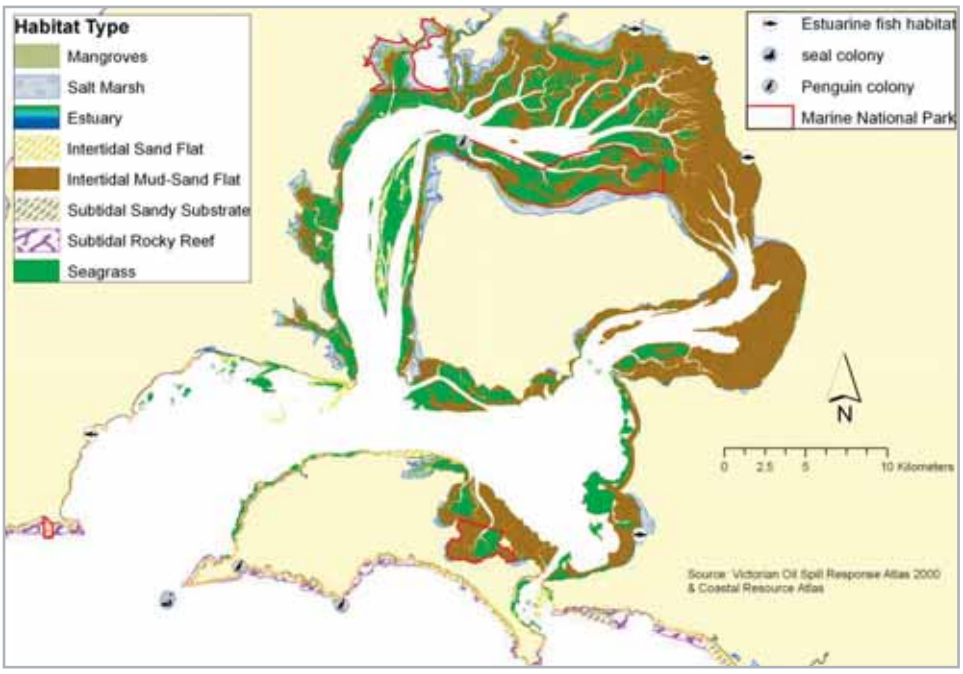


Figure 4.4 Marine habitat map of Western Port, showing the extensive intertidal flats that dominate the ecosystem.

Hydrodynamics

Overview

Western Port is an unusual and complex embayment of channelised mud flats, islands and multiple ocean entrances that generates complicated circulation patterns. While the bay has been the focus of past multi-disciplinary investigations (Harris *et al.* 1979, Hancock *et al.* 2001), it would be too costly to collect a comprehensive time series of sea levels or currents that could provide a full spatial and temporal coverage of flow conditions. However, since the 1970s hydrodynamic models of the flow patterns and resultant dispersal of catchment inflows have been developed. Two early models (Hinwood and O'Brien 1974, Hinwood 1979) focused on defining sediment transport with tide and wind driven flow. These two-dimensional models ran on relatively coarse grids, with an open ocean boundary set by local sea level measurements. They defined the broad dynamic dispersion of simulated pollutants responding to tides and winds (Figure 4.5). With two entrances defined by Phillip Island and modified by French Island, the tidal oscillations were shown to circulate around French Island, meeting at a tidal divide in the north-east. With the addition of prevailing winds, a resultant net eastward movement defined an overall clockwise circulation, with the majority of flow entering and exiting the system through the Western Arm.

More recent models are also two-dimensional and have a coarse scale, but are coupled with a catchment model (SedNET) that provided existing and projected loads of sediment to the bay (Hancock *et al.* 2001, Wallbrink *et al.* 2003, Hughes *et al.* 2003). While they have improved our knowledge of catchment sources and their relative input to the bay, there has been little improvement on the previous modelling in defining dispersion and settling processes in the bay.

An investigation of the Boags Rock sewage discharge on the southern coast of the Mornington Peninsula indicated that Western Port circulation patterns are influenced by a larger coastal region, including the open coast and Port Phillip Bay (Black and Hatton 1994). In particular, the results indicated a prevailing west–east pathway that could link exchanges from Port Phillip Bay with water in the Western Arm (Figure 4.6). It was demonstrated that this pathway could entrain the coastal discharge from Boags Rocks eastward, with diffuse levels entering Western Port.

Hinwood (1979) noted that finer model grids of about 100 metres were needed to advance specific studies beyond the coarser system-wide models. Other purpose-built models of Western Port have been developed in the 1990s and 2000s, to focus on specific issues (e.g. shipping movement) rather than encompassing a system-wide approach with appropriate resolution and complexity to represent all impacts and sources across the region and the bay. These have typically used off-the-shelf modelling suites employing two-dimensional dynamics with sediment transport modules.

More recently, the Better Bays and Waterways program (2004–2008) developed catchment and bay models to assess catchment loading coupled with in-bay dispersion processes. This project considered a range of catchment sources, and

Figure 4.5 Typical velocity vectors from the Hinwood (1979) model, where mean depth averaged peak tidal velocities are ~0.6m/s in the channels. (Source: Hinwood 1979.)

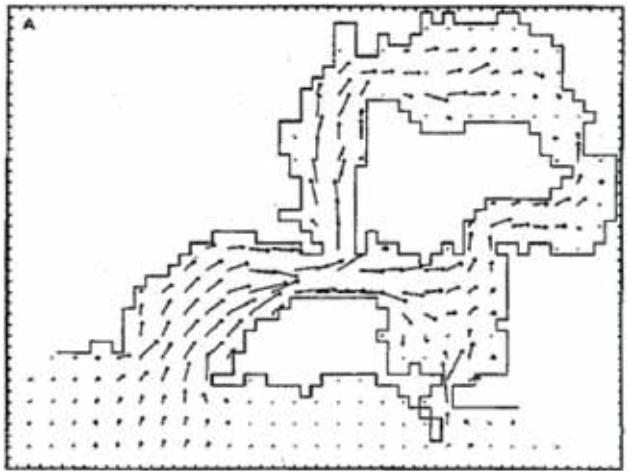


Figure 4.6 Two-dimensional nested model of localised Bass Strait hydrodynamics. (Source: Black and Hatton, 1994).

Figure 4.7. Bathymetry of Western Port used in numerical modelling for the Better Bays and Waterways program, 2004-08. The black contour line marks 0 m, or lowest astronomical tide used as chart datum. (Source: Harrison *et al.* 2011a)

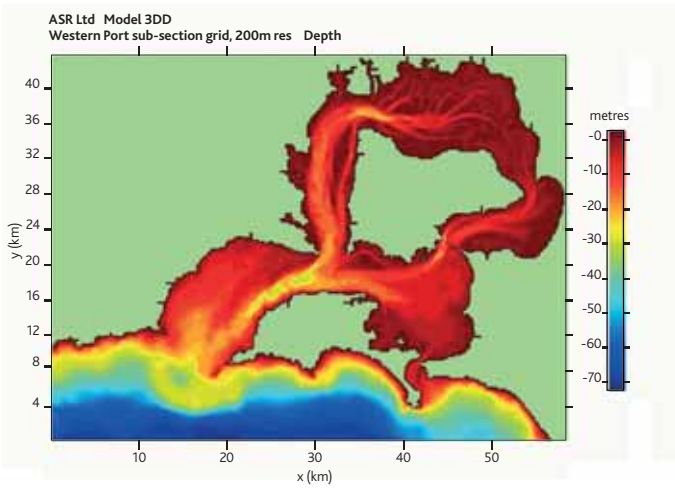


Figure 4.8. Aerial oblique infra-red photograph of the embayment head (looking west) from the area of the tidal divide. Darker red-brown areas have significant Caulerpa, and paler areas are dominated by *Zostera*. (Source: Harris *et al.* 1979.)

included existing and future scenarios to support ongoing management of the region by a variety of stakeholders. The PortsE2 catchment model used in this study was a Catchment Hydrology CRC product (Argent *et al.* 2009) developed specifically for the Port Phillip and Western Port regions. It provided a daily output of flow, nutrients, sediment, toxicants, salt and pathogens, based on functional units of land uses and rainfall–runoff relationships for 189 subcatchments in the region.

Because the waters of Western Port are rapidly mixed and well flushed, two-dimensional models are usually considered adequate for representing circulation. The continuing drought since 1998 has increased the salinity above that in Bass Strait (EPA 2011), and the practical salinity has been as high as 40 in the north-eastern part of the bay. To accurately represent these conditions a model must include atmospheric heat exchange and resultant evaporation to create higher salinities, which requires a stratified three-dimensional modelling approach.

The receiving bay model, based on the 3DD modelling software, was developed by Harrison *et al.* (2011a). Unlike most of the earlier models, these were fully stratified three-dimensional models that accounted for ocean–atmosphere exchange. The model integrated output from the PortsE2 catchment modelling to assess the fate of catchment loading in the bay. The model was configured on 200, 400 and 800 m grids with six vertical layers, although most longer-term runs used a 400m grid that was adequate for between-basin exchange patterns, but limited for within-basin detail. These models were integrated to assess a range of catchment and within-bay scenarios (Harrison *et al.* 2011a). This integrated approach is referred hereafter as the Receiving Water Quality Models (RWQM).

The following section provides details of the model bathymetry and validation that would be required to instil confidence in the outputs of any modelling study. This is done to illustrate some of the information shortfalls that currently limit modelling investigations and hence hydrodynamic process understandings in Western Port. The results of this recent modelling effort are also provided because this broadens our knowledge of system dynamics gained from previous studies and highlights ongoing issues to address.

Model bathymetry

The Hydrographic Service of the Royal Australian Navy released a chart of Western Port in 1995 (AUS 150). This was a compilation of earlier charts (AUS 151), British Admiralty charts (e.g. BA 149Y) shipping channel surveys and other local sources (EPA 1996). The bathymetry used in the RWQM (Harrison *et al.* 2011a) is based upon these naval charts, and knowledge from more recent habitat and channel surveys (D. Ball, Department of Primary Industries, pers. comm.). Figure 4.7 depicts the wide variety of physical environments in the bay, from deep and broad entrance channels to narrow braided channels in the inner reaches. From the bathymetry alone it is evident that the bay's dynamics should vary spatially, as found during the Westernport Study (Shapiro 1975). The braided channels in the inner reaches and the intervening banks (Figure 4.8) appear to strongly influence the position and growth of seagrasses. This may be expected because of the corresponding variations in depths and current velocities between the banks and channels. The complex fingering of the braided channels is maintained in the 3DD model bathymetry (Figure 4.9).

Recent bathymetry surveys have been undertaken in Western Port as part of the Victorian Government's Future Coasts program (DSE 2010). As part of a statewide comprehensive coastal survey utilising laser airborne mapping (LIDAR), digital elevation models have been created for coastal topography and bathymetry to about 20 metres. The effectiveness of this technique to resolve detailed bathymetry is constrained by the clarity of the water. In Western Port it was effective only in the clearer waters of the Western Entrance and Rhyll basins. The remainder of the channels were subsequently mapped using multi-beam radar. These surveys are now being melded to create an updated and highly detailed bathymetric picture of the bay (Figure 4.10). The results will be invaluable when undertaking more detailed modelling analyses of the bay to assess within-basin dynamics.

Figure 4.9 Current velocities in the embayment head from the 3DD model. (Source: Harrison *et al.* 2011a.)

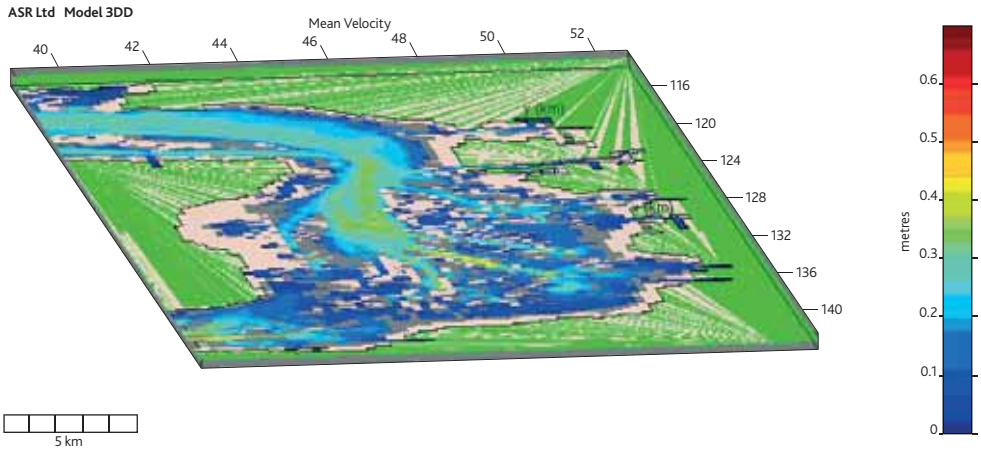
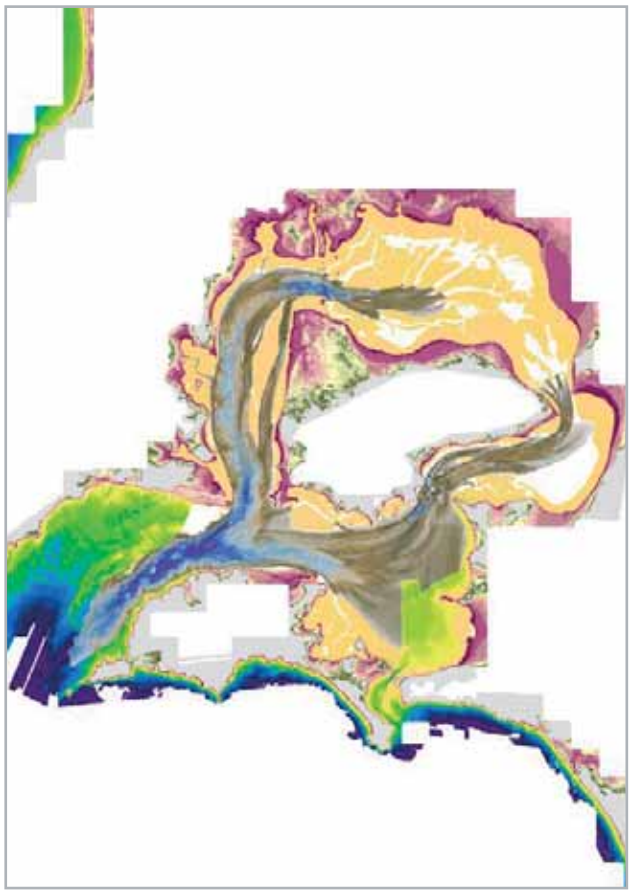


Figure 4.10 Recent LIDAR and multi-beam surveys from DSE's Future Coasts Program. (Source: DSE Future Coasts.)



Validating hydrodynamics

While validation and calibration of the earlier two-dimensional hydrodynamic models (Hinwood 1979) were undertaken at the time, limited data are available for subsequent modelling studies. Limited calibration within Western Port was undertaken for the RWQM, utilising photos, plots, and information in Sternberg (1979, Hancock *et al.* (2001), Wallbrink *et al.* (2003), Hughes *et al.* (2003) to reproduce the hydrodynamic environment of the bay.

Circulation patterns

Sternberg (1979) presented measurements of currents taken from nine stations within Western Port, focusing on the western side of the bay (Table 4.1, Figure 4.11). The 'residual circulation' — the net movement of water through all model cells when vectors are averaged in each cell over a selected long time period (Figure 4.12) — can be used to infer the likely residual path for dissolved pollutants, muds or sandy seabed sediments.

When the pattern from the RWQM is compared to several former predictions of net circulation made by Shapiro (1975) it is clear that the residual circulation is more complex than previously inferred (Figure 4.13). Some of the strongest patterns from the model (indicated by the superimposed white arrows in Figure 4.12) show strong similarities with the earlier predictions in Figure 4.13, although none of the earlier predictions are full agreement with each other. Good confirmation of the model results is provided by the strongest similarity with the seabed drifter patterns which integrate the currents over a long time period (1974–1975).

Table 4.1 Comparison of the measurements of bottom currents (100 cm from bottom) in Western Port presented by Sternberg (1979) with the 3DD two-dimensional depth-averaged model output. The measurements have been corrected to depth-averaged assuming a logarithmic velocity profile in 10 m depth over a bed-formed seabed with roughness length zo = 5 mm.
(Source: Harrison *et al.* 2011a.)

	Measured			Modelled	
Station Number	Range (cm/s)	Mean (cm/s)	Mean (log corrected)	Range (cm/s)	Mean (cm/s)
1	20–40	30	37	20–80	50
2	n/a	n/a	–	24–88	56
3	20–50	40	50	23–73	48
4	40–65	52.5	65	25–90	57.5
5	35–50	42.5	53	21–80	50.5
6	25–55	40	50	25–78	51.5
7	28–68	48	60	50–120	85
8	15–60	37.5	47	20–110	65
9	15–40	27.5	34	23–56	39.5

Figure 4.11 Stations used for the comparison of current velocities between the 3DD model and bottom-current measurements.
(Adapted from Sternberg, 1979.)

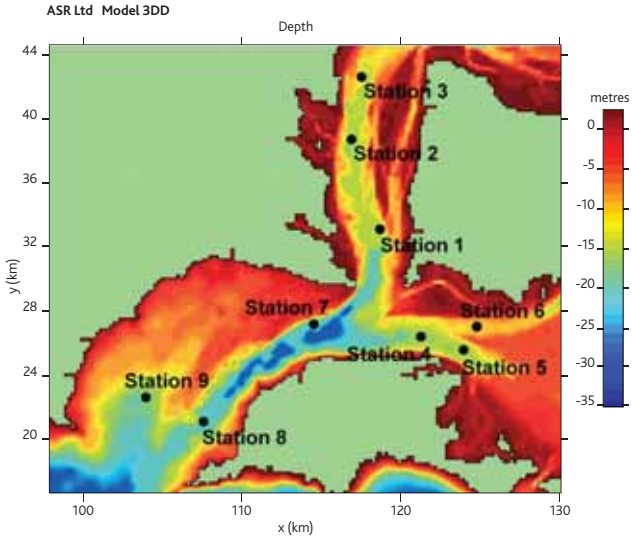


Figure 4.12 Residual velocity vectors and depth over a 15-day spring–neap tidal cycle, illustrating the dominant tidal driven currents of Western Port. (Note that the simulation does not include wind). White arrows indicate general trends in Western Port currents. (Source: Harrison *et al.* 2011a.)

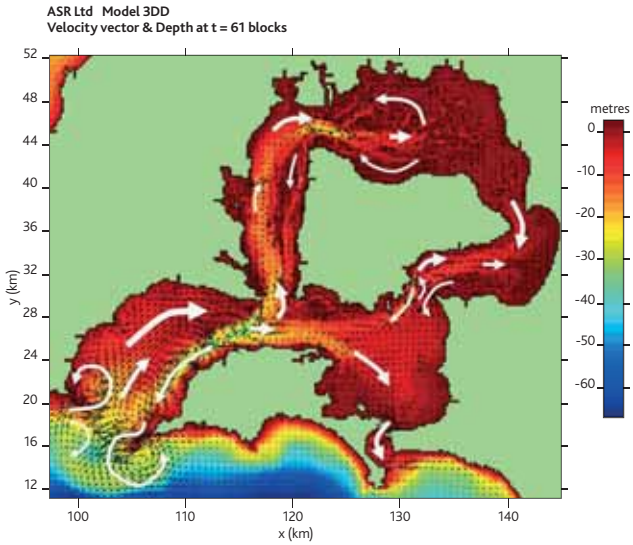
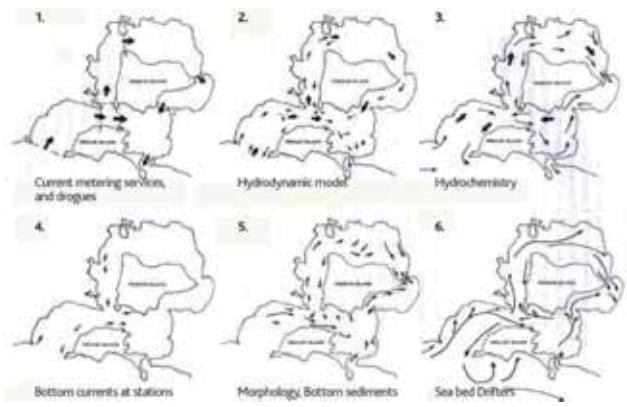


Figure 4.13 Independently developed net circulation patterns.
(Source: Harris *et al.* 1979.)
1. Hinwood and Jones (1979), 2. Hinwood (1979), 3. Harris and Robinson (1979), 4. Sternberg (1979), 5. Marsden et al. (1979), 6. Marsden (1979).



Sea level

Good calibration of water levels in a Western Port model is the key to accurately representing volume exchanges (i.e. flushing and dispersion) with Bass Strait. Figure 4.14 shows the example of the RWQM predictions against observed water levels for two months at Stony Point, at the entrance to the Upper North Arm. Figure 4.15 shows the linear regression of the field and model data. It can be seen from the figures that the model simulates the propagation of Bass Strait tides into Western Point for both the spring–neap modulation of tide range and the phasing of the tides.

Salinity and temperature

Because Western Port has large expanses of shallow mudflats and short flushing times (compared to Port Phillip Bay), its response to evaporation and fresh inflows is expected to be quicker and more severe than deeper bay systems. Detailed information about salinity and temperature dynamics is needed throughout Western Port from fixed moorings in order to effectively calibrate bay responses to heating and catchment inflows.

There is no available time series of (continuous) mooring data to compare salinity and temperature measurements to the RWQM output. The best available comparison is to use the monthly measured data at three locations in Western Port from the EPA's fixed site program to represent seasonal cycles of salinity and temperature (EPA 2011). To improve statistics and reliability, an average annual cycle (derived from over 20 years of more-or-less monthly sampling, from 1985 to 2009) was used to compare with the 2004 model output in Figure 4.16.

Salinity peaks in March and is lowest in September. The model data extracted from the Upper North Arm track the March peak and subsequent freshening with the onset of winter, and falls within one standard deviation of the observed monthly averages. Note that with these comparisons there is a slight over-estimation of salinity by the model (model = observed * 1.0047154; for coastal waters) as it uses absolute salinity (SA) measured in g/kg (or parts per thousand) based on the recent definition by UNESCO(2010), instead of the historically measured practical salinity (SP) that reports with a unitless practical salinity scale (UNECO, 1983)[3].

Figure 4.14 Time series comparison of the tides measured at Stony Point with the predictions of the model. Plot shows data for Julian days 240 to 360 (2004). (Source: Harrison *et al.* 2011a.)

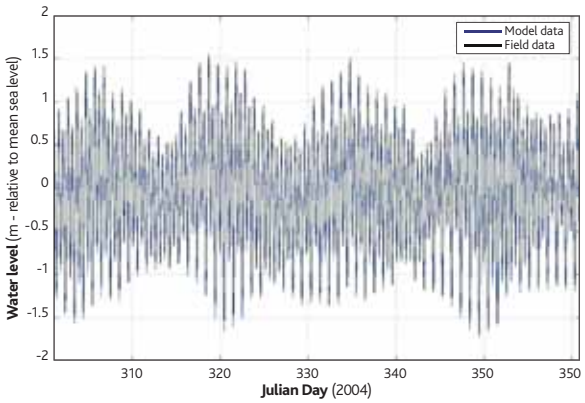


Figure 4.15 Regression comparison of the tides measured at Stony Point with the predictions of the model. Linear regression values: slope = 1.035, intercept = 0.003, r2 = 0.9022. (Source: Harrison *et al.* 2011a.)

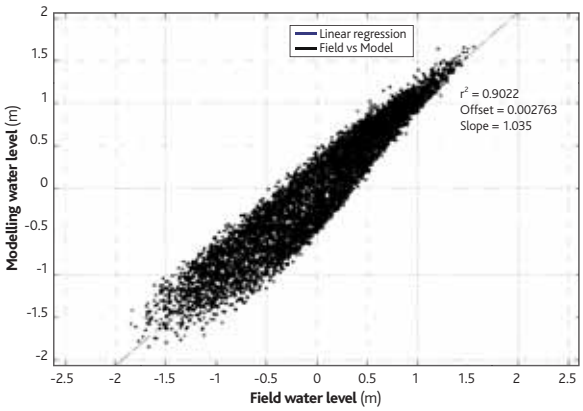


Figure 4.16 Average monthly salinity (practical salinity) for the period January to December (1985–2009) at Barrallier Island fixed site, compared to RWQM simulations of absolute salinity (g/kg) for 2004 in Upper North Arm. Dashed lines indicate one standard deviation either side of the mean. (Source: EPA unpubl. data.)

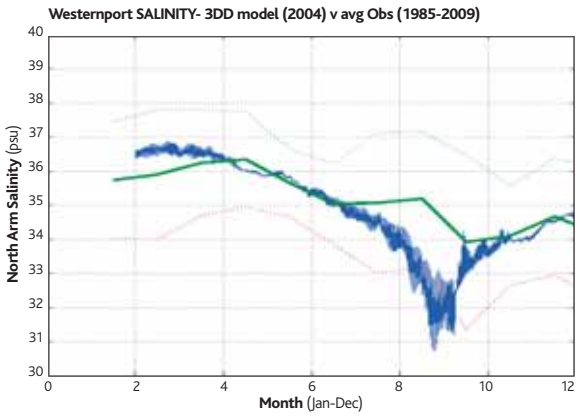
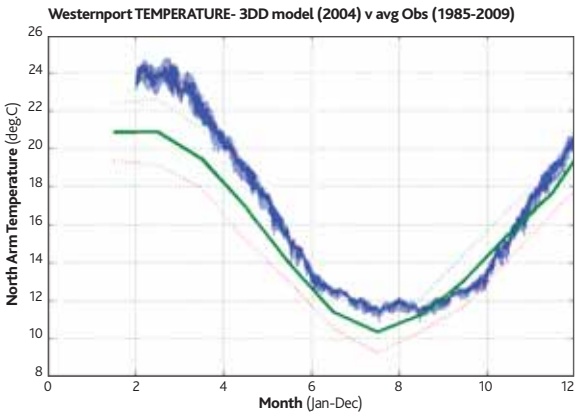


Figure 4.17 Average monthly temperature (January to December), 1985–2009, at Barrallier Island fixed site, compared to RWQM simulations for 2004 in Upper North Arm. Dashed lines indicate one standard deviation either side of the mean. (Source: EPA unpubl. data.)



3 **Practical Salinity S_P**
The observed salinity measurements have been historically reported as the practical salinity (S_P), which is from the unitless practical salinity scale (PSS-78). (UNESCO 1983)

Absolute Salinity S_A
Numerical models calculate an absolute salinity "S_A" as g/kg (or parts per thousand) in order to calculate density and other seawater properties. The thermodynamic properties of seawater, such as density and enthalpy, are now correctly expressed as functions of Absolute Salinity rather than being functions of the conductivity of seawater. The differences is based upon as a comparison to a reference salinity S_R where; $S_R = 35.16504/35 * S_P$; and $S_A = S_R + \delta S_A$ For coastal waters $\delta S_A = 0$ (UNESCO, 2010).

A similar result is apparent for temperature. The model tracks the observed seasonal cycle (Figure 4.17) generally within one standard deviation of the monthly averages during the 1985–2009 period.

It is likely that limited calibration available for the freshwater volumes coming from the PortsE2 catchment model contributed to flows appearing larger than expected for the 2004 drought year. As part of ongoing action from the Better Bays and Waterways Program (2009), a PortsE2 re-analysis was undertaken with improved calibrations from an improved measurement network and land use distinctions (Stewart *et al.* 2010). This information also found that the earlier work (BMT WBM 2007) had overestimated flows by about 15%, and also overestimated the nutrient and sediment (TP, TSS) loads to Western Port. The PortsE2 re-analysis data are now being incorporated into the RWQM for Western Port.

Given that the existing hydrodynamic data for Western Port has been limited, more robust verification of the RWQM was limited to the two-dimensional hydrodynamics. Additional time series data would be needed for temperature and salinity to calibrate a stratified three-dimensional hydrodynamic model that is able to represent fresh catchment impacts, and desiccating evaporation terms at appropriate dynamic scales. Some of this has recently been instigated by EPA Victoria in a two-year deployment of moorings in 2011 to characterise the water in the northern and eastern basins. This long-tem program is being augmented with additional short-term (two-monthly) deployments of current meters and water-quality loggers at key choke points in the system to ensure a comprehensive coverage of the bay for model calibration.

Hydrodynamic simulations

RWQM results are shown as average conditions for the first six months of 2004 for residual flow, temperature, and salinity in Figure 4.18. Although models were run for the entire 2004–05 period, poor salinity calibration data to assess accurate evaporation flux (especially for mudflats and areas of high turbidity) limited further model development to account for evaporative losses, and hence confidence in the three-dimensional model stability. Of the results shown, residual currents (or net flows) map established pathways, salinity indicates a well flushed system and temperature peaking in the Northern and Rhyll basins suggests memory of water drained from the mud flats of the adjoining Upper North and Corinella basins.

Figure 4.19 indicates significant salinity dynamics (about the mean in Figure 4.18 during this period of low flow (salinity peak in March) and during wet weather (June) periods. This highlights the sensitivity of the Upper North Arm and Corinella segment to evaporation and inflows that dominates the majority of Western Port. The wet weather data agree well with a comprehensive survey by Harris and Robinson (1979) in which wet weather conditions were mapped from 72 sampling sites (Figure 4.19).

Figure 4.18 Residual velocity mean temperature and absolute salinity (g/kg) in Western Port for January–June 2004. (Source: Harrison *et al.* 2011a.)

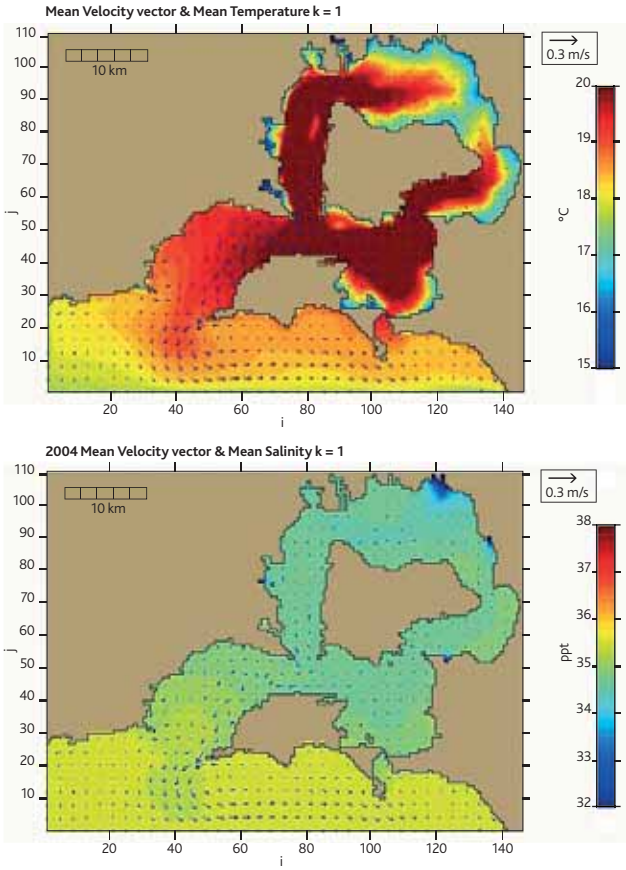
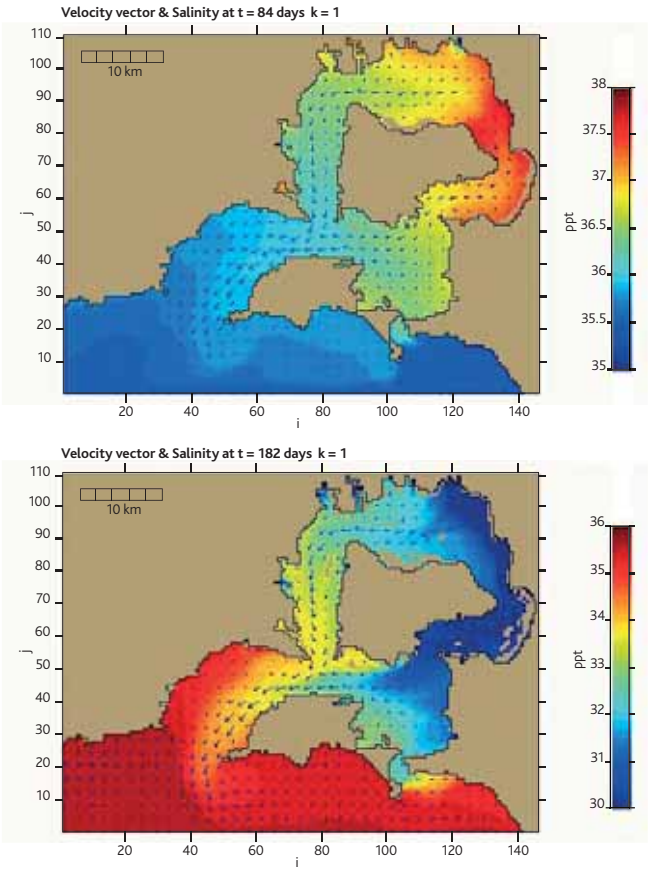


Figure 4.19 Model runs for 2004–2005 showing changes in absolute salinity (g/kg) associated with baseline (top) and wet (bottom) weather conditions. (Source: Harrison *et al.* 2011a.)



4 Physical and chemical setting

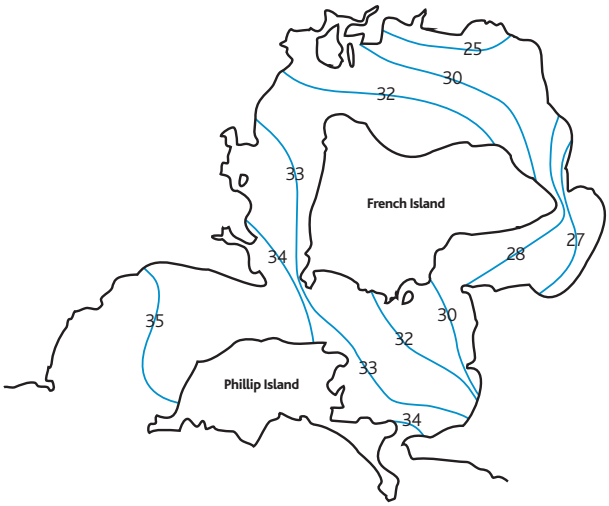
As well as having significantly different seasonal states, the bay exhibits strong shifts over shorter periods associated with weather forcing. Figure 4.21 shows model simulations over a six-day period and references thermal structure from an offshore mooring (in 24 m of water). In addition to diurnal heating, there are significant changes in bay temperatures that tie in with wind forcing and offshore temperature stratification. In this relatively short period there were upwelling and downwelling winds on the open coast that were picked up in the temperature structure.

Data from an opportunistic mooring in Yaringa channel in the Upper North Arm (Figure 4.22) highlight the effect of the shallow mud flats on adjacent deeper bay waters. Low tides at about 9 pm over a number of days corresponded with peak temperature, salinity and chlorophyll-a fluorescence. The data also show large diurnal variations that are synchronised with tides (e.g. chl-a from 2–5 µg/L).

The recent modelling and opportunistic measurements have highlighted the sensitivity of this system at a wide range of scales, from semi-diurnal to seasonal. Strong variations are evident even at semi-diurnal scales. This can be reflected in the water quality that drains off the mud flats and into the adjoining basins, and links to open ocean dynamics.

The application of finer resolution models to advance modelling capability in Western Port hinges on capturing these shorter-scale dynamics to investigate within-basin hydrodynamics to a suitable level of detail. It is anticipated that with a comprehensive and consistent time series of field measurements in the bay coupled with accurate bay bathymetry, the necessary validations are achievable.

Figure 4.20 Salinity (practical salinity) contours from a detailed survey of 72 sites during wet conditions in 1975.
(Source: Longmore 1997.)



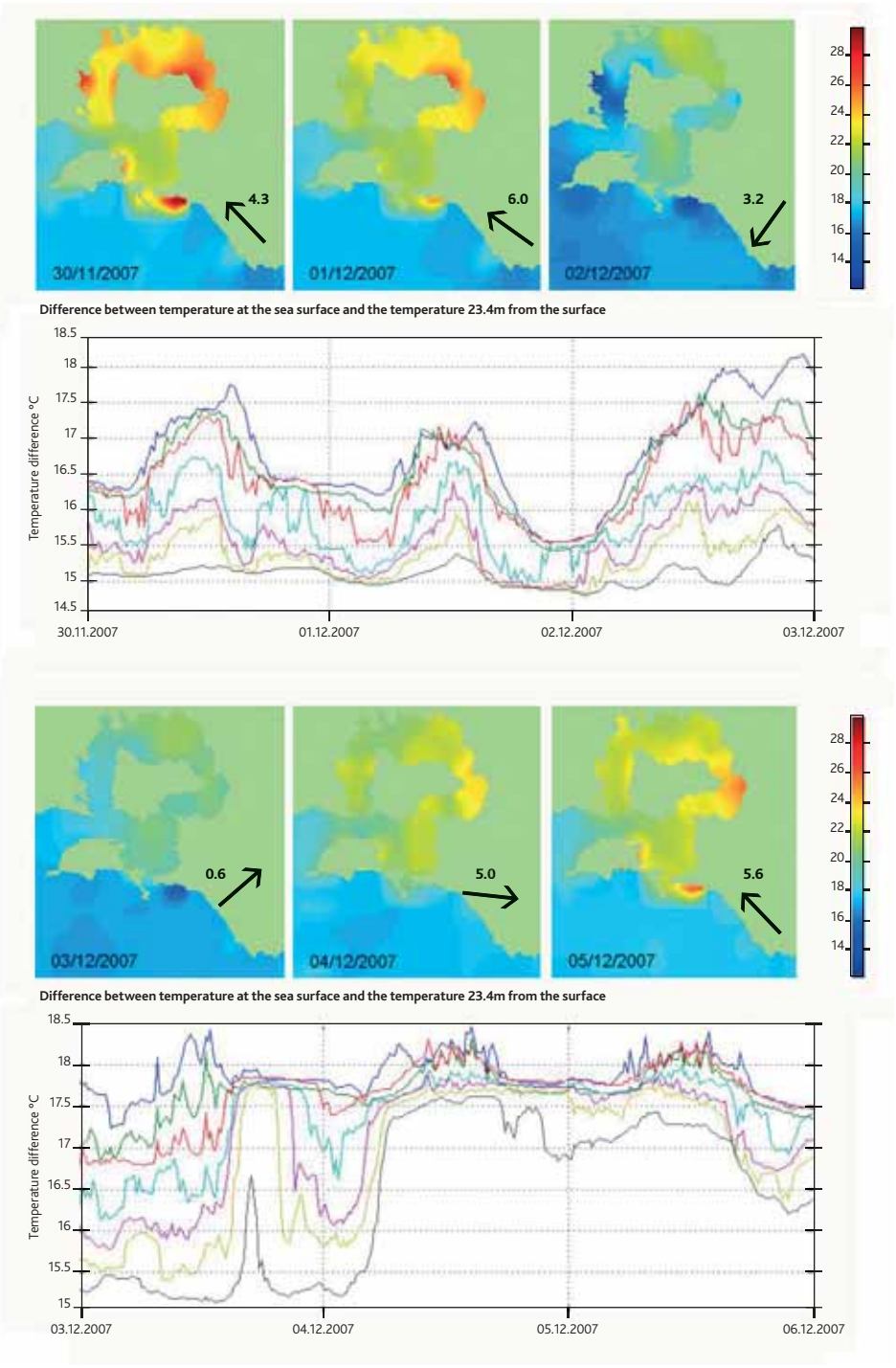


Figure 4.21 Modelled Western Port temperature dynamics and associated wind forcing over six days. The temperature difference plot (offshore in 24 m) indicates upwelling and downwelling dynamics associated with the shifting winds. (Source: ASR unpubl. data.)

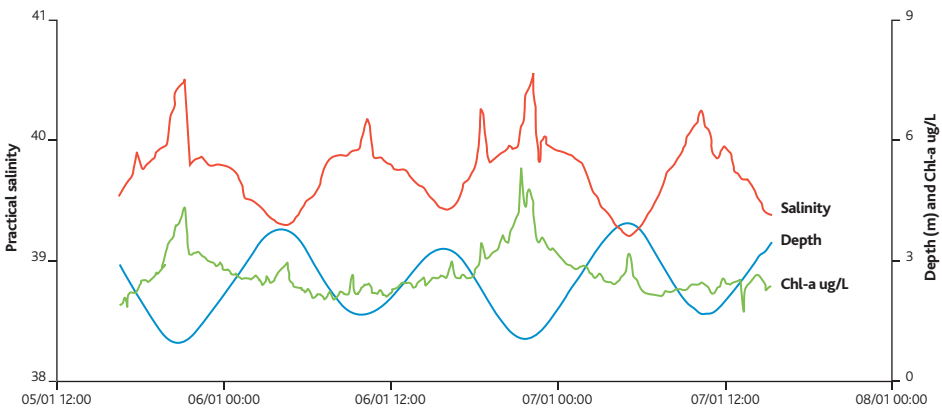


Figure 4.22 Moored water quality measurements over three days in Upper North Arm – Yaringa Channel. (Source: EPA unpubl. data.)

Water quality

Key issues

Despite the key issues of increasing pressure on Western Port and the potential impact of climate change, there has been little work on enhancing our understanding of the dynamic processes associated with in-bay water quality since the Westernport Bay Environmental Study in the 1970s (see chapter 14 for a detailed review). These key issues are most likely to impact the bay as a result of changes to sediment loading and and dispersion patterns through the bay.

Sediment mixing occurs to at least 18 cm in the Corinella segment, and to 12–24 cm in the Upper North Arm (Wallbrink *et al.* 2003). The distribution of sediment within the bay was mapped extensively in 1975 (Marsden *et al.* 1979) and also in 2001 (Hancock *et al.* 2001). Hancock *et al.* (2001) suggest there has been a significant redistribution of muds in the intervening period (Figure 4.23), and in particular a significant net loss from the Upper North Arm.

The resuspension and transport processes occurring in Western Port also show that sediment delivered to the Upper North Arm has the ability to affect seagrass habitat in the eastern and southern regions by the generation of persistent turbidity, and by the rapid accumulation of fine sediment in the Corinella and Rhyll segments (Figures 4.24, 4.25). A new study of the past seagrass losses at Western Port has also highlighted that the height and drainage patterns of intertidal mudflats are critical to the survival of seagrass. Seagrass in turn is critical to the stability of these mudflats (Parry, 2007). Turbidity generated by resuspension may have been exacerbated by the recent decline of the seagrass beds, and may continue to retard their recolonisation. Wallbrink *et al.* (2003) suggested that, if this were the case, reestablishing Upper North Arm seagrass beds would be a priority for improving seagrass survival elsewhere. Accurately assessing system dynamics may prove invaluable in guiding any targeted attempt at seagrass restablishment (see also Chapter 10).

Because the shallow waters of Western Port are dominated by the larger intertidal flats, and these sediments are unstable, sediment quality is linked to water quality. Rees *et al.* (1998), studied sediments in the bay for attainment and trends in toxicant concentration for the first time since the major studies undertaken in the 1970s. They found that most metals, organics and organometallics were below ANZECC (2000) guidelines for sediment quality. Arsenic distribution indicated a number of hotspots that were close to or exceeded the ANZECC low trigger level (20 µg/g). These were typically associated with the clay–silt fractions (low flow areas shown in Figure 4.26) and in streams draining the Koo Wee Rup swamp in the north. As is the case with Port Phillip Bay, it is probable that most of this arsenic derives from natural geological sources (Fabris and Longmore, 2005).

Figure 4.23 Suggested clockwise sediment redistribution in Western Port. (Source: Hancock *et al.* 2001.)

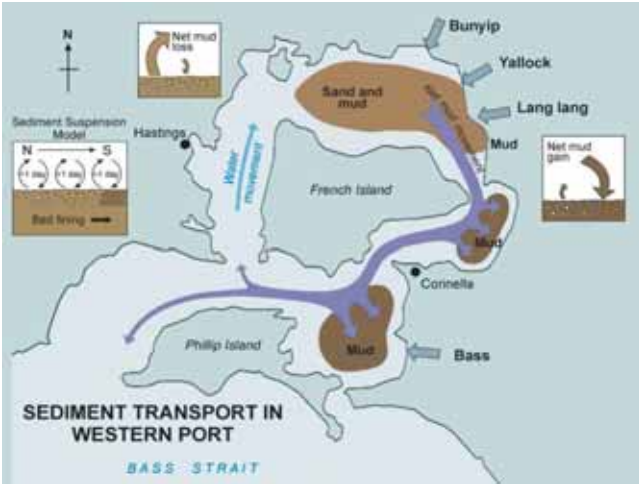
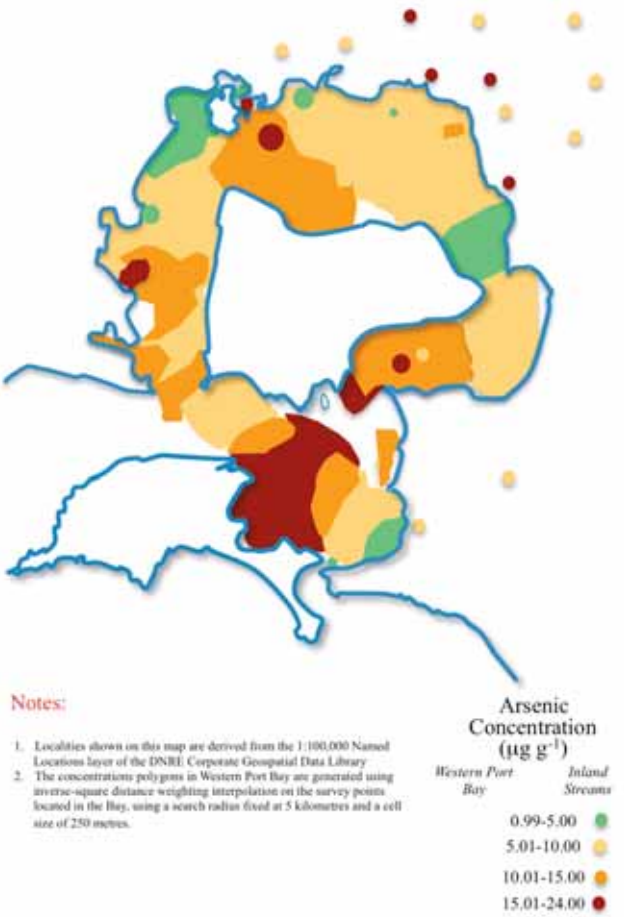


Figure 4.26 Total arsenic concentration (µg/g dry weight) in sediments from 44 sites in Western Port. Areas in red are near or above the trigger level of 20 µg/g in the ANZECC Interim Sediment Quality Guidelines. (Redrawn from Rees *et al.* 1998.)



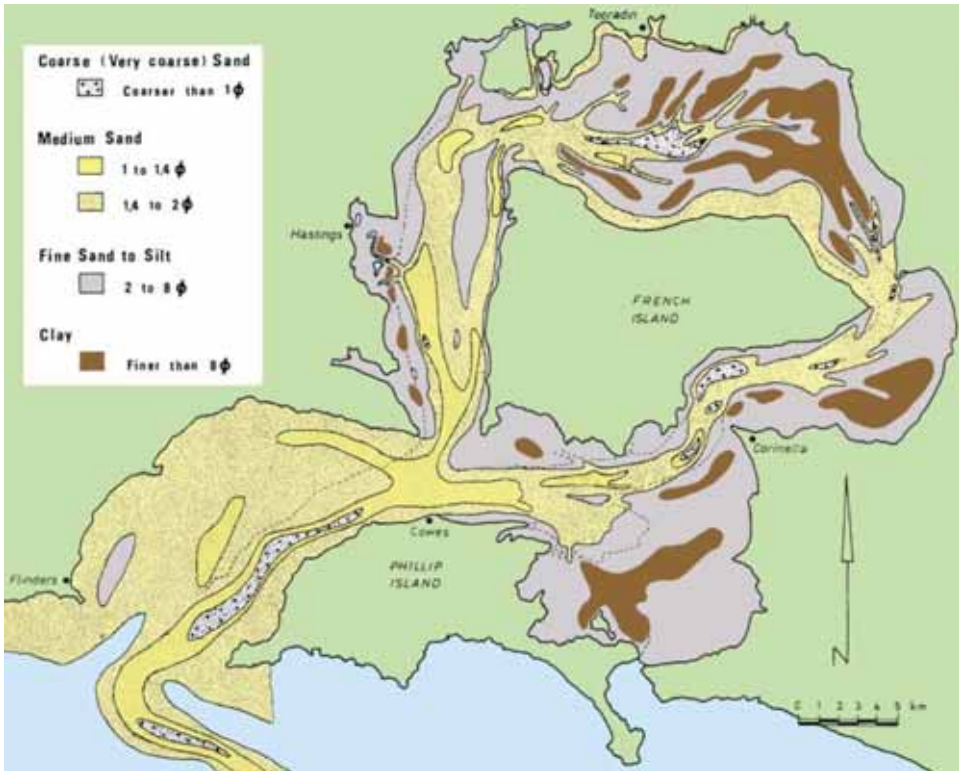


Figure 4.24 Sediment map from surveys in 1970s showing area where fines below 2 phi (clay to fine sand) were likely to settle.
(Source: Marsden *et al.* 1979.)

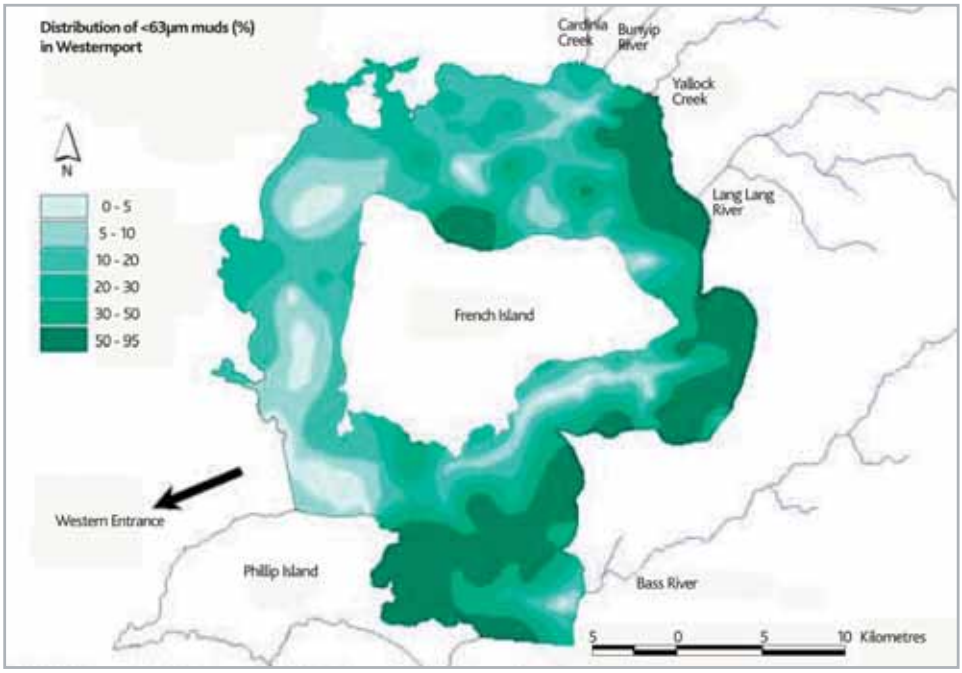


Figure 4.25 Particle size distribution for muds under 63 µm in diameter from a sediment survey in 2000.
(Source: Hancock *et al.* 2001.)

Condition assessment

To protect water quality in Western Port, the State Environment Protection Policy (SEPP) W-28 (Waters of Western Port Bay and Catchment) was declared in 1979. Although Harris and Robinson (1979) had divided the bay into six segments on the basis of physico-chemical and geographic information, the SEPP (from 1979) defined only one segment for the whole of Western Port.

After 18 years the SEPP segments were reviewed by Longmore (1997), who concluded that Western Port should be divided into at least two segments for water quality monitoring. This formed the basis of the two recognised water quality segments in the current Waters of Victoria SEPP Schedule F8 (Victorian Government 2001) for Western Port (Figure 4.27). Both segments are currently sampled by the EPA Marine Monitoring Network.

The SEPP Schedule F8 (Victorian Government 2001) provides for the protection of a number of beneficial uses of the bay:

1. aquatic ecosystems
2. passage of native fish or other biota
3. primary contact recreation
4. secondary contact recreation
5. aesthetic enjoyment
6. aquaculture
7. industrial and commercial water use
8. navigation and shipping
9. consumption of fish, crustacea and molluscs for recreational or commercial purposes.

Environmental quality objectives are numerical values for particular indicators of the condition of Western Port. The key indicators for the protection of the beneficial uses are transparency, turbidity, suspended solids, dissolved inorganic nitrogen, dissolved inorganic phosphorus, chlorophyll-*a*, *E. coli*, total arsenic, total cadmium, total copper, total lead, total mercury, total nickel and total zinc (Table 4.2).

Table 4.2 SEPP F8 Water quality indicators for Western Port.

		Segment Objectives	
Indicator and unit	**Parameter**	**Entrance and North Arm**	**East Arm**
Transparency (Secchi disc), metres	annual median	> 2.4	> 0.7
	annual 25th percentile	> 1.4	> 0.4
Suspended solids, mg/L	annual median	< 9	< 30
	annual 75th percentile	< 19	< 90
Total phosphorus, mg/L*	annual 75th percentile	< 0.03	< 0.03
Total nitrogen, mg/L*	annual 75th percentile	< 0.3	< 0.3
Dissolved inorganic nitrogen, µg/L	annual median	< 7	< 20
	annual 75th percentile	< 15	< 43
Dissolved inorganic phosphorus, µg/L	annual median	< 6	< 7
	annual 75th percentile	< 8	< 10
Chlorophyll-a, µg/L	annual median	< 1.6	< 2.5
	annual 75th percentile	< 2.1	< 5.0
Total arsenic, µg/L	maximum	< 3.0	< 5.0
Total cadmium, µg/L	maximum	< 0.05	< 0.05
Total copper, µg/L	maximum	< 1.0	< 2.0
Total lead, µg/L	maximum	< 1.0	< 2.0
Total mercury, µg/L	maximum	< 0.005	< 0.01
Total nickel, µg/L	maximum	< 1.0	< 3.0
Total zinc, µg/L	maximum	< 2.0	< 5.0
Dissolved oxygen, % of saturation	minimum	> 90	> 90
Temperature, °C	variation	< N + 1	< N + 1
Salinity (practical salinity)	variation	< N ± 1	< N ± 1
pH	range	7.5–8.5	7.5–8.5
Turbidity, NTU	annual median		
	annual 75th percentile		<10

* From SEPP Water of Victoria, 2003.

Figure 4.27 Locations of long-term monitoring sites and existing policy segments. The aquaculture zone and EPA licensed discharges (shown as industrial drains) are also represented. (Source: EPA 2011.)

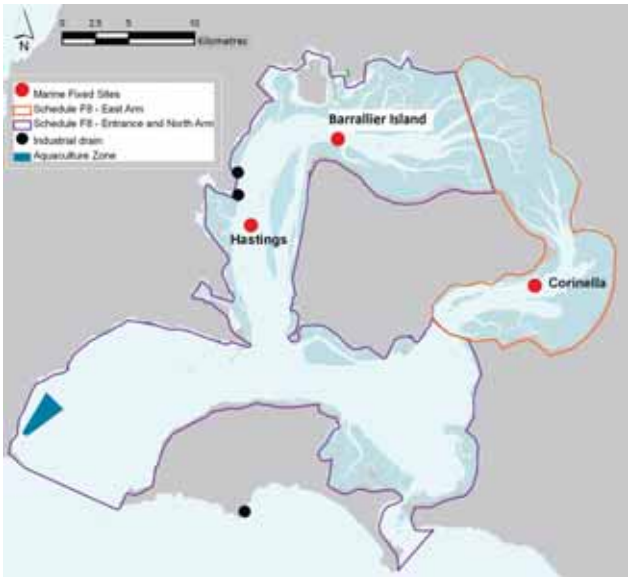
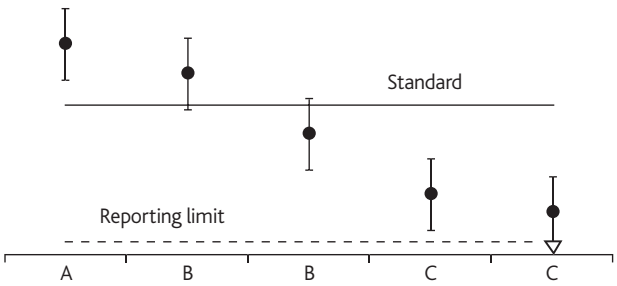


Figure 4.28 Interpretation of measurement interval (result ± measurement of uncertainty) against environmental standards. (Source: Goudey 2007.)



Based upon this framework, there have been a series of assessments undertaken for Western Port (Longmore 1997; EPA 2002, 2006), and one covering the period 1990–2009 (EPA 2011). It is from this most recent analysis that the majority of the following information is drawn.

Results from sampling undertaken at the three sites in Western Port are discussed in the following section in relation to the attainment of water quality objectives and long-term trends with respect to SEPP Schedule F8 objectives.

The monitoring results were assessed using a method based on an approach in Goudey (2007), to infer whether an objective was not met, using 95% confidence intervals (Figure 4.28). Table 4.3 summarises the results of this assessment for each sampling site, along with SEPP objectives and monitoring results for 2009.

In cases A and C, the measurement interval is sitting entirely above (case A) or below (cases C) the environmental objective; that is, the environmental standard was not met in A, while C complied in both cases. Complexity arises in case B, where the measurement interval is overlapping the environmental objective. In those cases the inference is equivocal, and more data are needed (highlighted in orange in Table 4.3). Weak inference also occurs when the standard is below the laboratory limit of detection (highlighted in blue in Table 4.3).

As previously described, differences in the prevailing circulation and sedimentation patterns result in consistently non-compliant or potentially non-compliant readings at the eastern Corinella site (Table 4.3). Only 7 out of 20 indicators (35%) complied with their objective for 2009. Secchi disc depth, suspended solids, total phosphorus, total nitrogen, dissolved inorganic nitrogen, chlorophyll-a, mercury and nickel are potentially non-compliant, as their 95% confidence interval overlaps with their respective objective. Dissolved oxygen and zinc are non-compliant for the year (Table 4.3).

The Hastings and Barrallier Island sites comply with 11 out of 20 objectives each (55%). Secchi disc depth, dissolved inorganic nitrogen, cadmium and nickel are potentially above their objective. Dissolved oxygen and mercury exceeded their objective for at least one sample in the year, but dissolved oxygen did so only marginally for 2009. Dissolved oxygen and zinc at the Corinella site, and mercury at the Hastings site (highlighted in red in Table 4.3), were the only water quality indicators that clearly did not meet their objectives.

When a significant failure to meet an objective occurs, it is necessary to assess potential risk to the environment. In the case of metals, the dissolved fraction (the most bioavailable fraction) is the best indicator of the potential threat to the aquatic ecosystem. The national water quality guideline (ANZECC 2000) defines guideline values for the dissolved fraction of metals present in the water column. When compared to the dissolved fraction defined in those guidelines for mercury (with 99% protection level, 0.1 µg/L) and zinc (with 90% protection level, 23 µg/L), exceedances at the Hastings and Corinella sites do not appear to pose a significant risk to the environment, as the dissolved fractions (0.1 µg/L and 12 µg/L respectively) were within acceptable limits when considering measurement uncertainty (0.115 µg/L and 4.13 µg/L respectively).

Site differences

The three monitoring sites are characterised by very different environments. Because 30% of the bay volume is exchanged in each tidal cycle, the waters of Western Port reflect a mixture of the adjacent marine waters and the large intertidal flats that they drain. The Hastings and Barrallier Island sites are more influenced by oceanic conditions, while the Corinella site is dominated mostly by water draining across the northern mudflats and by catchment inputs. The Corinella site is also strongly influenced by the deposition and resuspension of sediments associated with the strong tidal flows and wind-wave mixing that dominate this shallow environment.

Table 4.3 Compliance with SEPP F8 objectives at the three Western Port sites, 2009.

		Policy Categories		Transparency (Secchi disc, m)		Suspended Solids (mg/L)		Total Nitrogen (µg/L)*	Total Phosphorus (µg/L)*	Dissolved Inorganic Nitrogen (µg/L)		Dissolved Inorganic Phosphorus (µg/L)		Chlorophyll-a (µg/L)		Arsenic (µg/L)	Cadmium (µg/L)	Copper (µg/L)	Lead (µg/L)	Mercury (µg/L)	Nickel (µg/L)	Zinc (µg/L)	Dissolved Oxygen (% saturation)
	Sampling Site	SEPP segment	ANZECC Level of Protection	Annual median	Annual 25th percentile	Annual median	Annual 75th percentile	Annual 75th percentile	Annual 75th percentile	Annual median	Annual 75th percentile	Annual median	Annual 75th percentile	Annual median	Annual 75th percentile	Maximum	Maximum	Maximum	Maximum	Maximum	Maximum	Maximum	Minimum
Guideline Values	Hastings	Entrance and North Arm	99%	>2.4	>1.4	<9	<19	<0.3	<0.03	<7	<15	<6	<8	<1.6	<2.1	<3	<0.05	<1.0	<1.0	<0.005	<1.0	<2.0	>90%
Guideline Values	Barrallier Island	Entrance and North Arm		>2.4	>1.4	<9	<19	<0.3	<0.03	<7	<15	<6	<8	<1.6	<2.1	<3	<0.05	<1.0	<1.0	<0.005	<1.0	<2.0	>90%
Guideline Values	Corinella	East Arm	90%	>0.7	>0.4	<30	<90	<0.3	<0.03	<20	<43	<7	<10	<2.5	<5.0	<5	<0.05	<2.0	<2.0	<0.01	<3.0	<5.0	>90%
Monitoring Results (percentile 95% confidence interval in bracket)	Hastings	Entrance and North Arm	99%	3.5 (2.4; 5.5)	2.4 (0; 3.5)	3.1 (1.6; 5.9)	6.4 (3.1; 6.9)	0.14 (0.13; 0.16)	0.015 (0.011; 0.018)	9.9 (4.4; 10.5)	10.7 (9.9; 12.3)	1.6 (0; 3.8)	3.9 (1.6; 5.6)	1.20 (1; 1.4)	1.50 (1.2; 1.6)	2.3 ± 0.4	0.2 ± 0.22	BLR	BLR	0.2 ± 0.13	1.2 ± 0.52	BLR	89.61
Monitoring Results (percentile 95% confidence interval in bracket)	Barrallier Island	Entrance and North Arm		3.1 (2.3; 4.8)	2.6 (0; 3.1)	3.9 (1.9; 5.6)	6.4 (3.9; 9.7)	0.16 (0.15; 0.18)	0.012 (0.007; 0.018)	6.8 (5.7; 8.3)	8.4 (6.8; 8.6)	0 (0; 0.95)	1.5 (0; 2.2)	1 (0.9; 1.2)	1.3 (1.0; 1.6)	2.2 ± 0.38	BLR	BLR	0.4 ± 0.8	0.1 ± 0.12	1.1 ± 0.52	5 ± 3.75	89.20
Monitoring Results (percentile 95% confidence interval in bracket)	Corinella	East Arm	90%	0.8 (0.3; 1.5)	0.35 (0.1; 0.7)	13.1 (7.3; 50.2)	59.5 (13; 72.7)	0.33 (0.21; 0.38)	0.047 (0.016; 0.059)	9 (4.7; 16.8)	17 (8; 52.9)	1.1 (0; 2.3)	2.3 (1; 2.9)	1.8 (1.3; 2.6)	2.7 (1.7; 3.4)	2.6 ± 0.45	BLR	BLR	0.9 ± 0.39	0.1 ± 0.12	3.4 ± 1.04	14 ± 4.22	76.55

SEPP objective lower than laboratory limit of reporting
Monitoring results less than the laboratory limit of reporting
Given uncertainty monitoring results require additional data to be assessed against SEPP
Monitoring results exceeding SEPP
* From SEPP WoV 2003

Particle modelling, shown in Figure 4.29 for periods of one day, one week and one month, provides relative footprints of the integrated number of particle visits per cell and a snapshot of the particle location for a specified period. On the west side there is clearly connectivity between the two monitoring sites (Hastings and Barrillier Island) within a day, but it takes up to a month for these sites to interact with water from the Corinella site. At the end of a month the dominant excursion footprint (dark red) for each site is about 5 km long, skewed somewhat by a net clockwise movement around the bay (for all sites).

When comparing long-term monitoring data from the three sites, strong differences in water clarity, suspended sediments, metal concentrations and productivity are apparent between the Hastings and Barrallier Island sites, compared with the Corinella site (Figures 4.30–4.34).

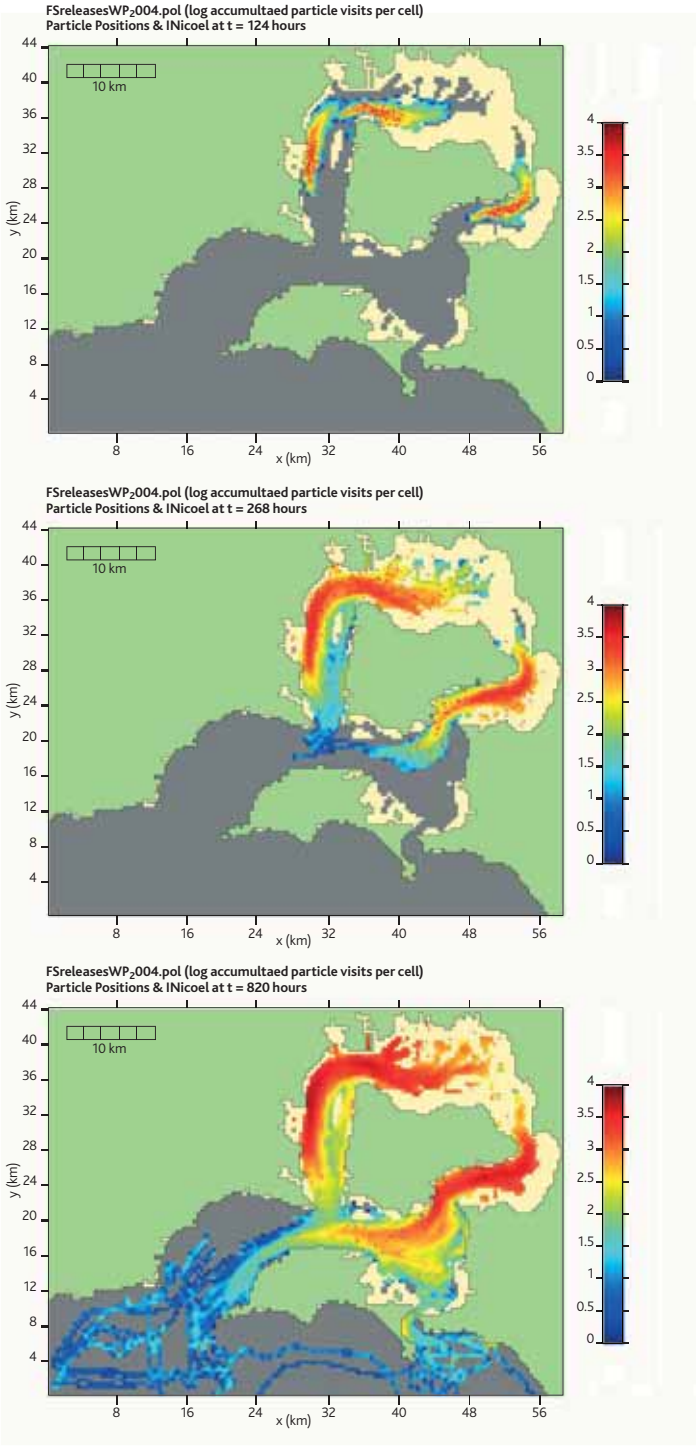
Salinity (Figure 4.30) exhibits the same seasonal pattern at all sites, but with a significantly greater range at the Corinella site. Slight differences can also be observed between the Hastings and Barrallier Island sites. With the adjacent ocean practical salinity typically at 35.5, seasonality is driven predominately by strong evaporation during summer across the shallow mudflats. The lower salinity in winter shown at Corinella indicates this site has greater responsiveness to the higher catchment inflows to the bay during winter–spring.

Associated with the observed salinity patterns, total suspended solids (TSS in Figure 4.31), nutrients (presented as NOx — the sum of oxidised nitrogen; Figure 4.32), chlorophyll-*a* (Figure 4.33) and metals (presented as nickel; Figure 4.34) show similar differences between the Corinella site and the other two sites.

For metals (Figure 4.34), the Corinella site exhibits consistently high measurements compared with the other two sites. These are mostly related to the differences in the observed suspended sediment load.

The relative composition of phytoplankton pigments (chlorophyll-*a, b, c*, carotenoids and pheopigments) was also assessed.[4] Although there were clear differences in the physico-chemical characteristics of the water, no major difference was observable in the phytoplankton pigments abundance between the Corinella site and the other two sites. Phytoplankton communities thus appear to be almost identical at the Hastings and Barrallier Island sites, but cyanobacteria are more abundant at the Corinella site.

Figure 4.29 Pol3DD Lagrangian particle dispersion model results for releases at the three EPA monitoring sites for (a) 1 day [100 + 24 hours], (b) 1 week [100 + 168 hours], and (c) 1 month [100 + 720 hours]. (Source: EPA unpubl. data.)



4 Note that the phytoplankton community can be inferred from pigment composition, but it can be confirmed only by phytoplankton identification and counting.

Figure 4.30 Median of monthly salinity (practical salinity) measurements from 1990 to 2009 at the Hastings, Barrallier Island and Corinella sites. (Source: EPA 2011.)

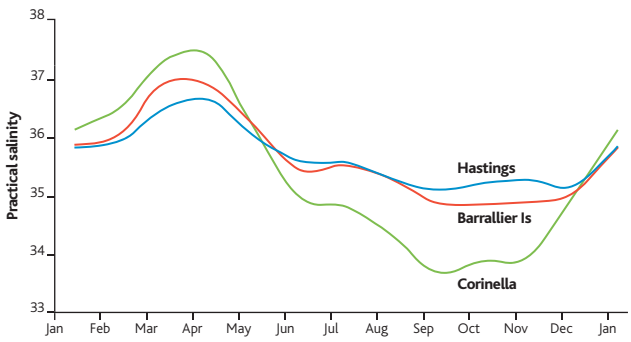


Figure 4.31 Median of monthly total suspended solids measurements from 1990 to 2009 at the Hastings, Barrallier Island and Corinella sites. (Source: EPA 2011.)

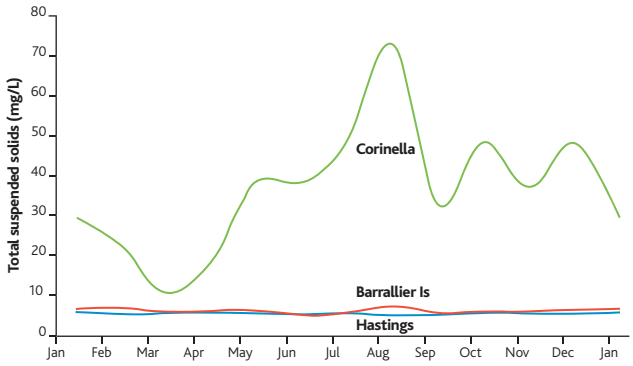


Figure 4.32 Median of monthly NOx measurements from 1990 to 2009 at the Hastings, Barrallier Island and Corinella sites. (Source: EPA 2011.)

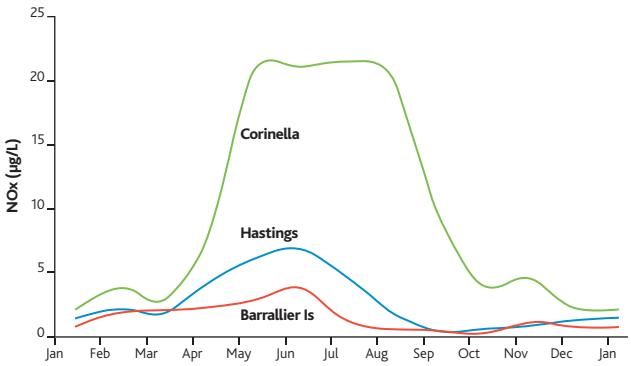


Figure 4.33 Median of monthly chlorophyll-*a* measurements from 1990 to 2009 at the Hastings, Barrallier Island and Corinella sites. (Source: EPA 2011.)

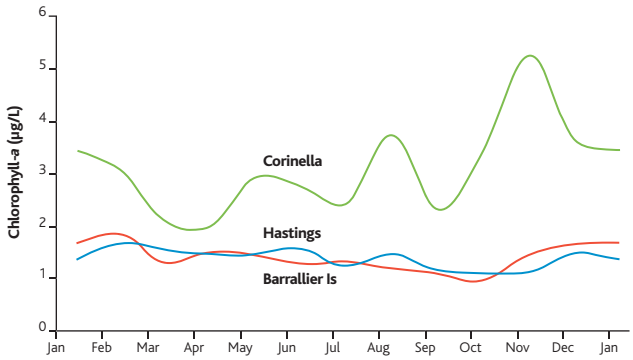
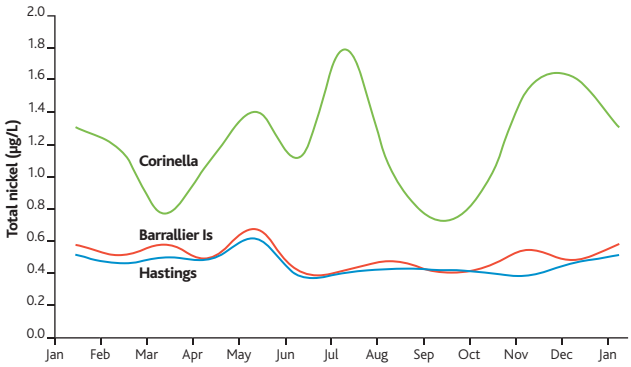


Figure 4.34 Median of monthly nickel measurements from 1990 to 2009 at the Hastings, Barrallier Island and Corinella sites. (Source: EPA 2011.)



Long-term trends in water quality

Long-term trends in the water quality of Western Port are presented in Figure 4.35 and Figure 4.36. Where sites show similar trends for the same parameter, results for only one site are illustrated. Regression models were derived for salinity and dissolved oxygen, based on previous values, seasonal cycles and linear trends. Diagnostic tests were applied to assess goodness of fit of the regression models.

Salinity has shown a significant upward trend at the Barrallier Island site with an increase in the practical salinity of 1 over 19 years, with a seasonal difference of 3 (Figure 4.35). All sites have followed the same significant upward trend with an increase of about 0.55 with a seasonal difference of 2 at Hastings and about 1.55 increase with a seasonal difference of 4 at Corinella.

Dissolved oxygen at all sites showed a significant downward trend of about 6 % saturation over 19 years at the Hastings and Corinella sites (Figure 4.36) and about 8% at the Barrallier Island site.

Baseline shifts in water quality

Western Port is under the combined pressures of rapid urbanisation of the catchment, coastal development, potential port expansion and the potential effects of climate change. Currently, Western Port point source pollution is limited, with only three EPA licensed discharges within the bay (two industrial facilities and one sewage treatment plant, Figure 4.37). Given that SEPP Schedule F8 (Victorian Government, 2001) states that 'Clause 29 also requires the operators of premises with a capacity exceeding 0.1 ML/day to ensure that by 2011, they do not cause detrimental change in the environmental quality of receiving waters', it is expected that pressure from point source discharges will be further reduced. Therefore the focus should be on the diffuse catchment sources and development/urban expansion sources and their effects in the light of likely climate impacts.

To analyse the contemporary impact of climate on the water quality of the bay, the fixed site data set was partitioned into two periods, separated by an observed climatic shift in 1997–98 associated with a strong El Niño event that marked the beginning of a long-term drought in south-eastern Australia. The 'before drought' period from 1990 to 1997 is taken as representing climatic conditions for a 'normal' year. The 'drought' period from 1998 to 2008 represents a dry year.

The Bureau of Meteorology stations at Cerberus and Lang Lang can be used to understand climatic changes around Western Port over that period (Figure 4.37). These stations are the only ones close to Western Port that offer data sets suitable for temperature (Cerberus) and rainfall (Lang Lang) assessment. The mean annual maximum air temperature was about 1°C higher in the drought period. The monthly mean maximum temperature was also consistently higher in the drought period throughout the annual cycle, the difference ranging from 0.4 to 1.7°C, but there was no detectable change in the timing of seasons.

Mean annual rainfall deceased by about 135 mm (18%) between the normal and drought periods. Mean monthly rainfall differences ranged from –31 mm to +10 mm. Normal years were characterised by higher rainfall from June to September (winter storms) compared to the drought years (Figure 4.38). Otherwise, rainfall patterns in the two periods were similar.

Figure 4.35 Mean maximum temperature at Bureau of Meteorology Cerberus station. (Source: EPA 2011.)

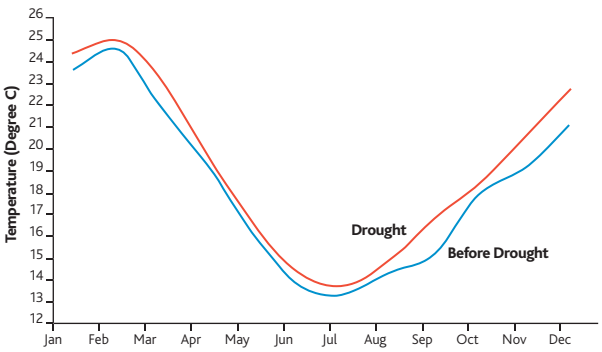


Figure 4.36 Long-term trend in salinity (practical salinity) from 1990 to 2009 at the Barrallier Island site. (Source: EPA 2011.)

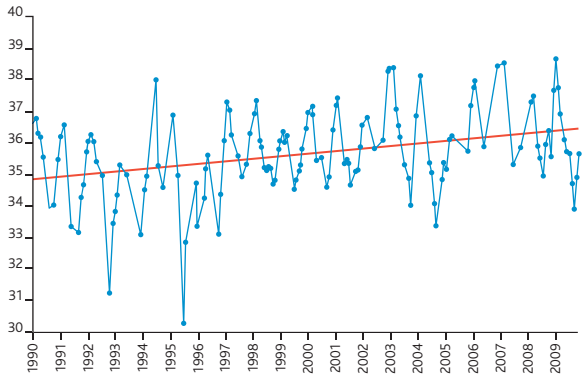


Figure 4.37 Long-term trend in dissolved oxygen from 1990 to 2009 at the Barrallier Island site. (Source: EPA 2011.)

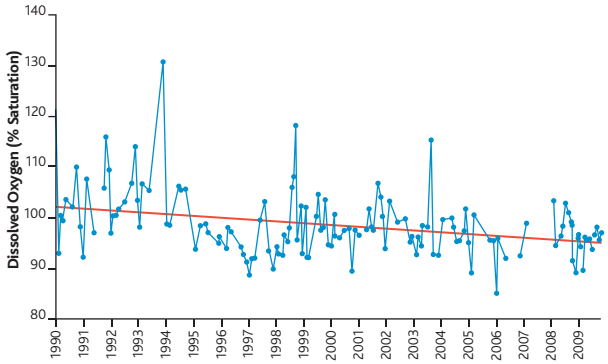
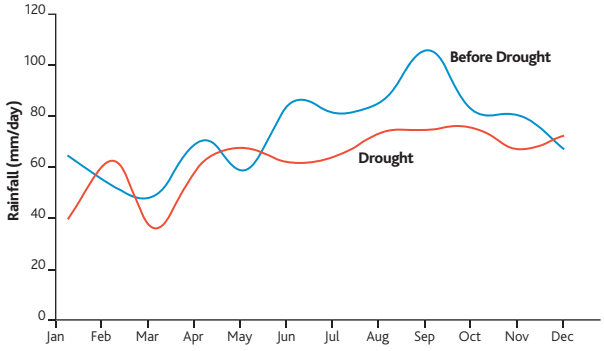


Figure 4.38 Mean monthly rainfall at Lang Lang Bureau of Meteorology station. (Source: EPA 2011.)



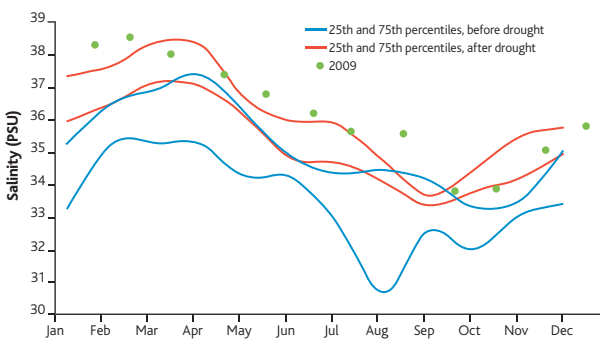
Few water quality parameters showed differences between the normal and drought periods. The most noticeable differences were in salinity and NOx concentrations at all sites. Figures 4.39 and 4.40 illustrate those differences at the Corinella site, and show that the data from 2009 were still indicating drought conditions. Salinity in the drought period showed two major differences to the normal years (Figure 4.39). First, the monthly interannual variability (as shown by the 25th and 75th percentiles) was much reduced during the dry period. That is, one year is more like the next during the drought compared to the normal period. This would be linked to the observed decrease in seasonal rainfall and reduced winter storm intensities. Second, salinities were generally higher, and elevated monthly salinities were most pronounced during summer (1–2 higher), most likely because of a higher evaporation rate.

With greater evaporation in summer and reduced catchment inflows in winter, Western Port is getting saltier and almost maintains hypersaline conditions all year round. This condition often occurs in poorly flushed bay systems during droughts (as in Port Phillip Bay), but is less common in well-flushed systems such as Western Port.

The observed seasonal differences in NOx between the normal and drought periods reflect the general decrease of fresh water input to Western Port. NOx concentration for the normal period (Figure 4.40) were characterised by a fairly low monthly variability (around 20 µg/L) all year round except from June to September, when difference between the 25th and 75th percentile reached about 75 µg/L. Also, the highest NOx concentrations (around 120 µg/L for the 75th percentile) were measured during July, associated with winter–spring rains. In the drought period monthly NOx concentrations were generally lower than during the pre-drought years, and the seasonal peak was significantly reduced.

Western Port exhibits a wide intra-annual variability in water quality, showing a strong ability to respond quickly to loading events (e.g. catchment inflows causing sediment resuspension and transport) associated with wet weather and tidal variations. Monthly sampling appears sufficient to provide a broad general view of the health of the bay, but perhaps is not sufficient to fully assess the ecological health of the system, which may respond to events at shorter time-frames (e.g. algal blooms). This is illustrated by the large number of indicators that were, or might have been, above their water quality objective.

Figure 4.39 Difference in salinity (SP)between drought (1997–2008) and normal (1990–1996) periods at the Corinella site. 2009 salinity (SP) measurements are also presented in this historical context. (Source: EPA 2011.)



The baseline shifts between the normal and drought periods show that the Western Port marine ecosystem in the 2000s was experiencing a different regime because of increased salinity and temperature stressors, and it may therefore be more vulnerable to further changes. This also represents a potentially new baseline against which the system's receiving capacity under the increasing pressure of development and urbanisation could be assessed.

The rainfall reductions experienced in south-eastern Australia during the 1998–2009 drought were similar to the forecast reduction of about 25% by 2070 as a result of climate change (Whetton and Power 2007). It is possible that this is not a 'step change' in climate but a regional drought that will pass in time with a return to rainfall patterns more similar to previous baseline conditions, and in line with climate changes projections.

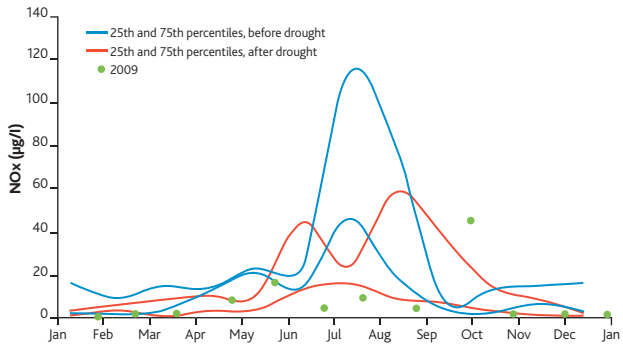
While fixed site sampling is able to identify these baseline shifts, the impact on the ecology of the bay is unclear. Understanding the impacts in the light of other impacts from existing or future anthropogenic loading to the bay will inform more effective management decisions.

Dispersion modelling

This section of the report outlines results from the RWQM dispersion modelling in Western Port. The model used time-series loads taken from the PortsE2 model (BMT WBM 2007), and introduced them at 21 input locations around the bay. Figure 4.41 shows the input locations for total nitrogen loading to the bay. This Lagrangian dispersion model tracks the simulated particles in three dimensions, subjecting particles at different levels to different currents predicted by the three-dimensional hydrodynamics. Although each particle has a precise position that is not limited by the resolution of the hydrodynamic results, the following results are presented as depth averages on the 400 m hydrodynamic grid. Further detail on the dispersion model settings can be found in Harrison *et al.* (20011a, 2011b).

With the combination of bay flushing and typical die-off rates of 24 hours adopted in the modelling, maximum pathogen concentrations occurred in the immediate vicinity of the Western Port sources (Figure 4.42). The results suggest a localised footprint from the catchment sources, although with the poorer flushing in the north there is a general increase in footprint size.

Figure 4.40 Difference in NOx concentration between drought (1997–2008) and normal (1990–1996) periods at the Corinella site. NOx measurements in 2009 are also presented in this historical context. (Source: EPA 2011.)



For total suspended solids the results for the integrated number of particle visits (Figure 4.43) show the influence of the relatively strong flows through the main channel of Western Port. Areas that a high number of particles have visited (shown in red) are regions where there is a large mass of sediment being transported but not necessarily high suspended sediment concentrations (Figure 4.44) or deposition (Figure 4.45). Areas of high suspended sediment concentration and highest deposition are limited to the immediate vicinity of the sources.

Model hydrodynamics indicate that most of these areas also correspond to areas where predicted tidal currents are low for extended periods of time, so they are likely to have high deposition rates. The areas where fine sand-silt are observed within Western Port (Marsden *et al.* 1979) show relatively good agreement with the areas where the model is predicting a medium level of deposition (see Figure 4.2).

Predicted concentrations of toxicants (Figure 4.46) and total nitrogen (Figure 4.47) show the same broad pattern, with highest predicted levels in the north-east of Western Port. Residual currents in the channels of that part of the bay tend to be flood-dominant, so contaminant levels there will be closely linked to catchment inputs along the northern shores of Western Port. Contaminants from other catchment sources tend to be effectively flushed from Western Port because of the relatively strong currents within the main channels and the entrances to Western Port.

The results from the sediment transport/settlement model for Western Port (Figure 4.45) represent the distribution of sediment discharged from catchments, as a result of fallout once discharge momentum has been absorbed by the bay's receiving waters. That is, when the vertical fallout rate of sediment particles overrides the horizontal flow. To account for the resuspension of material from in-bay processes such as strong tidal currents and wave mixing, the model was augmented with time-series wave data to generate mixing. A threshold velocity for sediment bedload transport (0.07m/s) and resuspension (0.1m/s) was applied to account for resuspended transport and resettlement for the predominant mud/silt fractions in Western Port (mean grain size of 0.01565 mm). Figure 4.48 compares the results when resuspension is included, indicating significant increases in total suspended solids (about two orders of magnitude) and enhanced focus of high turbidity in the eastern arm of the Western Port.

Figure 4.41 Predicted mean annual total nitrogen loads (kg/s) from the PortsE2 model for each catchment outlet used in the Western Port RWQM. (Adapted from Harrison *et al.* 2011b.)

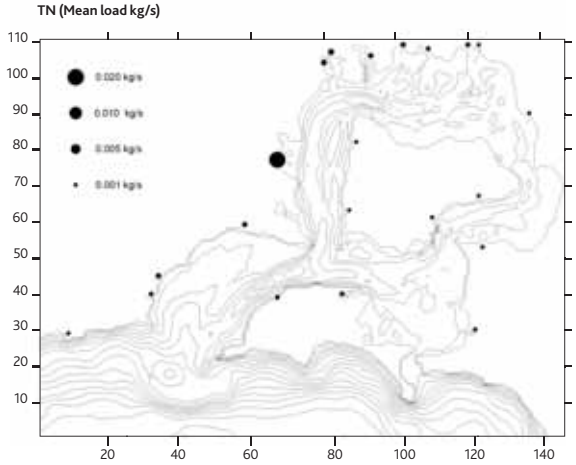


Figure 4.42 Mean concentration of pathogens (pathogens/mL) from rivers in Western Port in 2004, plotted on a logarithmic scale. (Adapted from Harrison *et al.* 2011b.)

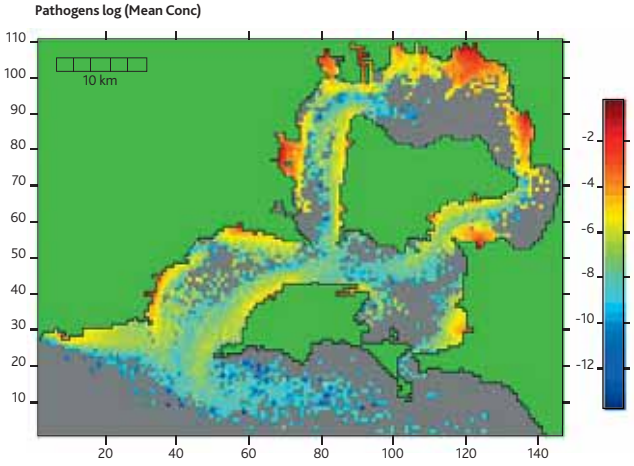


Figure 4.43 Mean integrated number of visits to cells by suspended solids (mud, 6 phi) delivered by rivers in 2004, plotted on a logarithmic scale. A larger integrated number count suggests increased availability of mud in suspension as well as indicating major transport routes. (Adapted from Harrison *et al.* 2011b.)

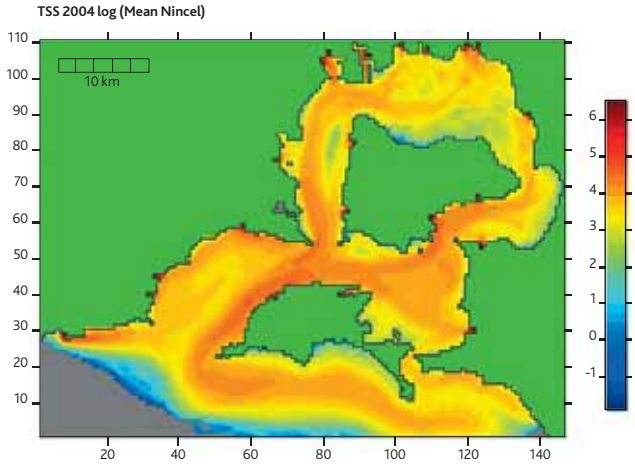
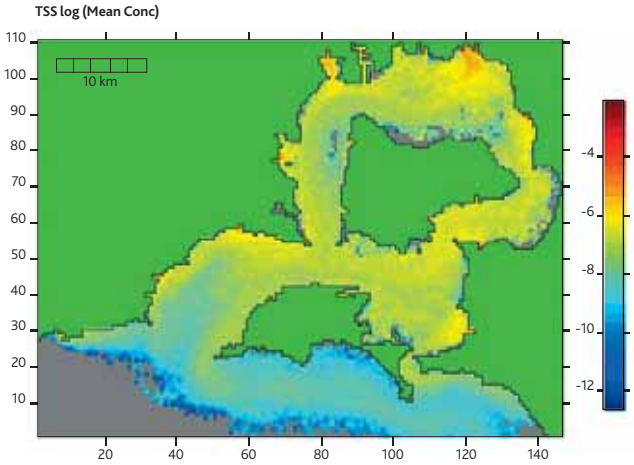


Figure 4.44. Mean concentration (kg/m3) of suspended solids (mud, 6 phi) from rivers flowing into Western Port in 2004, plotted on a logaithmic scale. Note the elevated levels around the Bunyip River near the tidal divide in the north-east of the bay. (Adapted from Harrison *et al.* 2011b.)



Discussion

Based on the broader process studies of Western Port (Shapiro 1975, EPA 1996, Wallbrink *et al.* 2003), the non-compliance of the Corinella site with water quality objectives in SEPP is likely to be caused by the resuspension of suspended particulates (remobilising nutrients and toxicants from sediments) within the bay. At present the sources, pathways, and sinks cannot be resolved by the monthly sampling program. But the decrease in rainfall and loss of winter storms has led to a decrease of nutrients in the bay, probably because of a decrease in inputs from the catchment (see Figure 4.40).

Inflow at the Western Entrance is also a possible source of nutrients and toxicants discharged from the Eastern Treatment Plant at Boags Rocks (EPA 2002). The monitoring program, however, is confined at present to embayment and catchment sources, so it cannot address this issue.

The monitoring program also has no specific ecological health indicators to link measured water quality to ecosystem health. Chlorophyll-a is used as a proxy indicator for phytoplankton, but events such as red tides (involving plankton dominated by red pigments) are not effectively detected with this green pigment indicator. Assessments using a broader spectrum of pigments can assist in identifying major plankton groups. However, *in situ* pigment proxies have a limited ability to define any plankton community shifts that may related to climate changes or increased loading.

The study by Wallbrink *et al.* (2003) concluded that the re-establishment of North Arm seagrass beds is a priority for improving seagrass survival elsewhere. At present there is no specific monitoring program focusing on seagrass health linked to the water quality program. Targeted monitoring is needed at temporal and spatial scales that reflect system dynamics to resolve threats and how their impacts are manifested within Western Port (see Chapter 10).

Figure 4.45 Mean settlement (log of number of particles) of river-sourced sediment (mud, 6 phi), 2004. (Adapted from Harrison *et al.* 2011b.)

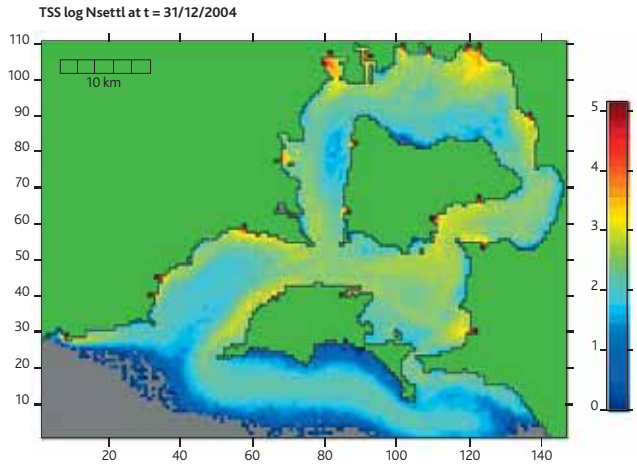


Figure 4.46 Mean concentration (kg/m³) of toxicants from freshwater input, 2004. Levels are higher near the Bunyip, Lang Lang, and Corinella segment in the east. (Adapted from Harrison *et al.* 2011b.)

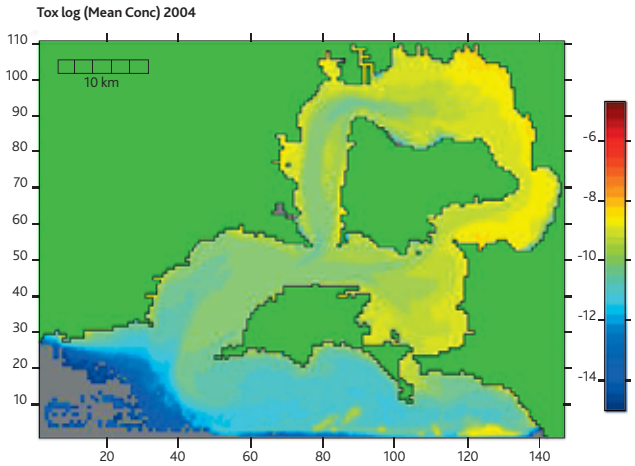


Figure 4.47 Mean concentration (kg/m³) of nitrogen (all types) from freshwater inputs in 2004, plotted on a logarithmic scale. (Adapted from Harrison *et al.* 2011b.)

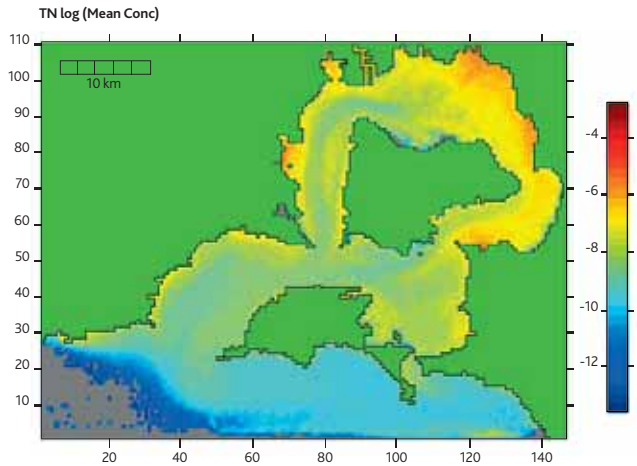
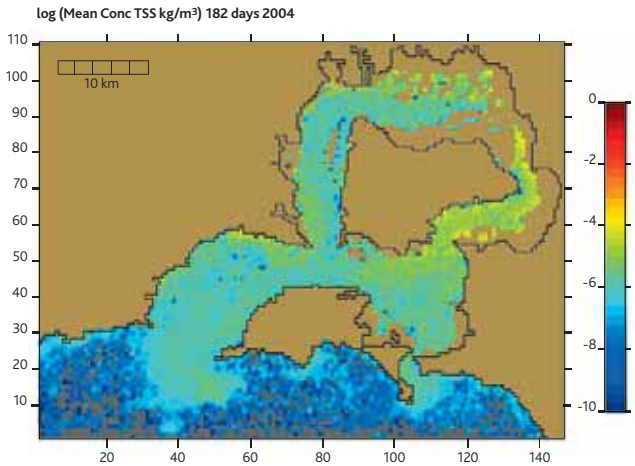


Figure 4.48 Total suspended solids (kg/m³) with wave mixing and sediment resuspension included. (Adapted from Lee *et al.* 2011.)



Climate change

The Victorian Coastal Strategy (2008) identifies the following broad level changes to Victoria's coast as the likely consequences of climate change:

'Over the medium to long term, climate change poses real and serious threats to our coast. During this century, it is likely the Victorian coastline will be impacted by sea level rise and increased frequency and severity of storm events leading to inundation and erosion. It is also predicted that higher temperatures will increase bushfire risk along the coast, and increased sea temperatures, changing sea currents and further acidification of the ocean will affect the marine environment.'

In Western Port several climate impacts are predicted to be among the highest extremes experienced across the state. The most robust findings are detailed in the following section, which focuses on the key physical impacts on an embayment. Sea level and inundation predictions are drawn from recent work by McInnes *et al.* (2009a,b). Projections for rainfall and evaporation (which incorporate the impact of both land and sea temperature increases) were made by Whetton and Power (2007) and have been incorporated into catchment and dispersion models by Lee *et al.* (2011). Other physical consequences of climate change, such as changes to wind and current patterns, could affect conditions in Western Port, but at a regional scale the predicted impacts are very uncertain (Whetton and Power 2007) and so have not been included here.

Sea-level rise and inundation

The global average sea level has risen at an average rate of 1.8 mm/year since 1961 and 3.1 mm/year since 1993 (Church *et al.* 2010). Locally, recording stations at Stony Point have measured sea-level rises of 2.4 mm/year since 1991 (McInnes *et al.* 2009a).

Sea level rise on its own will not have the greatest impacts on the coast but it is its combined effects with tides, storm surges and local conditions such as topography, elevation and geology that will produce a range of impacts and increased risks to coastal areas (McInnes *et al.* 2009b). The impacts are most likely in the lowest areas of land. The Victorian Government's Future Coast Project is assessing these impacts, focusing on the physical impacts of sea-level rise as a result of climate change (DSE 2010).

The largely westerly to southerly winds associated with the passage of cold fronts have been found to be the main driver of storm surges along the Victorian coast (McInnes and Hubbert 2003, McInnes *et al.* 2005). McInnes *et al.* (2005) noted that storm surge heights in Bass Strait respond linearly to changes in wind speed, with a 1% increase in wind speed corresponding, approximately, to a 2% increase in storm surge height. The spatial pattern of 1 in 100 year storm surge heights for the Victorian coast under late 20th century climate conditions is shown in Figure 4.49.

The term 'storm tide' combines storm surge with the astronomical tide, representing actual sea levels experienced along the coast during a storm. The spatial pattern of 1 in 100 year storm tide heights for the Victorian coast under late 20th century climate conditions is shown in Figure 4.50, with 2100 scenarios without and with sea-level rise. The highest coastal values, in excess of 2 m, occur in and around Western Port, with an extreme focus on the north-eastern shoreline, where storm tides would exceed 3.4 m by 2100 (McInnes *et al.* 2007).

Several scenarios were provided by McInnnes *et al.* (2009a) to assess the local impact of sea-level rise in Western Port. Scenario 1 considers the IPCC's high emissions scenario for mean sea level (IPCC 2007), Scenario 2 combines the high sea-level rise scenario with the equivalent high annual averaged wind speed change averaged over Bass Strait from the CSIRO and Australian Bureau of Meteorology (2007), Scenario 3 considers the upper sea-level rise scenario developed for the Netherlands Delta Committee, and Scenario 4 considers the upper sea-level scenario proposed by Rahmstorf (2007).

Figure 4.51 shows that present-day 1 in 100 year storm tides will be much more frequent in the future. At Stony Point the present 1 in 100 year storm tide height, 2.08 m, may be exceeded every 30 years on average by 2030. By 2070, sea-level rise alone may result in a 2.08 m event occurring on average every 8 years, and an increase in wind speed may further decrease the return period to 5 years. By 2100, under any of the climate change scenarios considered, such an event would be experienced at least once every 1.5 years on average (McInnes *et al.* 2009b). Melbourne Water have also undertaken an analysis of historical records and projected climate scenarios for Western Port (Melbourne Water 2010). Their less conservative treatment of historical data resulted in a higher current day estimate for the 1 in 100 year storm tide height of 2.7 m. Combined with an 0.8 m rise in sea level by 2100 brought this height to 3.5 m, which matches the upper estimate (Scenario 4) in McInnes *et al.* (2009b).

The northern shoreline of Western Port, where some of the highest projected storm surges in the state are predicted to occur, includes mangrove-fringed marshland that will become increasingly prone to inundation under future sea-level rise (Port of Melbourne 1992). The extensive wetland and intertidal areas along the coast provide important habitats for bird and marine species. The main settlements in this region are Warneet and Tooradin.

Figure 4.52 shows the existing and projected scenarios for 1 in 100 year storm tide inundation. Under the current climate about 43km^2 could be inundated. By 2030 and 2070 under Scenario 1 the area would be respectively about 15% and 50% larger, and the number of land parcels affected would be 30% and 110% greater. By 2100 the total area vulnerable to inundation would be 82 km^2 under Scenario 1, and 110 km^2 under Scenario 4 (McInnes *et al.* 2009a).

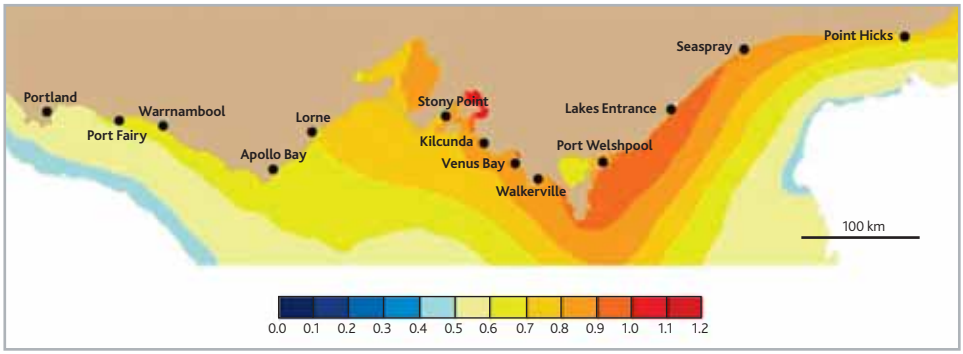


Figure 4.49 The spatial pattern of 1 in 100 year storm surge heights for the Victorian coast under late 20th century climate conditions. Note that these do not include a tidal component. Values are in metres relative to late 20th century mean sea level. (Adapted from McInnes *et al.* 2009a.)

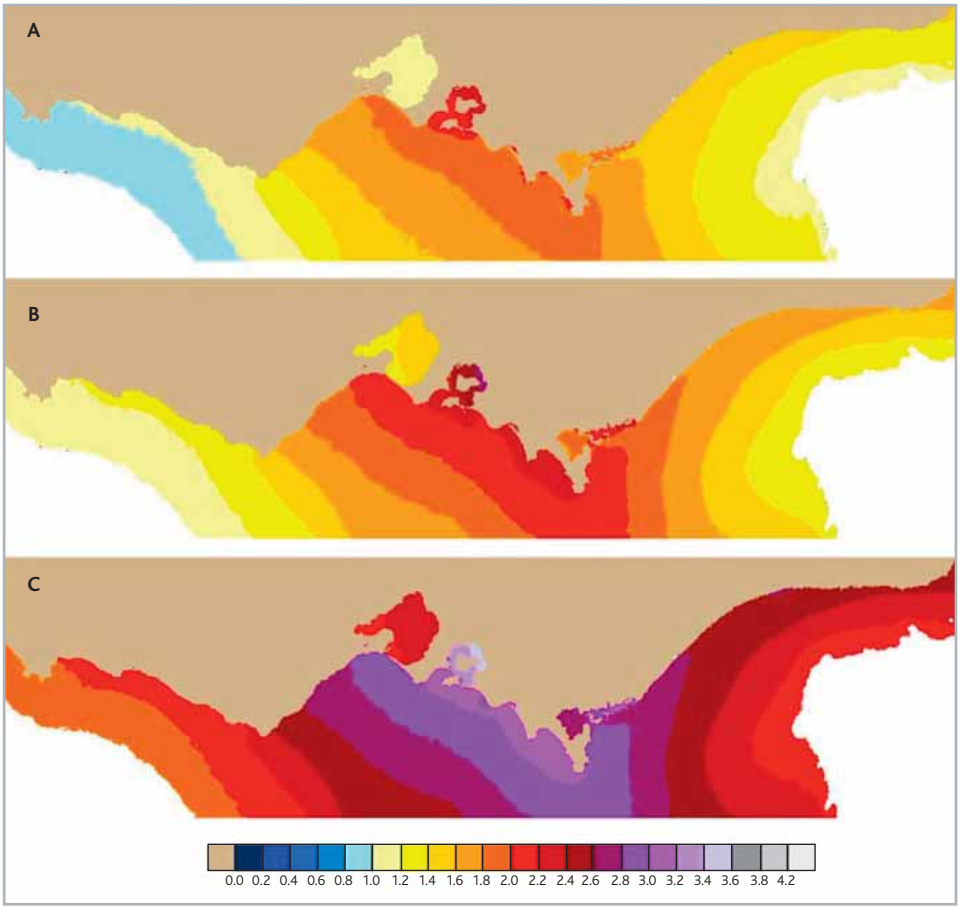


Figure 4.50 The spatial pattern of 1 in 100 year storm tide heights (m) for the Victorian coast under (a) late 20th century climate conditions, (b) including wind speed increases for 2100 without sea-level rise, (c) for wind increases and sea-level rise in 2100. Values are in metres relative to late 20th century mean sea level. (Adapted from McInnes *et al.* 2009a.)

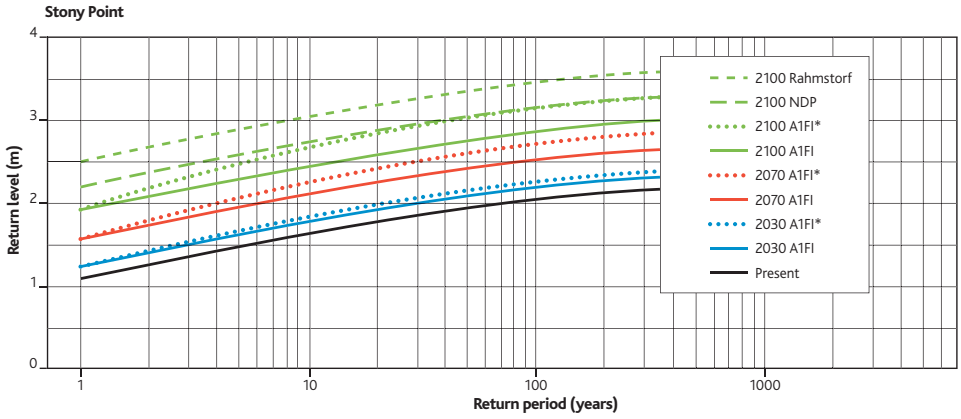


Figure 4.51 Storm tide height return period curves for selected Victorian locations under current climate conditions and climate change scenarios. Asterisks denotes scenarios that incorporate wind speed changes. (Adapted from McInnes *et al.* 2009a.)

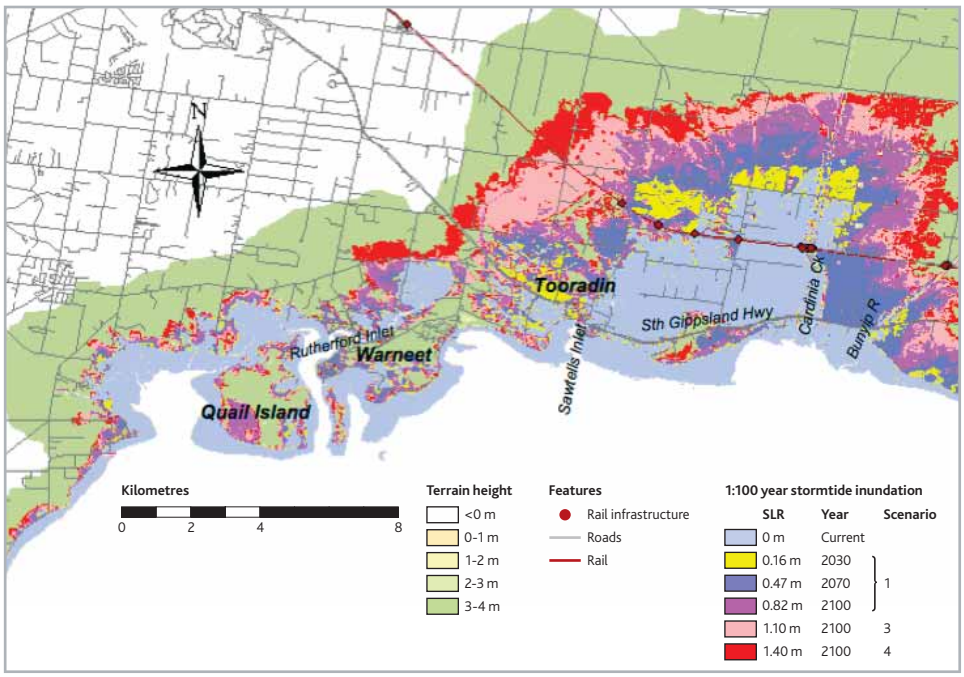


Figure 4.52. Land vulnerable to inundation during a 1 in 100 year storm tide under current climate conditions and various scenarios of future sea level rise for the Tooradin region. (Adapted from McInnes *et al.* 2009a.)

Rainfall and evaporation

The climate change projections adopted here (Figure 4.53) are based on the spatially coarse projection data freely available for the region from CSIRO's Ozclim model simulator, and subsequently from Whetton and Power (2007).

PortsE2 catchment flows have been simulated for 2030 climate change (BMT WBM 2007) using a blanket 4.7% increase in evaporation and a 2.7% decrease in rainfall based on widely varying CSIRO estimates for 2020 and 2050, although essentially based upon the CSIRO Mk3 2030 A1F1 scenario medium sensitivity run for annual conditions in the Melbourne region (Whetton and Power, 2007). This simulation resulted in an average decrease of about 10% in flows from the Western Port catchments.

Hydrodynamic models were run with both the adjusted in-bay 2030 evaporation and rainfall conditions and PortsE2 inputs. Compared to the 2004–05 model (Figure 4.54), they indicate that mean salinity in the bay would increase by 0.5 to 1, and temperature would increase by up to 1°C. These increases are similar to the climate shift during 1990–2009 that were discussed in the previous water quality section (see Figures 4.35, 4.37). This suggests that the 2030 conditions would be no worse than those experienced during the prevailing 1998–2009 drought.

Flushing effects

Flushing studies were done for Western Port to assess potential changes in residence time due to climate change scenarios. Both current (2004–05) and future (2030) scenarios were modelled. The Lagrangian particle dispersal model linked to hydrodynamic output from models of 2004–05 and 2030. In all cases the entire bay was filled with particles, which were then allowed to drift throughout the simulation. As particles left the bay they were counted and removed from the simulation. This enabled a time-series analysis of relative flushing rates. By fitting the flushing rate to an exponential decay, the T90 decay (flushing) rate of particles was approximated. This rate varies with time and is a function of both seasonal wind conditions (from north versus south) and freshwater load variations. The distributions of the T90 decay rates are shown in Figure 4.55. The largest flushing rates are associated with highest freshwater inputs and winds from the north. Conversely, the lowest flushing rates occur during periods of low freshwater input and in periods of light winds or when winds are from the south so that particles are effectively held within the bay. The comparison of the 2004 and 2030 cases indicate there would be a very small increase in flushing time in 2030, probably related to a reduced inflow and throughflow from the catchments combined with increased bay salinities.

Figure 4.53 Annual 50 percentile decrease in rainfall (top) and increase in evaporation (bottom) for Victoria (Source: Whetton and Power 2007.)

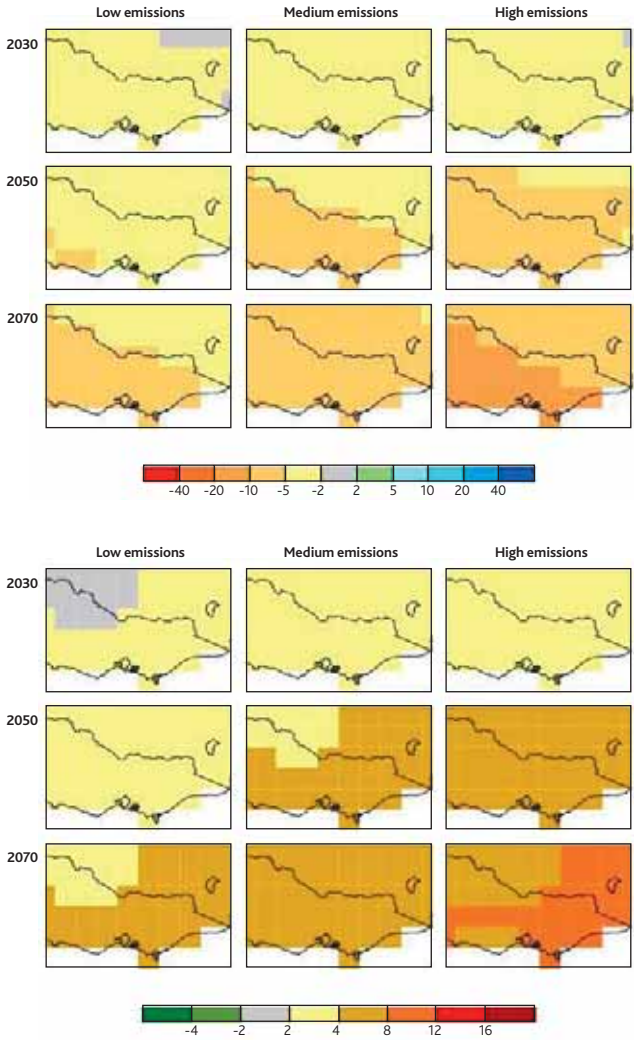


Figure 4.54 Mean Western Port salinity (absolute salinity – g/kg), temperature and circulation conditions for 2030 compared to 2004. (Adapted from Lee *et al.* 2011.)

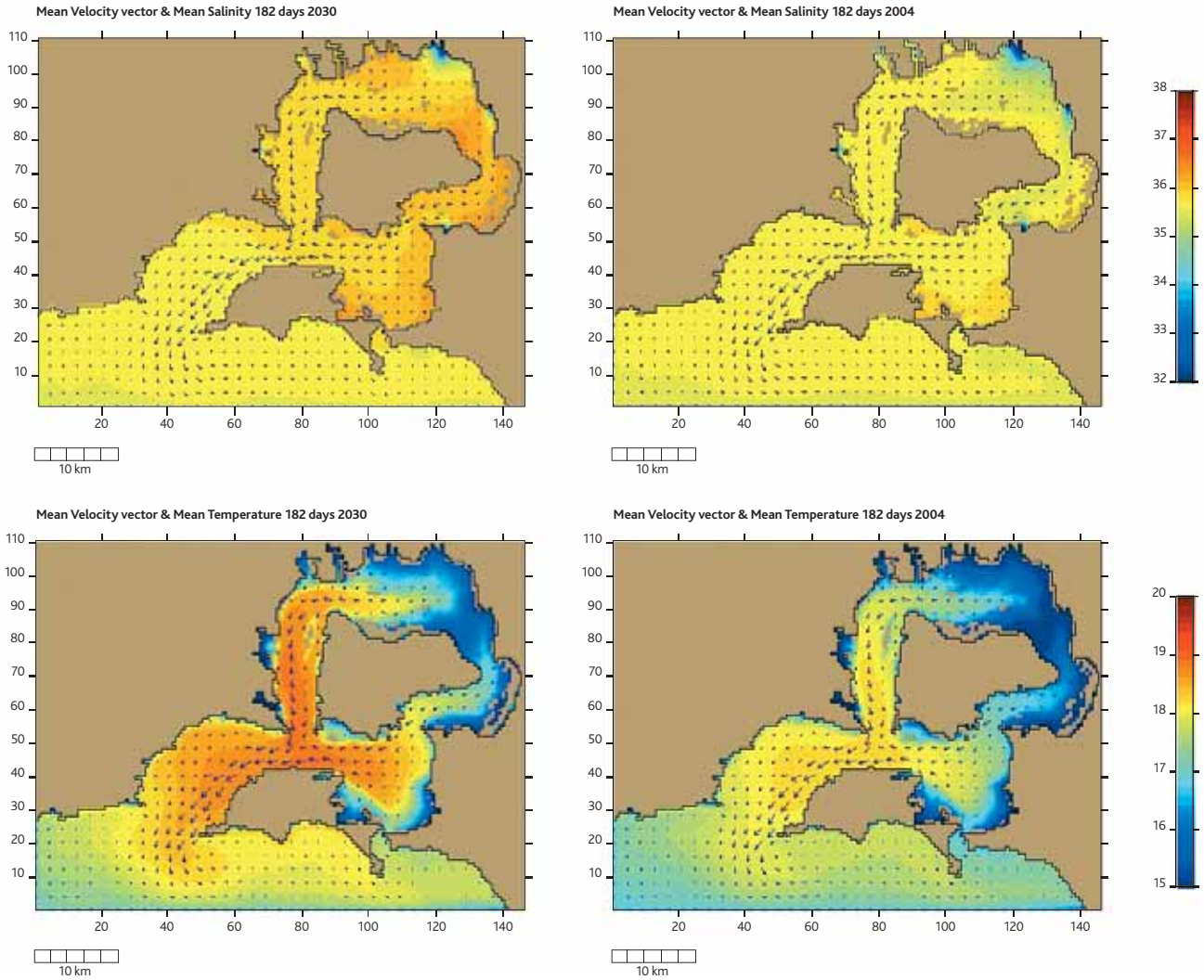
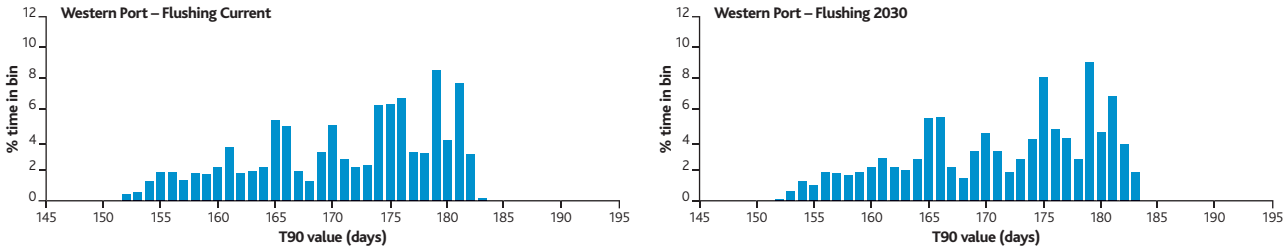


Figure 4.55 Distribution of Western Port T90 residence times (in days) throughout the 2004 (left panel) and 2030 (right panel) flushing simulations. Simulations based on the flushing of a conservative tracer from the Port. (Source: ASR, unpubl. data).



Knowledge gaps

Although comprehensive, most of the studies were done in the 1970s and are now less relevant because changes in climate, population pressure, seagrass coverage and sediment dynamics. Given the scale and complexity of the system it is not surprising that there remain significant process-based gaps in our knowledge. These constrain the robustness of planning and confidence in decision-making, which could lead to unexpected environmental impacts.

The Western Port Research Coordination Project conducted by the Coastal CRC and CSIRO (Counihan *et al.* 2003) provided one set of physical process information needed to inform the effective management of the system. While some of the issues they identified are now being addressed, many of their concerns are still outstanding and are discussed here.

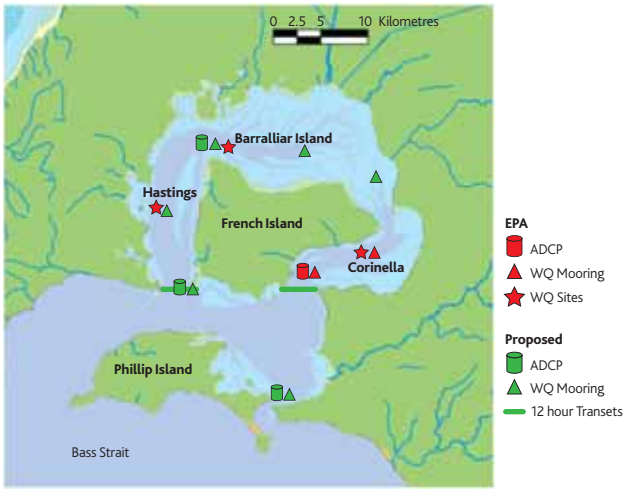
Hydrodynamics

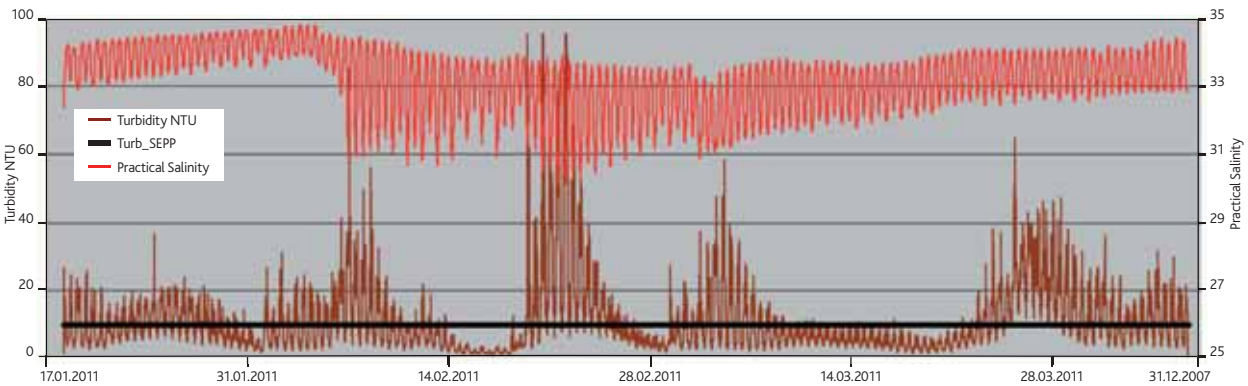
While system-wide hydrodynamics have been adequately described, less is known of the finer-scale hydrodynamics that need to be understood in order to understand the connectivity within the system. To gain any benefit from finer-scale model grids, the incorporation of the Future Coasts LIDAR surveys, digital elevation model products (of topography) and additional multi-beam surveys (of bathymetry) undertaken in the Western Port region will be required.

The application of hydrodynamic, dispersion and catchment loading models such as RWQM has been constrained by limited existing information to calibrate, refine and validate the models. This has also limited additional developments such as the inclusion of sediment transport and plankton dynamics (as undertaken in neighbouring Port Phillip Bay) that would improve the assessment of future scenarios (Black *et al.* 2011). This shortfall has been recognised by the Western Port Science Review in the light of Task 8 —'Identify opportunistic monitoring/studies which could be foreshadowed prior to completion of the review to align with and/or leverage other programs or investment opportunities.' In response, an array has been deployed at key choke points in the system for about two months to augment moorings recently deployed by EPA Victoria (Figure 4.56). This information, in combination with the recent bathymetric surveys, will provide a consistent dataset to validate hydrodynamic models for key forcing dynamics (tides, weather). Initial results from the Corinella site, shown in Figure 4.56, indicate the dominance of tidal oscillations and weather to salinity and turbidity measurements. Note also that the < 10NTU 75th percentile criteria for turbidity, is breached frequently depending on the tidal phase or catchment inflows.

Recent work has indicated that the shallow Western Port system is highly sensitive to conditions on the mudflats that transfer to the water column during tidal exchange. More sophisticated models are required to represent this process, supported by targeted continuous data collection of key physical parameters (temperature, salinity, solar radiation).

Figure 4.56 Proposed short-term network of mooring deployments in Western Port (supported by the science review) that would augment existing moorings to provide a consistent model calibration dataset. Salinity and turbidity from EPA's Corinella Mooring for the initial deployment period 18/1/2011 to 5/4/2011 (EPA unpublished data). The <10NTU SEPP criteria for turbidity (75th percentile) is shown for comparison.





This approach, supported by high-quality measurements, was used for Port Phillip Bay and resulted in models that were sensitive to salinity dynamics (Figure 4.57).

Although sediment has been identified as a contributing factor in seagrass decline, there is little evidence that quantifies the dynamics (at appropriate scales) in areas of seagrass that would relate to sediment accumulation thresholds (Wallbrink and Hancock 2003).

Because of the critical connection between sedimentary and catchment processes and the seagrass beds and Ramsar wetlands, a sound catchment model is essential to support environmental decision making. The recent redesign of the E2 catchment model as WaterCAST, and a re-analysis by Stewart (2011), has produced a consistent dataset spanning 1995–2008. This work showed that the earlier model overestimated catchment flows (Figure 4.58) and calculated uncertainties associated with TSS and nutrient loads. The results (Figure 4.59) are a good match to measured flows for the region. This upgraded WaterCAST model will soon be availableand will supersede the capability of the PortsE2 catchment model used by EPA Victoria and Melbourne Water for the Better Bays and Waterways program.

Water quality

Identifying the origin of nutrients from the catchment, atmosphere and within-bay processes is an important priority for the management of water quality in Western Port. While recent PortsE2 catchment modelling has provided an initial step, more robust modelling is needed to account for contributions from groundwater and instream processes. Other catchment modelling tools such as CAT (DSE 2007) and Source Catchments (eWater 2010) have been developed recently to address these issues..

The changes in sediment and sediment-associated nutrient inputs to the bay over time that will arise in response to catchment rehabilitation works has not been well documented (Counihan 2003). In many cases the changes will depend on the residence time of fine material in the tributary channels, which is unknown at present but can influence the loads of incoming sediment and sediment nutrients to the bay. This information will also assist in determining the time-frame in which benefits from remedial catchment action could become apparent.

Observations of significant and persistent erosion in North Arm, despite stabilisation efforts (mangrove and seagrass replanting), suggest a more thorough understanding of physically coupled processes is needed. The additional loading of nutrients associated with coastal erosion is not yet well quantified, and requires time-series wave modelling to be integrated with hydrodynamics and sediment processes.

Similarly, quantifying atmospheric inputs from prevailing emissions and episodic dust storms, ash and smoke has been limited by inadequate observation scales. This has resulted in broad estimates with high uncertainties, contributing to an overall loading budget that may overestimate as catchment fallout may already be accounted for in the calibrated (from instream observations) terrestrial loading calculations. Comprehensive atmospheric modelling can provide a consistent and dynamic means of representing this elusive loading term. The Air Pollution Model (TAPM) combined CSIRO atmospheric models (Hurley *et al.* 2005) and comprehensive EPA air pollution inventories to produce fine-scale dispersal and airshed models (Figure 4.60) (CSIRO 2007). The TAPM outputs show promise, and products may be available for testing as early as 2012.

Climate

Sea-level rise

Waves can contribute to extreme sea levels through wave set-up and run-up. Estimating the contribution of waves to extreme sea levels, which is generally much smaller than that of a storm surge, was beyond the scope of the study by McInnes *et al.* (2009a). Future work should aim to quantify the contribution of waves to sea-level extremes along the Victorian coast. High-resolution bathymetric LIDAR datasets that are being developed as part of the Future Coasts Program will allow these finer-scale studies to be carried out.

In addition to coastal inundation from extreme sea levels, a storm tide may also be accompanied by inundation from rainfall. This additional contribution to inundation, which was not taken into account by McInnes *et al.* (2009a), could increase the area affected. McInnes *et al.* (2007, 2009a,b) also assumed that the topography of the coastewould be constant throughout the 21st century. However, during this time environmental processes, such as the erosion of beaches and soft cliffs, could change the morphology of the shoreline. Superimposed on these environmental processes will be the adaptive responses of society to changes in the shoreline. This might include renourishing beaches to retain the existing coastline, building sea walls along the coast and embankments along watercourses to inhibit erosion and inundation, and infilling low-lying land. The consideration of environmental processes that change the shoreline and adaptive responses should be priority areas of future work.

Rainfall and evaporation

Integrating the bay models with other well-accepted catchment and airshed models will give a holistic picture of regional processes to more accurately represent present and predicted future conditions. Such models are becoming available. A recently commissioned project between CSIRO and EPA (2009–2012) is coupling the air pollutant model (Luhar *et al.* 2008) shown in Figure 4.55 with meteorology generated by decadal model runs for existing (1995–2005), 2030 (2025–2035) and 2070 (2065–2075) conditions (M. Banister pers. comm, EPA). This coupling will generate dispersion footprints of pollutants for existing and future air emissions. The model outputs are likely to include scenarios such as bushfires and controlled burns. The results of the project could be used as inputs to the catchment and marine models.

Figure 4.57 Calibration of modelled salinity in Port Phillip Bay using high-quality time series of water quality records from dedicated moorings. This example shows a 2-year salinity record from the bottom mooring in Hobsons Bay.

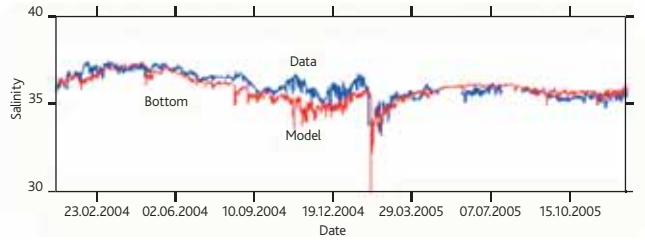


Figure 4.58 WaterCAST catchment model re-analysis by Stewart *et al.* (2010), with error bars showing the comparison with observed data.

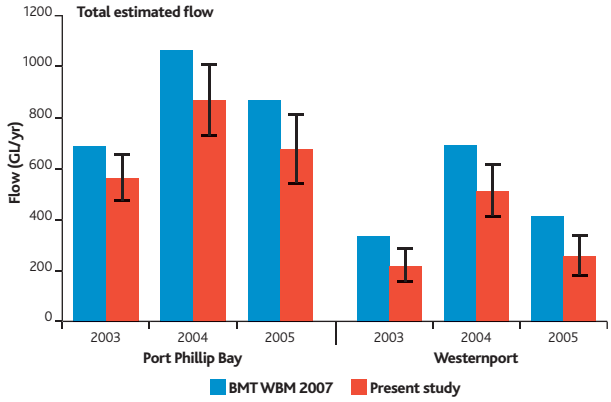


Figure 4.59 WaterCAST catchment model showing good agreement with measurements. (Source: Stewart *et al.* 2010.)

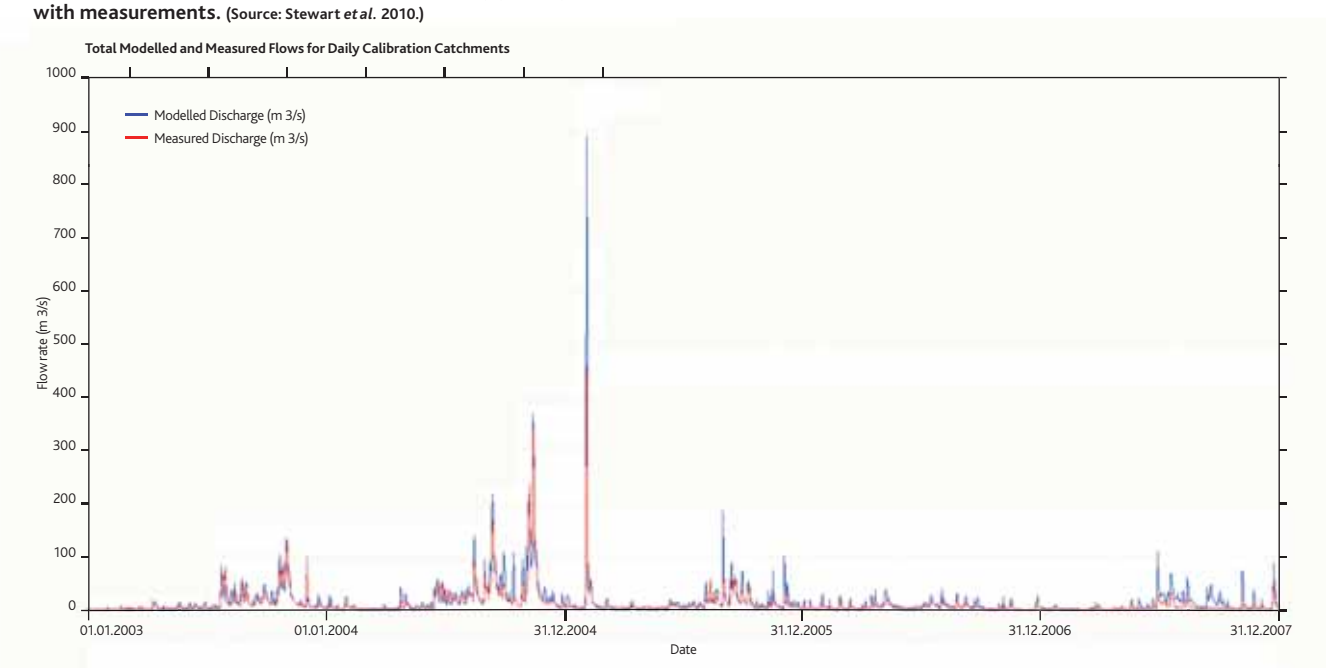
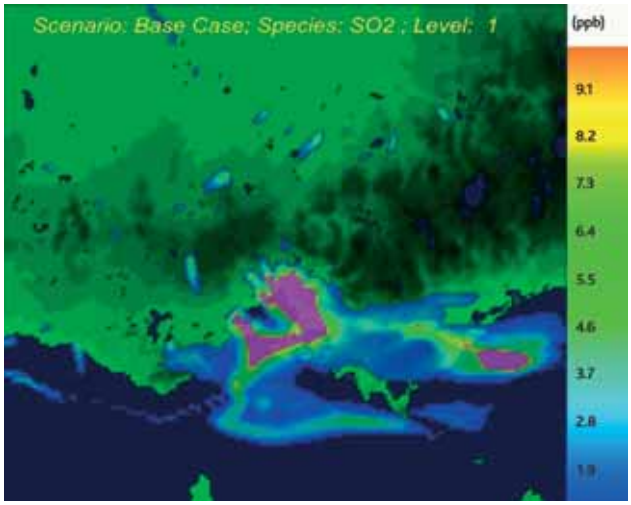


Figure 4.60 CSIRO's The Air Pollution Model (TAPM) result for air pollution plume emanating from Port Phillip Bay, 4 April 2006. (CSIRO 2007.)



5 Water column biota

Greg Jenkins

Photo courtesy Michael Keough.

This chapter summarises the present knowledge of the phytoplankton, zooplankton and jellyfish that, to a large extent, drift passively in the water column (Figure 5.1). The larger, more mobile inhabitants of the water column are reviewed in Chapter 11 (pelagic fish) and Chapter 12 (marine mammals).

Phytoplankton

Phytoplankton are single-celled organisms that drift in the water column and are important for food for larger animals and dissolved oxygen production. They can respond rapidly to changes in environmental conditions, particularly nutrient concentrations and light. Excess nutrients can result in a rapid increase in phytoplankton populations, which may in turn cause a change in water quality (for example increased turbidity and decreased dissolved oxygen concentrations). For this reason the concentration of chlorophyll-*a* pigment, an indicator of phytoplankton abundance, is often included in water column monitoring programs, including water quality monitoring by the Victorian Environment Protection Authority (EPA) in Western Port (Chapter 4). This monitoring has indicated both seasonality (temporal) and enhanced eastern arm (spatial) responses of phytoplankton in Western Port (Chapter 4). Chlorophyll measurements, however, do not provide species-specific information on phytoplankton, although the relative contribution of different chlorophyll pigments can give a broad indication of phytoplankton composition (Chapter 4).

Limited qualitative information on phytoplankton species composition in Western Port is available from Shapiro (1975). In that study, samples were collected from four sites in the northern arm of Western Port (Figure 5.2), but the dimensions and mesh size of the plankton net used for sampling were not reported.

Abundant species were the diatoms *Chaetoceros* spp. and *Ditylum brightwellii*, and a number of other species of diatoms were also common. These common to abundant diatoms occurred at all four sampling sites, with the exception of *Skeletonema costatum*, which did not occur at the most northern sampling site. In contrast to diatoms, the identified dinoflagellates were relatively rare in the samples. Dinoflagellate species generally occurred at the two southern sampling stations, the exception being *Gymnodinium* sp., which only occurred at the two northern sampling stations.

Shapiro (1975) presented more information on the spatial and temporal pattern of relative abundance of the diatom *Ditylum brightwellii*. Between June 1973 and September 1974, this diatom was common to abundant at all four sites in spring and summer, and the relative abundance was greatest at the site near Hastings.

Some qualitative and limited quantitative data are also available on potentially toxic or nuisance phytoplankton species sampled fortnightly between 1999 and 2009 in the Flinders Aquaculture Fisheries Reserve (Anon. 2004, 2005, 2007, 2008, 2010). On each occasion, two phytoplankton samples were collected. The first was a concentrated sample collected by towing a 30 cm diameter, 20 µm mesh plankton net vertically through the top 5 metres of the water column. The second was a one-litre vertically integrated sample taken using a five-metre hosepipe sampler. The net samples were examined in the laboratory for the presence of any known toxic or nuisance phytoplankton. If sufficient numbers of nuisance species were detected in the net sample, the second, 1L hosepipe sample was used for determining the concentration of these species.

In all years, the potentially toxic diatom *Pseudonitzchia* sp. was collected in samples throughout the year, but in low concentrations (Anon. 2004, 2005, 2007, 2008, 2010). The nuisance diatom *Rhizosolenia* cf *chuii* (which can cause a bitter taste in shellfish) was also collected in all years in low concentrations, but generally only in the colder months of late autumn to early spring. The potentially toxic dinoflagellate *Dinophysis* sp. was collected infrequently in some years and in low concentrations. Another potentially toxic dinoflagellate, *Alexandrium* sp., was collected once and in low concentration. During the sampling period the concentrations of potentially toxic or nuisance phytoplankton were not sufficient to trigger regulatory responses.

Zooplankton

Zooplankton are small (mostly microscopic) animals in the water column that drift passively with water currents. They may be holoplankton such as calanoid copepods (Figure 5.1), living their entire life cycle in the water column, or meroplankton such as larval forms, spending only part of the life-cycle in the water column. Zooplankton form a key component in the marine food chain, consuming phytoplankton and providing the major prey items of small pelagic fish such as anchovies and pilchards (see Chapter 11).

Figure 5.1. Scanning electron micrograph of Acartia.
(Photo: David McKinnon.)

Species composition, distribution and abundance

The first study on zooplankton in Western Port (Macreadie 1972) was undertaken from December 1971 to September 1972, a period of low rainfall. The four sampling stations were in the north arm of the bay, from approximately Cowes to Tooradin (Figure 5.2). Samples were collected with Clarke-Bumpus samplers with a 12.15 cm diameter opening and 195 µm mesh nets.

The zooplankton was dominated by calanoid copepods with marine rather than estuarine affinities. The dominant species was *Acartia clausii* — now recognised as two species, *A. tranteri* and *A. fancetti* — which made up over 50% of the total zooplankton numbers, and was most common in the middle reaches of the lower northern arm. The next most abundant copepod species, making up less than 10% of the total zooplankton, was *Paracalanus parvus* (now *P. indicus*), which was most common at the southern site near Cowes. Other copepods of minor abundance included *Pseudodiaptomus cornutus* and *Gladioferens inermis*. *A. tranteri* was much more abundant than *P. indicus* in Western Port compared with Port Phillip Bay. Cladocerans were very rare in Western Port (unlike Port Phillip Bay), with only a few specimens of *Podon intermedius* collected.

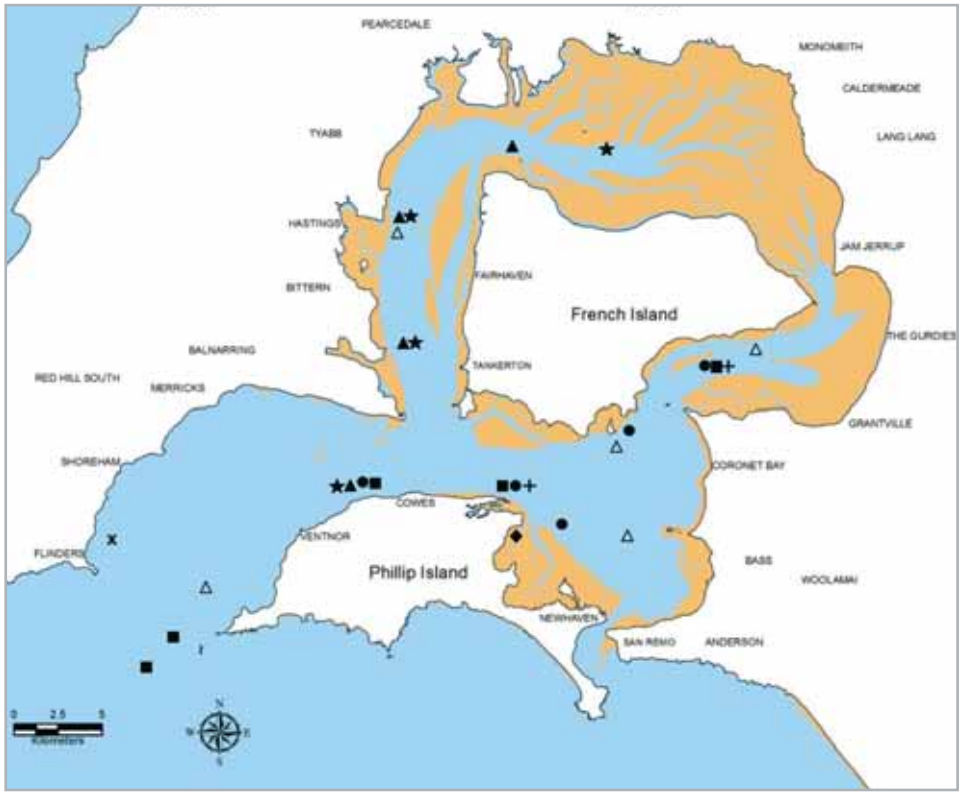


Figure 5.2. Sampling sites used in phytoplankton and zooplankton studies in Western Port.

Immediately following the study by Macreadie there was an expanded sampling program undertaken as part of the Westernport Environmental Study in 1973–74 (Arnott 1974; Shapiro 1975). Samples were taken with oblique tows of a modified WP-2 net with 190 µm mesh. Four sites sampled were the same as for phytoplankton sampling, and three were the same as sampled by Macreadie (Figure 5.2). Rainfall in Victoria during this study was higher than in the period of the Macreadie study, although the rainfall in January and February was relatively low. As Macreadie had found, *Acartia fancetti* was the dominant zooplankton, comprising 75% of the copepods sampled, and *Paracalanus indicus* was the second most abundant species. Apart from the dominance of *A. fancetti*, copepod species were similar to those in Port Phillip Bay, although there was a greater occurrence of oceanic species in Western Port. The differences in zooplankton between the two bays was attributed mainly to Western Port having greater detrital loads in the water column and a greater exchange rate with Bass Strait.

The most intensive period of zooplankton studies in Western Port was the early 1980s. Kimmerer and McKinnon (1985) undertook a comparative study of zooplankton in Western Port and Port Phillip Bay, again during a historically dry period from 1982 to 1983. Core sampling sites used for quantitative analyses are shown in Figure 5.2; additional sites were sampled on an ad hoc basis. Vertical hauls were conducted with a 50 cm diameter opening, 200 µm mesh plankton net. Both bays had a zooplankton fauna distinct from Bass Strait, and the abundances were low compared with temperate bays elsewhere in the world, possibly reflecting low nutrient input (Kimmerer & McKinnon 1985). While the zooplankton in Port Phillip Bay was about half copepods, dominated by *Paracalanus indicus*, the zooplankton in Westernport was dominated by *Acartia fancetti*; and, while cladocerans and larvaceans were very common in Port Phillip they were rare in Western Port, confirming the patterns found by Macreadie. It is possible that the high level of detritus in the water column in Western Port may make feeding conditions unsuitable for cladocerans and larvaceans (Kimmerer & McKinnon 1985). The main difference between the Kimmerer and McKinnon study and the study of Macreadie was Macreadie recorded more of the euryhaline copepod *Gladioferens inermis*, which may have reflected a difference in freshwater inputs (Kimmerer & McKinnon 1985).

Bass Strait species were more often found in Western Port than Port Phillip, but the resident community (i.e. consistently found in the inner region of the bay) of Port Phillip Bay was more similar to that of Bass Strait than to that of Westernport (Kimmerer & McKinnon 1985). *Paracalanus indicus* in Western Port appeared to more of an oceanic invader rather than a resident as in Port Phillip. Many of the zooplankton species showed a seasonal pattern, and most of these were more abundant in summer, possibly reflecting the fact that many species were adapted to warm water (Kimmerer & McKinnon 1985).

In the course of examining samples for the comparative study, it was discovered that three size morphs of *Acartia tranteri* occurred in Western Port (McKinnon *et al*. 1992). These 'large (L)', 'medium (M)' and 'small (S)' morphs had very similar characteristics other than size (McKinnon *et al*. 1992). Allozyme electrophoresis confirmed that there were fixed gene differences between the medium and large morphs, demonstrating that they were separate species. The medium morph was described as a new species, *Acartia fancetti* (McKinnon *et al*. 1992). The sample size was too small to conduct a similar analysis on the 'small' morph, but it is also suspected to be a different species. In Western Port, the large and small morphs of *A. tranteri* were collected in the western entrance in water strongly influenced by Bass Strait, while *A. fancetti* was collected at sites well within Western Port (Figure 5.2).

The reasons for the dominance of *Acartia fancetti* in Western Port, in comparison to high abundances of *Paracalanus indicus* offshore and in Port Phillip, were investigated in laboratory experiments and field sampling (Kimmerer & McKinnon 1989). The laboratory experiments showed that *Acartia fancetti* did not to have an advantage over *P. indicus* in feeding, growth and reproduction in Westernport water, nor was predation by A. fancetti on nauplii of *P. indicus* sufficient to explain the difference in net mortality rates. However, laboratory experiments and field sampling showed that *P. indicus* was preyed on twice as much as *A. fancetti* by visually feeding fish, at least partly because of differences in escape response. Kimmerer and McKinnon hypothesised that visual predation is more important in shallow waters than in deep waters (such as in Port Phillip Bay), resulting in the exclusion of species vulnerable to visual predators. They suggested that the apparent contradiction between this proposed mechanism and the relatively high turbidity in Western Port may relate to the fact that most of the work was carried out in the southern part of the bay.

Marine larvae

The previous section primarily discussed holoplankton, which complete their entire life-cycle in the water column. However the water column is also inhabitated by meroplankton — planktonic stages of species that otherwise live in other habitats — which are primarily the planktonic larvae of marine species. Planktonic larvae can be highly abundant in the water column. For example, in the study by Macreadie (1972), gastropod larvae comprised 25% of the total zooplankton concentration, and crab zoea larvae comprised 4%. Decapod larvae, including crab zoea and larvae of callianassid and carid shrimp, were found to be common in the water column over seagrass-covered mudflats at Crib Point (Robertson & Howard 1978).

Ecological studies on the role of larval supply in determining settlement patterns of the barnacle, *Elminius covertus*, were carried out near Rhyll (Satumanatpan & Keough 2001) (Figure 5.2). A pumping system was used to sample cyprid larvae from different zones in the mangroves and also from different heights above the substratum. A variable supply of

cyprids was found to be important in determining settlement, and was the result of both differences in immersion times between the seaward to landward zones, and also a decrease in the density of cyprids from the seaward to landward zones. Experiments using transplantation of wooden settlement substrata indicated that larval settlement behaviour also had an important role in determining settlement patterns. The vertical pattern of settlement on mangrove pneumatophores, where settlement was greater close to the sediment, could not be explained by larval vertical distribution, which was homogeneous through the water column (Satumanatpan & Keough 2001). A lack of vertical stratification in cyprid densities was also found by Wright (1996) in studies of *E. covertus* at Rhyll jetty.

Fish eggs and larvae (ichthyoplankton) are also an important component of the marine larvae in the water column in Western Port (see Chapter 11).

Growth, mortality and production

Kimmerer and McKinnon (1987a) estimated the growth, mortality and secondary production of the dominant copepod, *Acartia fancetti*, from 1982 to 1984 at one of two sites (Figure 5.2). Growth rate was measured by size fractionation to produce an 'artificial' cohort, followed by field incubation in sealed containers of natural bay water. *A. fancetti* was the dominant zooplankton in the bay, with a mean abundance of all life stages of 12 000 per cubic metre. The mean growth rate of *A. fancetti* was 0.11 per day[5], and temperature and chlorophyll explained 50% of the variance in growth rate, which was usually food-limited. The annual secondary production of *A. fancetti* was less than 1% of phytoplankton primary production, and was quite low compared to Acartia species in bays in other countries (Kimmerer & McKinnon 1987a). *A. fancetti* in Western Port was characterised by low mortality and high longevity. Overall, zooplankton production was considered a minor constituent of the energy budget of the bay.

The data from the comparative study by Kimmerer and McKinnon (1985) were used to estimate net population growth rates of the resident *Acartia fancetti*, *Paracalanus indicus* and *A. tranteri* (L) populations (Kimmerer & McKinnon 1987b). The rates of gain or loss of populations were estimated from horizontal distributions and information on water exchange rates in a simple box model. The resident population of *A. fancetti* was subjected to a median loss to washout of 0.8% per day, but the net population growth needed to offset this loss was easily achievable by the species. The two species from Bass Strait had median rates of gain to the bay, corresponding to negative net population growth or net mortality rates, of 1.5% per day for *A. tranteri* (L) and 3.2% per day for *P. indicus*. Therefore only small differences in net population growth rates between species were sufficient to keep bay and neritic (nearshore coastal) populations separate (Kimmerer & McKinnon 1987b).

Vertical migration

The residency of the *Acartia fancetti* population in Western Port was found to be assisted by vertical migration (Kimmerer & McKinnon 1987c). *A. fancetti* was found to migrate vertically in synchrony with the tides, in a direction (downward on ebb, upward on flood) that reduced losses to mixing out of the bay (Kimmerer & McKinnon 1987c). No other zooplankton species showed vertical migration in synchrony with tides (with the possible exception of *Pseudodiaptomus cornutus*); other resident species may have had alternative strategies to avoid washout, while Bass Strait species may not have experienced selective pressure to do so (Kimmerer & McKinnon 1987c).

Plankton sampling over a mudflat covered by *Zostera* seagrass showed that calanoid copepods (dominated by *Pseudodiaptomus* spp.), decapod larvae and some gammarid amphipod and ostracod species were more abundant in the water column at night than in the day, suggesting vertical migration at night (Robertson & Howard 1978). Diets of planktivorous fish in the same area suggested that migration towards the substratum during the day was most likely a means of predator avoidance (Robertson & Howard 1978).

Further studies on diurnal vertical migration in the demersal copepod, *Pseudodiaptomus* (*P. cornutus* and *P. colefaxi*), were conducted by sampling in a tidal channel near Rhyll (Fancett & Kimmerer 1985). They found that older stages of *Pseudodiaptomus* remained near the bottom by day, rising into the water column at night or on cloudy days. This pattern was most pronounced for ovigerous females, which were also preyed upon most heavily in aquarium experiments by juvenile Yellow-eye Mullet (*Aldrichetta forsteri*), a common visual planktivore (Fancett & Kimmerer 1985). Predation by mullet was also found to be higher on adult Psuedodiaptomus than on *Acartia*. *Pseudodiaptomus* did not feed when on the bottom during the day but experiments showed that this discontinous feeding did not affect egg production, in contrast to *Acartia* where discontinuous feeding did affect egg production. The demersal behaviour of *Pseudodiaptomus* therefore poses no penalty on reproduction while providing a mechanism to avoid visual planktonic predation (Fancett & Kimmerer 1985). Apparent examples of tidal (Kimmerer & McKinnon 1987c) and diurnal (Fancett & Kimmerer 1985) vertical migration in Western Port suggest considerable behavioural plasticity in *Pseudodiaptomus*, as displayed by other members of the genus elsewhere (Kimmerer & McKinnon 1987c).

5 Proportion of body weight increasing per day. Growth rate calculated as the slope of the regression of mean log weight (µg) plotted against time (d) (Kimmerer and McKinnon 1987a).

Photo courtesy Michael Keough.

Jellyfish

The planktonic stage of jellyfish is the free-swimming medusa that generally alternates in the life-cycle with the benthic polyp stage (Richardson *et al.* 2009). Although jellyfish can be classified as part of the zooplankton because they tend to drift with currents, they are in fact strong swimmers and have considerable control of their vertical position in the water column. Predation by jellyfish can have a significant impact on zooplankton, including fish eggs and larvae (Fancett & Jenkins 1988). Jellyfish are thought to be increasing in some areas of the world because of overfishing and other factors such as eutrophication (Richardson *et al.* 2009).

The jellyfish *Catostylus mosaicus* was sampled in Western Port and Port Phillip Bay in April and May 1998 (Hudson & Walker 1998). In Western Port, surveys were carried out in Hastings Inlet, Watsons Inlet, Rutherford Inlet and Bagges Harbour. Abundances were estimated visually from a boat along predefined transects, and samples of individuals were taken with a dip net to measure bell diameter. Unlike Port Phillip Bay, where adult *C. mosaicus* were very common over the sampling period, no adults were observed in Western Port. Small numbers of immature individuals (5 to 20 cm bell diameter) were observed in the Corinella segment (near Jam Jerrup, Figure 5.2) but most of these were in poor condition.

Discussion

There is a significant knowledge gap with regard to the species composition, assemblage structure and ecology of phytoplankton in Western Port. Information on species dominance patterns and how they change spatially and temporally (both within and between years) is completely lacking, as is their behaviour with respect to identified nutrient sources and circulation patterns within the bay. Broader comparisons with phytoplankton community data available from studies in Bass Strait and in Port Phillip Bay are also not possible.

Western Port appears to provide a unique environment for zooplankton by virtue of its shallow depths, high detrital load in the water column, and significant exchange rate with Bass Strait. Its zooplankton is characterised by the dominance of one species, *Acartia fancetti*, which may have an advantage in shallow waters where there are large numbers of visual predators (Kimmerer & McKinnon 1989). Studies also suggest that *Pseudodiaptomus* was dominant over the seagrass-covered mudflats. There is little information on marine invertebrate larvae, and the biology of the larvae of most species is not known.

Another major knowledge gap exists for jellyfish, for which only one brief study on a single species has been undertaken. It is uncertain whether the apparent lack of *Catostylus mosaicus* in Western Port compared to Port Phillip is a real phenomenon or simply an artefact of the conditions that existed at the time of sampling.

6 Western Port as an ecological system

Michael Keough, Diana Walker and Gerry Quinn

Photo courtesy Michael Keough.

Western Port has an extraordinary diversity of habitats, from rocky shores to deep channels with strong currents, mangroves, saltmarshes, seagrass beds, along with intertidal mudflats that are so important to shorebirds and subtidal soft sediments that harbour a diverse invertebrate fauna. Often these habitats are close together, resulting in areas of high diversity, such as the southeastern corner, where we find a diverse reef fauna close to rhodolith beds and important breeding areas for elephant fish. The proximity of these habitats means that they are interdependent.

The geography of Western Port also makes for complex relationships within the bay, and its strong currents move sediments, nutrients and toxicants around, as well as providing a path for plants and animals to disperse. This means, for example, that nutrients entering the bay may be processed and removed in areas distant from where they entered. Some of Western Port's plants and animals also use different parts of Western Port during different stages of their life cycle, or only live part of their lives here. While it is helpful to consider individual assets of Western Port or particular threats, we need to keep in mind the critical linkages within this ecosystem.

Introduction

The geomorphology of Western Port (Chapter 1), along with its broad range of physical processes (Chapter 4) provides a large number of different habitat types within a relatively confined area. These habitats range from reefs typical of the open coasts in the south-west of the bay to sheltered mangroves and mudflats in the north. The large tidal range, particularly in the north of the bay, produces extensive intertidal habitats.

Subtidally there are extensive seagrass meadows, deep channels with strong tidal currents in the north-west and south-east, and unusual steep-walled reefs and rhodolith beds. These habitats are often close to each other, creating a mosaic of habitats. Nowhere is this more striking than in the north-west, where the mangroves and saltmarshes of Yaringa Marine National Park are close to deep channels and the very diverse pinnacle of Crawfish Rock (Figure 6.1).

In the following chapters we describe the main habitat types, with their associated important ecological processes, and the major threats to them. We also consider individual species that are of particular interest, including shorebirds, waterbirds, marine mammals, and fish important to recreational anglers.

These habitats and species cannot be considered in isolation. The mosaic of habitats means that highly mobile species will move between different habitats. Water currents will also move sediments, nutrients and contaminants between habitats. As in other marine environments, most of the animals and plants reproduce by tiny dispersive stages (spores, seeds and larvae) that often spend considerable time being moved about by currents. These stages must pass across other habitat types, through different kinds of water, before reaching another patch of their own habitat. While dispersing they are part of the water column biota, which includes plankton and larger pelagic animals such as fish and jellyfish. Because the water column biota crosses most habitat boundaries, we have considered it separately (Chapter 5).

In addition to the linkages between different components of Western Port, the juxtaposition of different habitat types means that there may be emergent features — areas of Western Port where the overall diversity or natural values may be greater than the sum of the individual habitats.

These considerations depend on treating Western Port as a single ecosystem. This holistic view should be considered when interpreting individual assets, and is considered explicitly in later chapters when we consider ecosystem processes and develop a consolidated set of research needs. Here, we focus on three aspects of this ecosystem view: connectivity of ecosystem components and habitat patches; emergent features of biodiversity; and how models can clarify this ecosystem view.

A fourth, critical aspect of the Western Port ecosystem is the movement of nutrients (and contaminants), and their fate as they move through the water column and sediments and pass into marine waters or the atmosphere. These ecosystem processes, which are critical to the health of most temperate coastal estuaries and embayments, are considered separately in Chapter 14.

Ecosystem linkages and connectivity

The links between different parts of a marine landscape are important, but not generally well understood. They can be of two types: movement across the landscape between patches of the same kind of habitat type (e.g. different seagrass beds), and movement between different habitat types (e.g. movement of seagrass-derived detritus into mangroves). In some cases, movement between habitat types can occur at different stages of an organism's life history (e.g. juvenile vs adult habitats, breeding vs non-breeding habitats).

Movement between comparable habitat patches

Movement across the landscape — larger organisms

Generally, movement across the landscape is associated with larger organisms that are capable of extended locomotion. This might include large fish such as Australian Salmon, which move between seagrass areas and may spend only short periods within individual patches of habitat (Hindell *et al.* 2000b; Hindell *et al.* 2002).

These movements can reflect the larger habitat requirements of animals that must seek patchily distributed or rare food (a likely phenomenon for large predators). Movements may also happen when patches of habitat change or become fragmented, as has occurred with seagrass beds in Port Phillip Bay (Macreadie *et al.* 2009; Macreadie *et al.* 2010).

The risk (e.g. of predation) associated with this movement will vary. It may be a low risk for large organisms but a high risk for small ones or those forced to move.

Measuring movements generally involves tagging or marking individuals, an activity routinely done for birds and marine mammals, and increasingly done for fish, where techniques such as acoustic tags and networks of acoustic detectors allow movements to be followed (e.g., Hindell *et al.* 2008).

Movement across the landscape — dispersive propagules

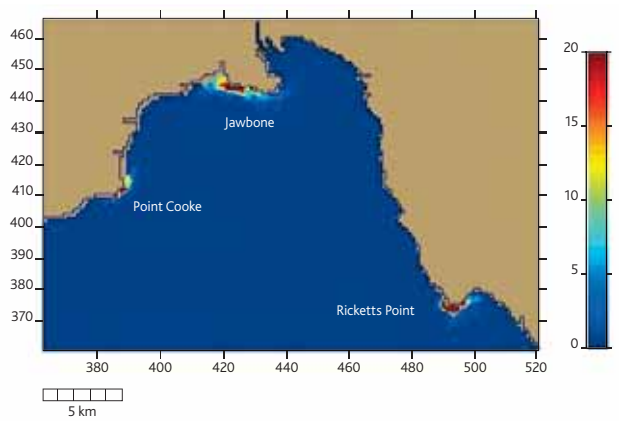
Our understanding of these links is almost always cited as a substantial knowledge gap, whether to justify research funding, a scientific paper, or a statement of uncertainty in an Environmental Effects Statement, such as was done for the desalination project at Wonthaggi, close to Wonthaggi (DSE 2008; IEG 2008). The main reason this knowledge gap persists is that the dispersive stage of marine organisms is generally very small; algal spores, seagrass seeds

Figure 6.1. Habitat mosaic around Yaringa MNP, including mangroves, saltmarsh, intertidal flats and tidal channels.

(e.g. *Zostera* and *Halophila*), and most invertebrate larvae are less than 1 mm long, and even 'large' fish larvae and mangrove and seagrass seeds, are only tens of mm long. This makes them difficult to observe directly in the field, and their size makes it hard to mark them or attach a transmitter or tag. Their size and generally limited mobility also means that they are at the mercy of most currents, so understanding their dispersal requires not only an understanding of large scale currents but also hydrodynamic processes at very fine scales, which increases the difficulty manyfold.

This problem has engaged marine scientists for many years, and we have developed a range of tools to tackle it, ranging from molecular genetics through to oceanographic models (See the broad account in Keough & Swearer 2007.) These tools differ in their spatial and temporal resolution. Genetic tools can show when populations in individual habitat patches are isolated from those in other pieces of habitats, and modern high-resolution techniques can sometimes show when movement occurs, but they can not show how much movement goes on. Mathematical models that incorporate water circulation allow us to track 'virtual' larvae inside the model, describing how a larva with particular characteristics would move from a particular point of origin, and where it would end up (Figure 6.2). Sophisticated models allow us to study how the larvae behave — whether they swim near the surface or near the sea bed, how their behaviour changes from day to night, etc. — and we can make them 'behave' like real larvae. This can give us a better understanding of how and where real larvae move. This has been done in Port Phillip Bay to understand the movements of King George Whiting (Jenkins *et al*. 1999; Jenkins *et al*. 2000), and has been used to predict the fates of larvae produced by animals in marine protected areas (Bathgate 2010). No such work has been attempted for movements within Western Port.

Figure 6.2. Example of the output from modelling of the dispersal of larvae of intertidal snails from marine parks in Port Phillip Bay. The colours indicate numbers of larvae settled 3 days after the release of larvae from each of the three marine parks. **(From Bathgate 2010.)**

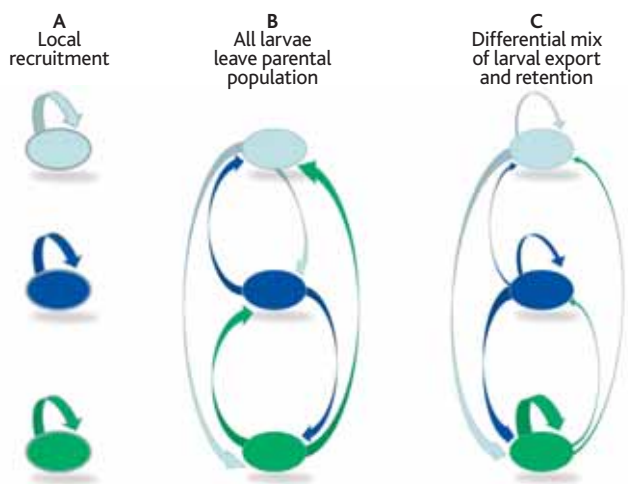


Movement across the landscape has two components: movement within the patches of 'desirable' habitat, and movements across the matrix of unsuitable habitat, where the risks of mortality are higher. These risks might be simple and direct, such as predation for a pipefish moving between patches of seagrass, when it might encounter Australian Salmon while moving, or they might be complex and indirect, such as can be experienced by the larvae of an invertebrate or fish, which are tiny, are likely to suffer massive directly mortality rates in transit, compounded by those surviving arrive in their new habitat in poor condition after a long journey, so that they may struggle to perform well after arrival.

Measuring these connections is difficult, but assessing their importance to populations is even harder.

For individual species these considerations support a metapopulation view, in which each patch of habitat (and the individuals in it) might not be important for the survival of the species, but is part of a group of such patches that is important (Figure 6.3). the demographic connections between patches can be positive, because they allow populations in local habitat patches to be buffered against change; that is, if one local population declines, it can be replenished from nearby populations. Connections can also pose a risk; for example, pathogens may be transmitted readily between well-connected populations, leaving no habitats unaffected. The type of metapopulation varies between species, even those living in the same habitat, and depends on the characteristics of the species' dispersal and its relation with the local habitat. Modelling metapopulations in detail for individual species is complex and requires models of varying complexity, each involving trade-offs between the ease of obtaining an answer and a loss of realism.

Figure 6.3. Examples of metapopulations in which there are three self-contained populations (A), populations in which no larvae are retained and supply other populations (B) to a metapopoulation with a mix of locally retained larvae and dispersers, with populations varying in the importance of these processes (C). Hypothetical local populations are indicated b ellipses, and larval dispersal by arrows, with the numbers of larvae indicated by thickness of arrows. **(Modified from Keough & Swearer 2007.)**



6 Western Port as an ecological system

Photo courtesy Parks Victoria.

Within Western Port, connections between habitat patches are important, and in some cases the circulation patterns discussed in Chapter 4 are a guide to the kinds of connections. It is unlikely, for example, that invertebrates living in small patches of *Zostera* seagrass in Rhyll Inlet will be dispersed by currents to small patches to the north of Flinders. In other cases, hydrodynamics indicate some likely links.

Connections may also be very local, as some species have dispersive stages that do not travel very far. For example, seadragons using the *Amphibolis* seagrass meadows along the south-western edge of Western Port produce young that develop in pouches of the males, emerging as small seadragons a few centimetres long. Their swimming ability is as limited as that of their parents, so (at least initially) they will be in the same small patch of seagrass as those parents, with no dispersal.

There may also be important links to populations outside Western Port, particularly in species with long dispersive periods. For example, most King George Whiting in Western Port and Port Phillip Bay are thought to originate from spawning near the Victoria – South Australia border, and most Snapper caught in Western Port probably did not originate there (see Chapter 10).

Links across habitats

Animal movements

Some organisms move between different habitats at different stages of their life-cycles. This is seen most commonly when species arrive from the plankton into one particular habitat, which has the requirements for tiny juveniles, but then move to different locations as they get larger. In Western Port, examples include species such as King George Whiting and Snapper, which spawn elsewhere, recruit into specific habitats or locations such as seagrass, and move to deeper water later in life. Other fish species might move in and out of estuaries, and others may have particular breeding requirements. For example, small areas in the south-east of Western Port are important breeding sites for Elephant Fish, and large numbers congregate there annually.

Species with broad diets may also feed in a range of habitats. This is particularly true of wide-ranging species such as Australian Salmon, which travels large distances and is piscivorous, taking fish from open, unvegetated areas and also fish associated with seagrass beds (Hindell *et al*. 2000a,b, ; Hindell 2006).

Energy transfers and subsidies

Water currents move organic material and other matter through an aquatic ecosystem. For natural ecosystems this transfer of material is important, and results in 'subsidies' in which energy moves from one habitat to another. These subsidies can be extremely important if that transferred material is needed for the growth or sustenance of organisms. For example, seagrasses grow constantly, and their tips erode and drift away. Often, they die back in winter and the last summer's growth is lost, and sometimes whole plants can be lost. When this happens, the organic material bound in the seagrasses is carried to other areas. It may end up washed up on beaches or trapped in coastal mangroves, where it is broken down and becomes available to plants and animals living there. It may also be carried elsewhere and consumed by herbivores or detritivores. In all these cases the seagrass debris transfers energy and nutrients from one habitat to another. Those nutrients may themselves have been obtained from sediments by the seagrasses, but, more likely, they have been transported into the seagrass bed from another habitat or arrived from estuary discharges.

The transfer of energy between ecosystem components can be via material or through organisms themselves, as matter moves through food chains. These transfers can sometimes be identified, particularly if different habitats have unique chemical signatures. Techniques such as stable isotope analysis have been used to demonstrate the transfer of organic material originating in seagrasses to fish living in open sandy areas (Chapter 11).

Emergent features

The overall marine biodiversity of Western Port is considered high. The only systematic surveys of its fauna and flora, done as part of the Shapiro study, reported over 1350 species of invertebrates alone, a figure 3–4 times that reported from Port Phillip Bay at the time, after completion of the first Port Phillip Bay study. This diversity was spread across a wide range of invertebrate groups. Algae are also diverse, although most likely limited by turbidity in northern sections of the bay. Shapiro and colleagues also reported that approximately two-thirds of Victoria's bird species have been recorded in Western Port and its surrounds (see also Chapter 12).

There have been other large-scale analyses of diversity patterns along the Victorian coast. These have included an analysis of the database of the Marine Research Group, which listed species present, with a focus on a few invertebrate groups (molluscs, decapod crustaceans, echinoderms), and intertidal surveys across a wide range of groups from rocky shores (O'Hara *et al.* 2010). Individually, these studies encompass only subsets of diversity and habitats, and generally involve single snapshots of biodiversity. Despite these cautions, a consistent picture emerges of an overall high diversity for Western Port, even though individual sites may not be more diverse than comparable sites outside Western Port, e.g. Honeysuckle Point vs Mushroom Reef (Handreck & O'Hara 1994).

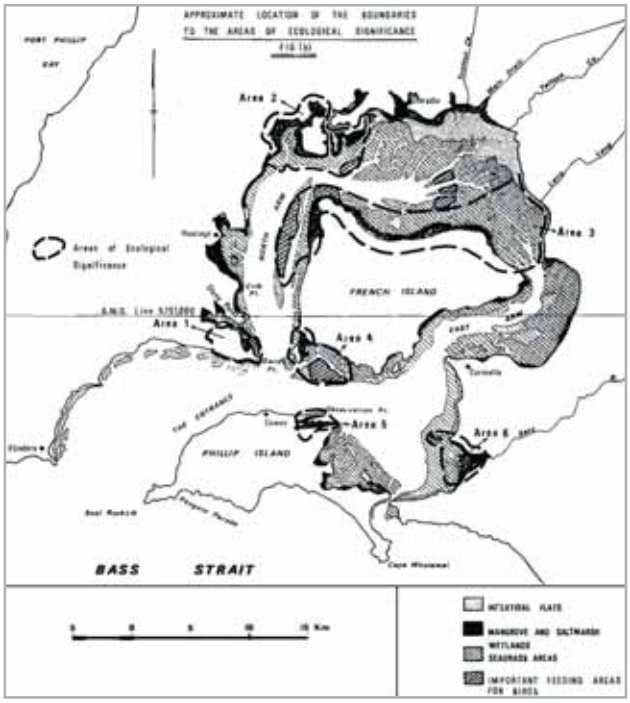
The reasons for this higher diversity compared to Port Phillip Bay are not clear, although one suggestion is that the great diversity of habitats is one contributor. The geological history of Western Port may also be important, giving it affinities with the eastern coast of Australia and a clear past separation from Port Phillip Bay. We note elsewhere (Chapters 7, 11, 13) that this information is dated: there have been no systematic biological surveys for many years, and there is evidence of considerable change in some parts of Western Port (Shepherd *et al.* 2009).

Although our focus in the following chapters is based around the major habitats, the mosaic of habitats means that particular places may have particularly high diversity because they have a combination of features that alone may not be significant, but together are worth highlighting.

The significance of Western Port's habitat mosaics has been recognised for some time; Shapiro (1975) and colleagues identified several such areas (Figure 6.4), including Rhyll Inlet, and the area around Quail Island. One such example not known to Shapiro and colleagues is the area around San Remo, which is important as part of the breeding area for Elephant Fish, for its rhodolith beds, and for its diverse nudibranch assemblage.

These collections of biodiversity are not necessarily functional ecological units, although they are linked in the ways described elsewhere in this chapter and in Chapters 7–13.

Figure 6.4. Areas of ecological significance identified by Shapiro (1975).



Models

At the scale of a whole ecosystem there are complex relationships involving water movements, nutrients, and the organisms living there. The organisms have complex relationships among themselves, ranging from highly specific relationships between some predators and their prey to broader ones in which individual species might alter the habitat in such a way as to cause a cascade of changes.

As relationships become more complex, it becomes harder to predict the consequences of changes to one particular part of the broader ecosystem. Models provide a way of organising and communicating our understanding of a system. They take a variety of forms, and can be used to:

- formalise our understanding of a system — Do we understand the relationships between components? Have we forgotten anything?
- provide a basis for identifying important linkages — If A changes, what else does? If A is an important asset, what does it depend on? Which of the many relationships is most important?
- provide a predictive framework — What will be the state of the system in response to a particular management action? What will be the state of the system under a new set of environmental conditions?

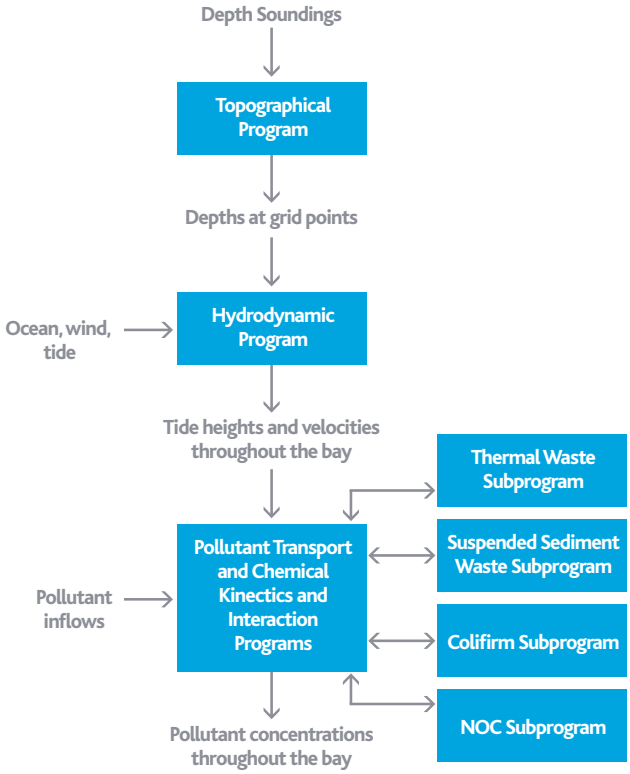
Models can be constructed in several ways. At one extreme, they can be qualitative or conceptual, designed to denote important relationships without any measure of the strength of those relationships. At the other extreme they can be completely quantitative, and include many complex mathematical relationships. For example, the hydrodynamic models described in Chapter 4 incorporate complex models of Western Port circulation, and have as their inputs the results of other complex mathematical models that explain tides, wave patterns on the open coast, winds, and catchment flows. In developing an understanding of a system, the first stage is usually conceptual and can be used as the basis for a more quantitative model. These models are also excellent communication tools. Intermediate semi-quantitative models, in which some of the links are quantitative, can indicate unexpected consequences of management actions, but do not provide quantitative predictions.

Shapiro

A broad conceptual overview was intrinsic to the original study (Shapiro 1975). That study developed a series of coupled Western Port models encompassing land-use patterns and their catchment links, a detailed hydrodynamic model, and a socio-economic model that examined developments around Western Port against a background of natural values, making initial attempts to quantify those natural values. Some of these models were integrated into a water quality model that included hydrodynamics and several geochemical submodels (Figure 6.5).

Although the study did not formally link this model to ecological processes (most likely because of time and funding constraints, and the lack of the powerful computational tools readily available today), these links were considered by the study's authors

Figure 6.5. Water quality model developed by Shapiro (1975), incorporating several submodels. Redrawn from Shapiro.



Coastal CRC/CSIRO

The need for ecosystem models for Western Port has been reinforced in recent publications. The most recent was the Western Port Research Coordination Project, Stage 1, summarised by Counihan *et al*. (2003). That modelling approach consisted of pictorial conceptual models that summarised broad relationships and separate models for the five major divisions (Figure 6.6).

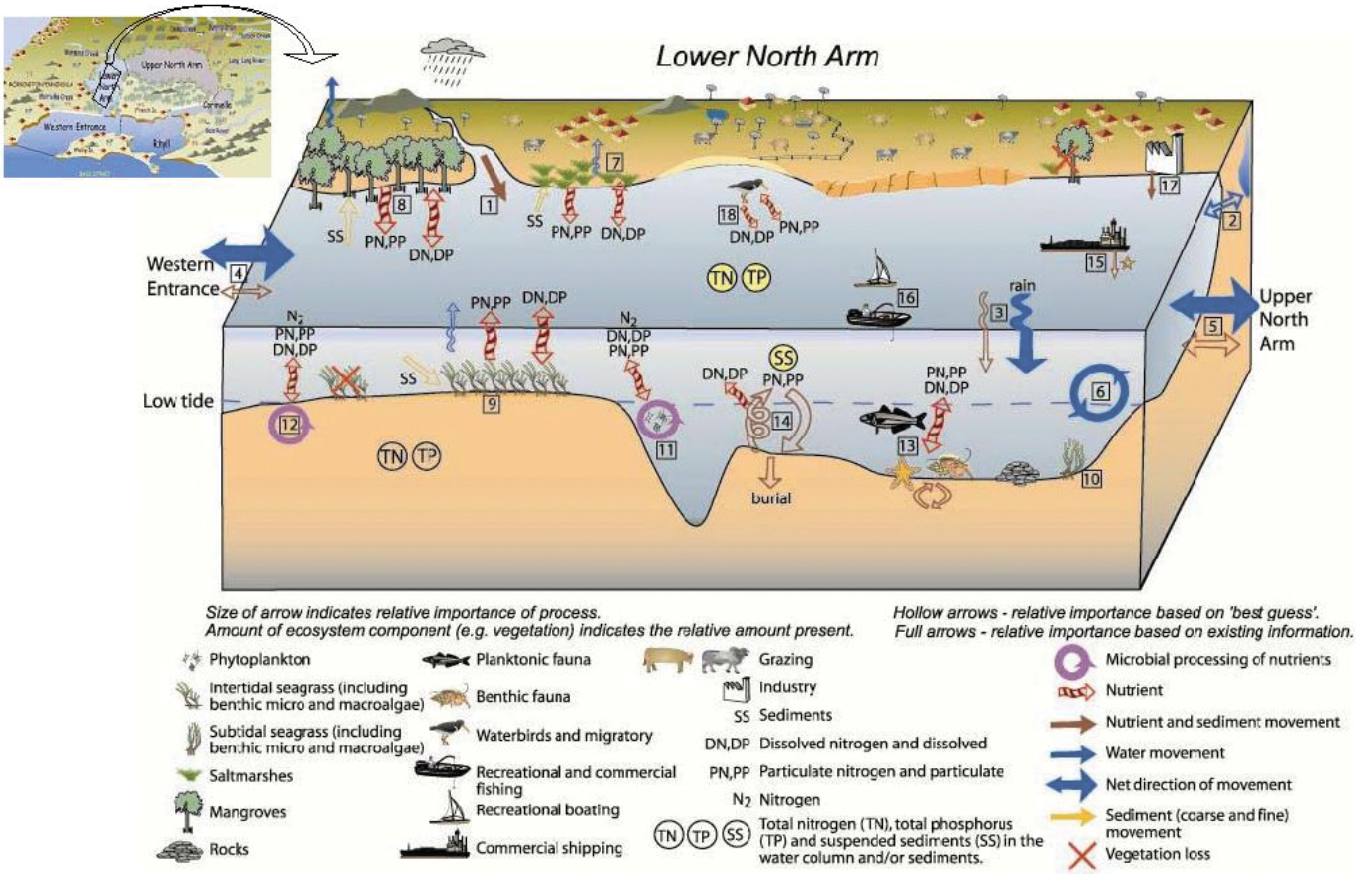
Other

There are many other models in use, such as the Atlantis model developed by Fulton (Fulton & Smith 2004; Fulton 2010) for Port Phillip Bay, which was an extension of the primarily geochemical model developed by CSIRO for Port Phillip Bay. These models have been used elsewhere in Australia. Other groups have developed additional models, such as the Ecosystem Response Models used by the Department of Environment, Climate Change and Water (DECCW) in NSW to assist with the management of estuarine environments (Dr P. Scanes, DECCW, pers. comm.). (See also www.ozcoasts.org.au and particularly the Integration and Application Network developed by the University of Maryland.)

All of these modelling approaches involve a series of trade-offs. It is impossible to represent all of the complexity of a natural ecosystem, so decisions must be made about how much of the ecosystem detail should be included. Less detail means that the models are less realistic and may not capture the response of the full ecosystem, but it also means that the models can be run more easily and can explore a wider range of scenarios. Complex models may require much greater investment in building the model and in accurately characterising the data that need to be fed into it. These decisions are generally made on a case-by-case basis. For example, the Atlantis model for Port Phillip Bay model has limited details about higher trophic levels, particularly individual fish species, whereas its implementation elsewhere includes considerable detail on individual fish and fisheries (Link *et al.* 2010).

Elsewhere in this document we describe some important components of a Western Port model, including hydrodynamics and sediment transport and catchment inputs (Chapter 4) and ecosystem processes (Chapter 14), and in our consolidated priorities we advocate the development of a coupled geochemical model (Howarth *et al.* 2011).

Figure 6.6. Example of pictorial model for major segment of Western Port. (From Counihan *et al.* 2003.)



7 Intertidal and subtidal sediments

Robin Wilson, Sabine Dittmann and Jeff Ross

Photo courtesy Museum Victoria.

Soft sediments are the prevailing habitat in Western Port, covering about two-thirds of the bay. The area of unvegetated sediments has increased following the loss of seagrass beds. Extensive intertidal flats are important foraging grounds for shorebirds. Several hundred species of infaunal and epifaunal organisms have been recorded, including a high diversity of ghost shrimps, brachiopods that are 'living fossils', rare rhodoliths, and other species listed as endangered. The benthic fauna occurs in defined assemblages subject to sediment characteristics and water depth. A depauperate fauna was found at sites with a history of disturbance and eutrophication.

Most of the research on soft sediments in Western Port is 30-40 years old, and a survey to assess the current biodiversity in comparison with past records and adjacent bays is an urgent task. This information could also inform assessments of various disturbances and invasive species. Further areas requiring research attention are the functional roles of benthic organisms and how they contribute to the productivity, sediment dynamics and nutrient fluxes in Western Port.

Sediments of Western Port

Soft sediments comprise the most extensive environment in Western Port. The most recent survey found that 525.5 km^2 (77%) of Western Port marine environment are unvegetated soft sediments (Blake & Ball 2001). Since the 1970s the extent of bare sediments has been increasing at the expense of seagrass beds, which declined by 70% between 1973–1984 (Shepherd *et al.* 1989), although some seagrass recovery occurred up to 2009 (Figure 6.8).

The tidal divide north-east of French Island, which has extensive intertidal mudflats, is of international significance (Rosengren 1984). About 40% of the Western Port area is intertidal mudflats (Edgar *et al.* 1994).

The geomorphology of Western Port (Chapter 1), along with its broad range of physical processes (Chapter 4) provides a large number of different habitat types within a relatively confined area. These habitats range from reefs typical of the open coasts in the south-west of the bay to sheltered mangroves and mudflats in the north. The large tidal range, particularly in the north of the bay, produces extensive intertidal habitats.

History of benthic studies in Western Port

Soft sediment invertebrates in Western Port were first sampled scientifically in the 19th century (Smith *et al.* 1975). Quantitative investigations have been made in the following studies:

- an intensive survey at Crib Point in 1964–65 (Coleman 1976)
- a bay-wide benthic survey in 1973–74 (Coleman *et al.* 1978)
- a bay-wide survey on the crustacean genus Callianassa in 1977 (Coleman & Poore 1980)
- trophic studies by (Robertson 1984; Edgar *et al.* 1994; Edgar & Shaw 1995a, 1995b)
- a monitoring study of the three Marine National Parks in Western Port (Butler & Bird 2010b)
- study of rhodolith beds north of San Remo (Harvey & Bird 2008)
- a description of the Flinders Aquaculture Fisheries Reserve (McKinnon *et al.* 2004).

A number of other studies listed in a report on benthic communities (EPA 1996) were principally on seagrass environments and are not discussed further here.

Following these earlier studies, which focused on species distributions and biodiversity linkages, more recent investigations have addressed species-specific feeding ecology (Boon *et al.* 1997; Stapleton *et al.* 2002), as well as burrow morphology and bioturbation effects of prominent ghost shrimps in sediments throughout Western Port (Bird & Poore 1999; Bird *et al.* 2000).

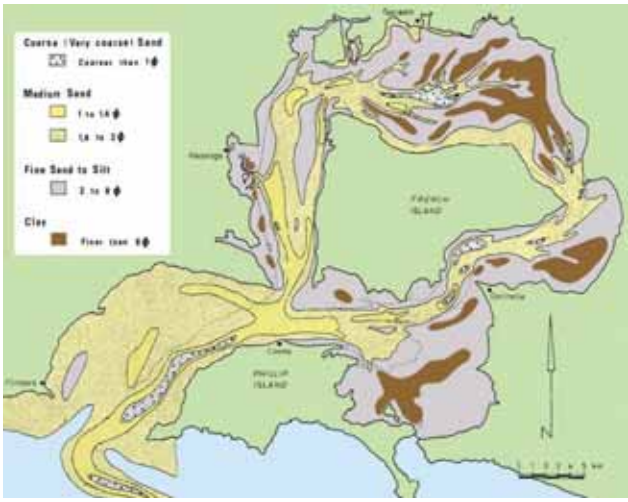
Distribution

Physical description of the sediments

In soft sediments, the strongest influence on the fauna is the physical nature of the sediments. The distribution of sediments around Western Port was first studied by Marsden *et al.* (1979). At that time well-sorted sandy sediments were found in the Western Entrance and in the channels surrounding French Island, and finer muds were found in the North Arm and in the Rhyll and Corinella segments on the eastern side of the bay (Figure 7.1). Edgar *et al.* (1994) suggested that between 1979 and the 1990s muddy sediments were replaced by coarser sands in northern Western Port. Other changes were summarised by (Hancock *et al.* 2001), including three major findings:

- a general trend of increasing sand content in cores from northern sites, representing a span of about 40 years
- the disappearance of clay deposits offshore of Bunyip Drain and Cardinia Creek since 1979
- the appearance of clay-rich deposits offshore from eastern Phillip Island since 1979.

Figure 7.1 Distribution of sediments in Western Port.
(Source: Wallbrink and Hancock 2003, after Marsden *et al.* 1979.)



7 Intertidal and subtidal sediments

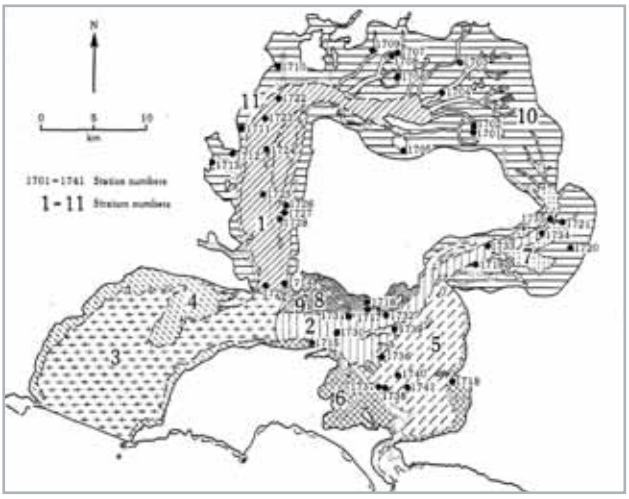
Detailed sediment particle size distributions were presented in Hancock *et al.* (2001), which is the most recent study on Western Port sediments. Most of the terrestrial sediment input to Western Port is to the North Arm via the Bunyip and Lang Lang Rivers, which together account for 69% of the sediment inputs from the Western Port catchments. These sediment inputs are believed to have increased since European settlement (Sargeant 1977), but estimates over that time span are poorly quantified (Wallbrink & Hancock 2003). Sediment input from shoreline erosion is also not well quantified and is a local source of coarser sediments, e.g. at Stockyard Point and Lang Lang, but it is probably a minor input overall (Wallbrink & Hancock 2003). Coarser sands in the Western Entrance Zone are of marine origin. Net movement of water and suspended sediments in Western Port is clockwise around French Island (Wallbrink *et al.* 2003). The export of sediment to Bass Strait has not been quantified (Wallbrink *et al.* 2003). Modelled estimates of sediment deposition rates range from about 0.2 to 0.5 cm/year, with a maximum of 1.6 cm/year for a site in the Corinella segment (Hancock *et al.* 2001).

Distribution of soft sediment assemblages

Sediment infauna

The only comprehensive study of the fauna living within in soft sediments ("infauna") in Western Port is that of Coleman *et al.* (1978). That study took grab samples from 41 randomly located stations (Figure 7.2) and provisionally identified 572 species. However, the species richness was much greater than this number, since half of all crustacean species and the majority of polychaete species could not be named below genus level. Of the polychaetes, crustaceans and molluscs, which together accounted for 93% of the individuals and species, only a few species were abundant and recorded at several sampling stations (Coleman *et al.* 1978). Further benthic studies by Edgar *et al.* (1994) documented a prevalence of polychaete species in unvegetated sediments, whereas crustaceans accounted for more of the species found in seagrass beds.

Figure 7.2 Strata on which sampling for the 1973–74 survey was based. (Source: Coleman *et al.* 1978, Figure 2.)



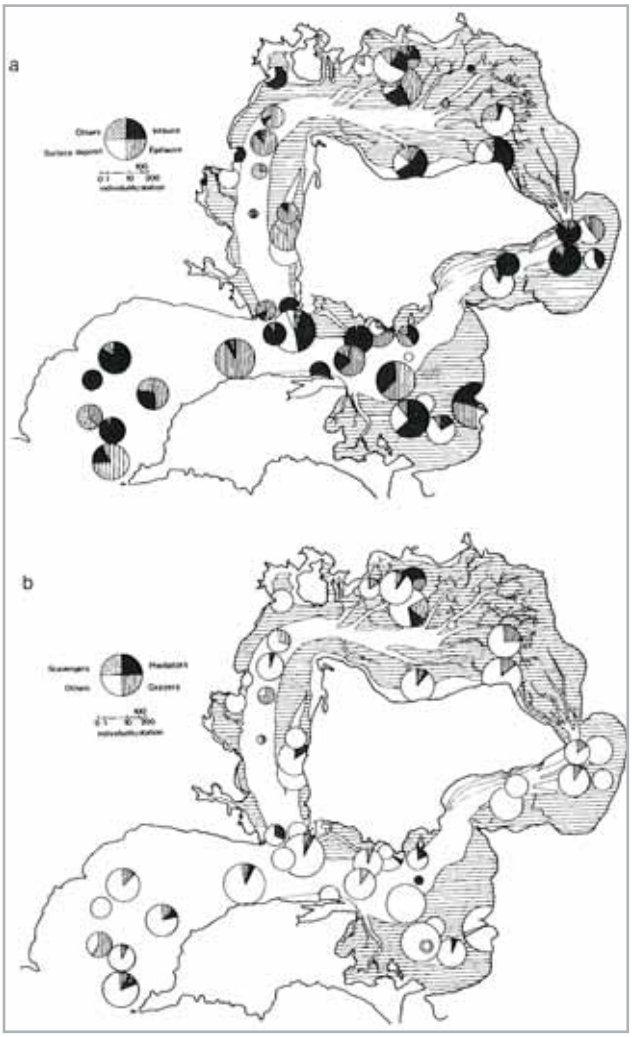
Two major assemblages were identified by Coleman *et al.* (1978), based on multivariate analysis of faunal similarities between stations: a 'clean medium sand' assemblage was found in deeper channel areas, and a 'fine sand and mud' assemblage occurred in intertidal and shallow (< 5.5 m) sublittoral areas. Both of the major assemblages, and the polychaete, mollusc and crustacean species typical of each assemblage, were widely distributed around Western Port Coleman *et al.* (1978). Sediment type and depth were strongly correlated with the two major assemblages. A small group of stations nearest to Hastings was identified as a third assemblage, characterised by a depauperate fauna interpreted as reflecting disturbance (dredging and nutrient input) at Hastings. Within these major assemblages, 13 further groupings of stations could be identified which had a similar fauna and were mostly geographically close stations, referred to as strata. The importance of sediment properties (e.g. mud content, total organic carbon) for explaining patterns in benthic assemblages was further corroborated by Butler & Bird (2010) for the tidal flats in Western Port.

A similar assemblage pattern with depths and sediment properties as defined by Coleman *et al.* (1978) for the entire soft sediment benthos also became apparent by analysing the mollusc data alone, from 96 mollusc taxa, dominated by bivalves Coleman & Cuff (1980). The mollusc fauna from an earlier (1965) intensive study of Crib Point was reported on in a descriptive study by Coleman (1976); in that location the mollusc fauna was dominated by gastropod species. Coleman & Cuff (1980) re-analysed the mollusc data from Coleman *et al.* (1978) using methods less influenced by common species, and found the same assemblages. Their work included an analysis of the distribution of trophic groups of molluscs around Western Port (Figure 7.3). Molluscs that fed on suspended matter or surface deposits (such as *Tellina mariae*) were most abundant, especially in muddier sediments, even though overall mollusc diversity was highest in coarser sediments. Grazing, predatory and scavenging molluscs were less abundant, but accounted together for half of the mollusc species.

Monitoring of benthic communities was conducted in the North Arm for 17 years, commencing in 1972, as part of the environmental requirements for effluent discharge from a cold strip steel mill (Watson 2009). The sampling method in this monitoring was different from that used earlier by Coleman *et al.* (diver-operated airlift samples vs Smith-McIntyre grabs), but the methods were broadly comparable. As in the earlier bay-wide study, there was no attempt at species-level identification (although a reference collection is maintained), but the taxonomic structure of the community was similar: polychaetes and crustaceans comprised 40% and 45% respectively of the taxa present over the monitoring period. Molluscs comprised 10%. Dominant molluscs included large bivalves, particularly *Neotrigonia margaritacea*, *Notocallista diemenensis* and *Sigapatella calyptraeformis*. However, 'Populations of these bivalves declined markedly with increasing water turbidity and deposition of fines on the bed during the seagrass decline period of the 1980s to 1990s.' (Watson 2009).

Rhodolith bed off San Remo. Photo courtesy DPI Queenscliff

Figure 7.3 Distribution of molluscs by feeding mode from the 1973–74 survey. (Source: Coleman & Cuff 1980, Figure 3.)



Epibenthic macroinvertebrates

The observations from previous benthic studies summarised above pertain to the infauna of soft sediments – predominantly small invertebrates that live on and under the sediment surface and are collected by grab or diver suction sampling. There is also a diverse invertebrate epifauna – animals living on the surface of the sediments – in Western Port, comprising large invertebrates rarely collected during infaunal studies but readily observed in studies by scuba divers. Such a study by Watson (2009) reported on epibenthic macroinvertebrates in the North Arm deep channel system where the fauna consisted of small sponges, the ascidians *Pyura stolonifera* and *Stolonica australis*, the seapen *Sarcophyllum* sp. (Figure 7.4), the brachiopod Magellania flavescens and various species of hydroids and tube-dwelling polychaetes. Most elements of this epibenthic fauna require hard objects on the sediment surface for attachment, such as dead and subfossil bivalve shells exposed by water movement in the channels (Coleman *et al.* 1978).

Special features

Marine Protected Areas

Yaringa Marine National Park occupies about 930 hectares about 9 km south-west of Tooradin, adjacent to the Quail Island Nature Conservation Reserve. The Park is part of the Western Port Ramsar site. A private marina adjoins the park. Intertidal and subtidal soft sediments are the dominant environment in the park, although saltmarsh, and mangroves are also present (ECC 2000). Although there are few bare mudflats in the park, they are important foraging habitats for shorebirds and form part of the Ramsar wetlands (See Chapter 12 and references therein). Subtidal sediments contain some 'living fossil' brachiopods and molluscs (see below) (Edmunds *et al.* 2010).

French Island Marine National Park occupies 2700 hectares, about 10 km south of Tooradin on the northern shore of French Island, adjoining French Island National Park. The Park is part of the Western Port Ramsar site. Intertidal soft sediments (both mudflats and sandy beaches) and subtidal soft sediments are well represented in the Park, and extensive areas of seagrass beds, mangroves and saltmarsh are also present (ECC 2000; Edmunds *et al.* 2010). Stations WBES 1701, 1702 and 1705 of the bay-wide study by (Coleman *et al.* 1978) are within French Island Marine National Park; the faunal affinities from that study show these stations to be most similar to one another and to station WBES 1719, directly south on the other side of French Island. The intertidal flats are important foraging grounds for migratory waders (Chapter 12 and references therein).

Churchill Island Marine National Park occupies 675 hectares immediately south of Rhyll, including the entire south-west facing shoreline of Churchill Island. The Park is part of the Western Port Ramsar site. Intertidal soft sediments (beaches and mudflats) and subtidal soft sediments are present in the park, along with saltmarsh, mangroves, seagrass beds and rocky intertidal cobble and shingle shores (ECC 2000; Edmunds *et al.* 2010). The intertidal flats are of national significance as feeding grounds for shorebirds (Chapter 12). None of the stations in the bay-wide study by Coleman *et al.* (1978) are within the park.

These three Marine National Parks, declared in 2002, are representative of the species richness and diversity of macrobenthos occurring in intertidal soft sediments in Western Port (Butler & Bird 2010). Five years after the parks were declared, subtle differences were found in abundances within and outside them, but there was no difference in species diversity. As each of the three parks provides different tidal flat habitats with regards to sediment characteristics, distinct benthic assemblages were found for each park, and no consistent pattern as to differences in assemblages inside and outside (Butler & Bird 2010). Several species occurred predominantly within the parks (*Alpheus richardsoni*, *Paragrapsus* sp., *Paratanaidae* sp., *Armandia* MoV sp. 282, *Musculista senhousia*, and *Phoronopsis albomaculata*), while others, such as the ghost shrimp *Trypea australiensis*, were rarely recorded within the protected areas (Butler & Bird 2010). For further monitoring, Butler & Bird (2010) proposed several key variables, including *T australiensis*, *Biffarius arenosus*, *Macrophthalmus latifrons*, *Lumbrinereis* sp. and also total oxygen concentration and sediment temperature.

Special Management Areas

Bass River Delta Special Management Area occupies 635 hectares on the eastern shore of Western Port at the mouth of Bass River, immediately south of Stony Point. It includes extensive areas of intertidal and shallow subtidal soft sediments, along with vegetated sediments (algae and seagrass) (ECC 2000). Soft sediment communities in this SMA support waders and other waterbirds, as well as commercial and recreational fisheries, the principal target species being King George Whiting and flathead. The Bass River delta has been identified as a nursery area for sharks and whiting. The introduced cord grass Spartina is invading the SMA (ECC 2000). Station WBES 1718 of the bay-wide study by Coleman *et al.* (1978) is within the Bass River Delta Special Management Area. The faunal affinities of that station were closest to two other nearby stations along the eastern shore of Western Port.

Figure 7.4 Undescribed species of seapen Sarcophyllum sp., North Arm channel. (Photo: J.E. Watson, Marine Science and Ecology.)

Figure 7.5 Trypaea australiensis, one of several ghost shrimp species occurring in Western Port. (Photo: M. Marmach, Museum Victoria.)

Rhyll Special Management Area occupies 375 hectares immediately surrounding the township of Rhyll. The Area includes large areas of intertidal and shallow soft sediments (mudflat and a dynamic sand spit) as well as rocky reef, mangroves and saltmarsh, and is used by 32 species of migratory waders (ECC 2000). Station WBES 1715 in the bay-wide study of Coleman *et al.* (1978) is within the Rhyll Special Management Area; that station was found to have low faunal affinities with other stations in the study.

Three other Special Management Areas within Western Port — Honeysuckle Reef, Crawfish Rock and San Remo — include small areas of soft sediment (ECC 2000). However, the habitat that is the focus of these SMAs is rocky reef, so they are not discussed further in this section.

Other sites

Rhodolith beds

Rhodolith beds — fields of mobile, roughly spherical coralline red algae — are known in Victorian waters from only a few localities, including a shallow bed up to 4 m deep in Western Port and much deeper beds in Point Addis Marine Park in Bass Strait (Harvey & Bird 2008). While these are the only published records of rhodoliths in Victoria, it is possible that there are other beds in state waters. The Western Port rhodolith bed is about 1.5 km north-east of Newhaven and 1.5 km north of San Remo (38°30′30.0″S, 145 °22′41.4″E). In many other parts of the world, rhodolith beds are protected for their biodiversity value (Harvey & Bird 2008). Preliminary surveys of the bed revealed it covers an area at least 1 km^2, situated close to and including the main shipping channel. The bed is 1–4 m deep on a broken rhodolith, sand and shell bottom. Four species of rhodolith-forming algae make up the bed: *Hydrolithon rupestre*, *Lithothamnion superpositum*, *Mesophyllum engelhartii* and *Neogoniolithon brassica-florida*. These species also occur elsewhere as non-rhodolith growth forms (Harvey & Bird 2008). The Western Port bed was the first place in Australia where *Mesophyllum engelhartii* was found to form rhodoliths, and the first place where *Hydrolithon rupestre* was found to form rhodoliths world-wide. The majority of rhodoliths were dead; no more than 37% were living, a lower percentage than for beds studied elsewhere, possibly a result of temporary burial or high turbidity from suspended sediments (Harvey & Bird 2008). Rhodoliths are here treated as unvegetated sediments because they are not attached. The cryptofauna of the rhodoliths was investigated by Harvey & Bird (2008), who found that the community was different in taxonomic structure from soft sediment communities elsewhere in Western Port: rhodoliths are dominated by polychaete worms (89% by abundance, with Terebellidae the most common family) with bivalve molluscs next in importance. The cryptofauna did not differ between growth forms of rhodoliths. Harvey & Bird (2008) did not attempt species-level identifications, and they did not sample the communities within the soft sediments under the mobile rhodoliths.

Aquaculture zones

The Flinders Aquaculture Fisheries Reserve occupies 440 hectares over a depth range of 7–11 metres immediately north of Flinders. The reserve includes leases for the growth of abalone and mussels (ECC 2000). The Flinders Aquaculture Fisheries Reserve comprises mostly unvegetated sediments, with some sparse seagrass. (McKinnon *et al.* 2004) recorded 76 provisional species in the reserve, but their identifications were only to family rank.

Summary of current understanding

The following discussion includes some general comments, followed by discussion of environmental assets identified as likely to be of interest to stakeholders and environmental managers responsible for the Western Port marine environment.

Unvegetated sediments in Western Port have increased at the expense of vegetated sediments (specifically seagrass), which has experienced an overall decline during the period 1973–2000 (Chapter 10). Since unvegetated sediments have been found to be less species-rich than vegetated sediments (Edgar *et al.* 1994), the decline of seagrass in Western Port can be expected to be associated with an overall decline in benthic species richness. The contrast is more extreme in intertidal environments where seagrass loss has been most evident: 185 species were identified from unvegetated intertidal sediments by Edgar *et al.* (1994) while the same study found subtidal unvegetated sediments (channels) to include 265 species and were thus closer to the diversity of seagrass communities where 300 species were identified. Besides the presence or absence of seagrass, benthic diversity increased also from the intertidal into the shallow subtidal (Edgar *et al.* 1994).

Western Port has a typical shallow marine 'embayment fauna' in southern Australia (O'Hara & Barmby 2000). The diversity of macrobenthos and sediment properties within and outside the Marine National Parks of Western Port was investigated by Butler & Bird (2010), who concluded that species richness and diversity within the parks was representative of that found throughout Western Port. Overall, Coleman *et al.* (1978) estimated from their own studies and museum records that Western Port contains about 2000 macrofaunal species.

Biota

Crustacea

One of the outstanding characteristics of the soft-sediment fauna of Western Port is the high diversity of ghost shrimps. Decapod shrimps of the genus *Callianassa* were reported in detail by Coleman & Poore (1980; Figure 5). *Callianassa* spp. — now recognised as belonging to several genera (Tudge *et al.* 2000) — were more abundant in unvegetated sediments. As with other common members of the main assemblages distribution of individual species of ghost shrimp was strongly correlated with depth and sediment type. *Callianassa* and related genera are significant bioturbators and their burrowing activity substantially affects sediment properties. They are also sought by anglers as bait, and their extraction using bait

pumps is known to have impacts (see below). *Callianassid* shrimps also contribute to infaunal production in soft sediments (Robertson 1984; Edgar *et al.* 1994).

Two ghost shrimp species occurring off Crib Point have been listed as threatened species under the Victorian *Flora and Fauna Guarantee Act 1888* because of their very restricted distribution: *Paraglypturus (Eucalliax) tooradin*, a rare species otherwise only known from a single specimen from Swan Bay, and *Michelea microphylla*, a local endemic known only from Crib Point (O'Hara & Barmby 2000; Butler & Bird 2010). A third species, *Laomedia healyi*, occurs from Queensland to Victoria and reaches the south-western limit of its range in Western Port (Butler & Bird 2010). Other crustaceans in the bay, such as the rare *Alpheus australosulcatus*, may be endangered but are considered 'data deficient' at present (O'Hara & Barmby 2000).

Amphipoda are another group of crustaceans represented by a very high number of infaunal species in Western Port, which has led to the suggestion that they have undergone adaptive radiation, such as for the 48 species of Phoxocephalidae (Shapiro 1975; Barnard & Drummond 1978).

Polychaeta

Polychaetes constitute most of the species of macrofauna found in sediments, and many species are abundant and are more widespread throughout Western Port (Coleman *et al.* 1978). Prominent polychaetes recorded in benthic surveys are the capitellids *Barantolla lepte* and *Mediomastus* sp., *Lumbrinereis* sp., *Nephtys australiensis*, *Scoloplos* spp., *Isolda* sp. and *Spionida*e (Coleman *et al.* 1978; Edgar *et al.* 1994; Butler & Bird 2010). On sandier foreshores the beach bloodworm *Abarenicola* sp. occurs in small numbers (Smith *et al.* 1975).

Brachiopoda

Brachiopods are much more widespread and diverse in the fossil record than they are as extant fauna, and some living species are apparently little changed from their fossil forms (Richardson 1997) and are often called 'living fossils'. Brachiopods are more diverse and common on calcareous sediments on the continental shelf, such as Bass Strait (Richardson 1981); one species is known in Western Port:

Magellania flavescens is patchily distributed but widespread in Western Port, requiring hard objects on the substrate for attaching its pedicel. It is unknown in Port Phillip Bay (Museum Victoria collections). Although *Magellania flavescens* occurs widely along the southern coast of Australia, the populations in Western Port are the largest known (Chidgey *et al.* 2009).

Mollusca

Mollusca typically comprise only about 10% of the benthic fauna as measured by individual abundances (Coleman 1976; Coleman *et al.* 1978; Chidgey *et al.* 2009), but they contribute greatly to biomass and productivity, with particular species dominating the biomass in different habitats (Edgar *et al.* 1994). The following species are of note for other reasons.

Neotrigonia margaritacea belongs to the superfamily Trigoniacea, a group of bivalves that dominated shallow inshore seas worldwide during the Mesozoic (250–265 mya); there are six Australian members of the group, and *N. margaritacea* occurs widely in south-eastern Australia (Smith *et al.* 1975; Morton 1987). Genetic divergence east and west of the Bass Strait land bridge indicates recent recolonisation from the west into areas like Western Port (Glavinic 2010).

Anadara trapezia is a large and conspicuous ark shell that is prominent in mudflats and seagrass sediments, often in association with *Tellina* spp. (Smith *et al.* 1975; Coleman & Cuff 1980; Edgar *et al.* 1994). It is not particularly abundant but has a large biomass. It is common along Australia's east coast but rare in Victoria, except in Western Port (Smith *et al.* 1975).

Neotrigonia margaritacea, *Notocallista diemenensis* and *Sigapatella calyptraeformis* were dominant in North Arm when monitoring commenced there in 1973, but populations declined markedly with increasing water turbidity and deposition of fine particles during the seagrass decline period of the 1980s to 1990s (Edgar *et al.* 1994; Watson 2009). *N. margaritacea*, *Barnea australasiae* and *Spisula (Notospisula) trigonella* contribute most to the biomass in unvegetated sediments (Edgar *et al.* 1994).

Echinodermata

The brittle star Amphiura triscacantha has been found in northern Western Port; the only other occurrences in Victorian waters are in Corner Inlet (Butler & Bird 2010). It is listed as threatened under the Flora and Fauna Guarantee Act, yet classified only as vulnerable for by the Victorian Department of Sustainability and Environment (Edmunds *et al.* 2010). O'Hara & Barmby (2000) judged that the Western Port population is possibly extinct.

Fish

Fish are treated in detail in Chapter 11, and in Chapter 10 in terms of the seagrass environment on which fish production is primarily dependent (Edgar & Shaw 1995a, 1995b, 1995c). Unvegetated sediments are most likely to be significant to fish communities in Western Port during autumn, when the lower production of crustaceans in their preferred seagrass habitat cannot support fish production (Edgar & Shaw 1995b). Crustaceans were found to be such important dietary items for fish in Western Port that their availability may be limiting fish production, while fish predation may affect the species composition and size structure of benthic assemblages (Edgar & Shaw 1995b).

The meiofauna has not been studied in Western Port, apart from Foraminifera (Bell 1971, cited in Shapiro 1975), and size-specific productivity estimates by (Edgar *et al.* 1994). The latter study revealed an almost continuous relationship between body size and production from meiofaunal to macrofaunal size categories, unlike previous studies in northern temperate areas.

Photo courtesy Michael Keough.

Comparison of Western Port with other Victorian embayments

Soft sediment communities are the dominant environment within Western Port and also in Victoria's other significant embayments, each of which has been the subject of benthic studies of soft sediment communities: Port Phillip Bay (Poore *et al*. 1975; Poore & Rainer 1979; Wilson *et al*. 1998), Corner Inlet (Morgan 1986; O'Hara *et al*. 2002) and Gippsland Lakes (Poore 1982). However, the focus of each of the above studies has been on investigation of patterns within each embayment. Nonetheless (Coleman *et al*. 1978, p. 460) commented: 'Some of the commonest species in Western Port (e.g. *Neotrigonia margaritacea*) are rare or entirely absent in Port Phillip Bay and vice versa. Western Port also has a greater species diversity, average values of the Shannon–Weaver diversity index (H[]) per station being 2.93 in Western Port and 2.36 in Port Phillip Bay.'

(Kott 1976) compared the ascidian faunas of the two bays and found that 26 species occurred in Port Phillip Bay but not in Western Port, while 45 species were found in Western Port but were absent from Port Phillip Bay. The Western Port ascidian fauna was also more diverse and contained a high percentage of species at the southern limit of their range. However, the Western Port ascidian collections studied by Kott were taken from both reef and soft sediments, whereas the Port Phillip collections were largely from soft sediments alone.

Anecdotal evidence and observations of individual species suggests that the soft sediment benthos of Western Port also has distinct faunal elements. Species that are present in Western Port but are absent from Port Phillip Bay include the bivalve mollusc *Neotrigonia margaritacea* and the brachiopod *Magellania flavescens*, which both occur in Bass Strait and elsewhere in eastern Australia (Darragh 1986, Museum Victoria unpubl. data).

Other than the above anecdotal comments and the inconclusive study by (Kott 1976), and the difference in zooplankton communities of Port Phillip Bay and Western Port reported by Kimmerer and McKinnon (see Chapter 5) the question 'How distinct is the benthic fauna of Western Port?' has not been the subject of any quantitative analysis (nor has the question been addressed for the other major Victorian embayments). Progress with understanding the taxonomy of benthic fauna, especially of crustaceans and polychaetes, means that some limitations of the data collected by Coleman *et al*. (1978) could now be addressed, and all samples are still available in the Museum Victoria collection. A re-examination of specimens of selected taxa from this material would be required if a second benthic study of Western Port were to be conducted.

Spatial variation in soft sediment assemblages within Western Port

Spatial patterns of distribution of soft sediment communities are well understood (Coleman 1976; Coleman *et al*. 1978; Coleman & Cuff 1980a; Coleman & Poore 1980) and discussed above (see 'Distribution of soft sediment assemblages'). In summary, they are as follows:

- Two assemblages of species can be recognised, one from well-sorted sediments > 5.5 m depth, mainly in channels, the other from intertidal and shallow fine sand and mud < 5.5 m depth.
- Common species in each assemblage are widely distributed.
- The distribution of well-studied species is typically strongly correlated with depth and sediment type.
- A greater similarity of assemblages at stations reflects geographic proximity; depauperate communities typify disturbed sites.

Temporal changes in soft sediment communities of Western Port

Few of the studies of soft sediment communities in Western Port had a temporal component, and each of the studies listed above had other goals and sampled different sites with different methods. There is thus little information on change over time in the composition and distribution of components of the soft sediment community. Shapiro (1975) mentioned sampling between 1966 and 1970 that revealed considerable variation in species numbers and abundances between seasons and years. Edgar *et al*. (1994)

resampled some of the areas sampled by Coleman *et al.* (1978) two decades later, and found large changes in the presence and abundance of several species from both seagrass and unvegetated sediments, including a decline in the abundance of the bivalves *Notocallista diemensis* and *Katelysia rhytiphora*. Temporal variation can be impact-related or reflect natural variability; for example, the recruitment of many benthic macroinvertebrates occurs in winter and spring (Edgar *et al.* 1994). Butler & Bird (2010) also found differences in benthic abundances between their two consecutive study years and identified a subset of physical and biological variables that could be used to monitor change over time in benthic studies. To date there has been no monitoring using the variables identified by Butler and Bird (2010).

Productivity

The annual macrofaunal production in Westernport was lower in unvegetated soft sediments (57.3 g/m2, 3.3 g of which was epifaunal) than in seagrass beds, where epifaunal production was high (17.2 out of a total production of 79.2 g/m2; Edgar *et al.* 1994). Yet infaunal biomass and production was higher than of epifauna at most sites (Edgar *et al.* 1994). The planktonic-to-benthic ratios of unvegetated sediments ranged between 2.5 and 3, similar to the range reported in the literature, including an earlier study at Crib Point (Robertson 1984; Edgar *et al.* 1994).

Bioturbating infauna

Several species of ghost shrimp occur in Western Port (Coleman & Poore 1980). Ghost shrimps are ecosystem engineers, with burrows stretching > 50 cm into the sediment and forming complex underground structures with branching tunnels (Bird & Poore 1999; Bird *et al.* 2000; Butler & Bird 2007). These physical structures and bioirrigation by the ghost shrimps create a deeper extension of aerobic surface sediment conditions, facilitating aerobic microbial activities (Bird *et al.* 2000). When ghost shrimps were removed by experimental bait pumping, the sediment became more anaerobic and muddier, with less porosity and organic matter (Contessa & Bird 2004). Ghost shrimp densities were significantly reduced by bait pumping and recovered slowly (Contessa & Bird 2004). Ecosystem processes relating to the bioturbation are addressed further in Chapter 14.

The co-occurrence of species with seemingly similar ecological roles raises questions about niche differentiation. Although species-specific distributions based on sediment properties and water depth have been found (Coleman & Poore 1980), coexistence has also been reported, even to the extent of burrow sharing (Butler & Bird 2007).
The two most studied species, *Biffarius arenosus* and *Trypea australiensis*, appear to differ in their feeding behaviour and preferred particle size, allowing coexistence (Stapleton *et al.* 2001). Competition is further reduced by diet ranges; stable isotope studies have shown that *B. arenosus* feeds on a diet of detritus derived from seagrass and its epiphytes, while *T. australiensis* can also incorporate further algal matter (Boon *et al.* 1997). However, when ghost shrimp species co-occur, one may become dominant (Coleman *et al.* 1978).

Major threats

Major threats to soft sediment ecosystems are related to habitat loss or modification, or to directly effects on the biota from unnatural inputs (e.g. toxicants) or extractions (e.g. bait-pumping). Altough the threats are addressed individually here, they are often concurrent (Thrush *et al.* 2008b). Our evaluation of the threats relies largely on studies carried out elsewhere.

Water and sediment quality

Risks

Further deterioration of water or sediment quality will increase the risks to soft sediment biota. Community structure will be affected and species richness is likely to be depressed wherever extensive nutrient or pollutant input occurs, as has already been demonstrated locally at Hastings (Coleman *et al.* 1978) and in other embayments (Pearson & Rosenberg 1978; Warwick *et al.* 1987; Jackson 2008). Nitrogen and phosphorous, as well as pesticides, enter Western Port, although loads were lower than neighbouring Port Phillip Bay in the 1970s, apart from some more intensive localised input (Shapiro 1975). Individual species and communities may be at risk, especially those that have local distributions within Western Port, such as the rhodolith bed north of San Remo, and the localised ghost shrimp species currently only known from near Crib Point.

Consequences

Nutrients

Chapter 14 discusses eutrophication in detail. The loss of larger, deeper-dwelling benthic organisms with increased eutrophication reduces bioturbation, further accelerating degradation of sedimentary conditions (Pearson & Rosenberg 1978; Gray *et al.* 2002). A dominance of capitellid polychaetes is typical of such disturbed environments; sibling species of *Capitella* spp. are world-wide indicators for nutrient enrichments in sediments (Tsutsumi 1990; Chareonpanich *et al.* 1994; Ramskov & Forbes 2008). In extreme cases of eutrophication, only microbial mats remain on the sediment surface (Pearson & Rosenberg 1978; Jones & Pinn 2006).

Sediment input and resuspension

The effects of sediment input depend on the type of sediment (e.g. marine, terrigenous) the grain size composition, and whether a gradual or pulsed addition of sediments occurs (Miller *et al.* 2002; Widdows & Brinsley 2002; Thrush *et al.* 2004). Smothering is possible if large deposits occur in a short time so that organisms (e.g. suspension-feeding bivalves) cannot maintain a connection to surface (Newell *et al.* 1998; Norkko *et al.* 2002). Resuspension of fine material can also remobilise pollutants accumulated in sediments, increase turbidity and thus reduce productivity of the microphytobenthos, and inhibit the filtration efficiency of suspension feeders. Gradual changes in sediment properties are likely to cause a shift in assemblage structure (Miller *et al.* 2002; Lohrer *et al.* 2004; Thrush *et al.* 2004).

In Western Port, the sedimentary environment has already seen major changes over the decades following terrestrial inputs, shoreline erosion and loss of seagrass beds (Hancock *et al.* 2001; Wallbrink & Hancock 2003), and Edgar *et al.* (1994) relate some of the changes in benthic assemblages between 1970 and the 1990s to these modified sediment conditions. Changes in sediment quality and associated biota can have wide-ranging effects. The relative abundances of dominant taxa in soft sediment communities is polychaetes > crustacea > molluscs at most sites (Coleman *et al.* 1978), but the relative importance of the same taxa in fish diets is crustacea >> polychaetes + molluscs (Edgar & Shaw 1995a, 1995b, 1995c). Thus, an increase in sedimentation rate that is sufficient to transform vegetated sediments, or rocky reefs, into unvegetated sediment is likely to have a negative impact on fish production in Western Port.

The rhodolith bed north of San Remo is likely to be particularly vulnerable to sedimentation and increased turbidity. It is possible that this community (one of only two rhodolith beds known in Victoria, and the only one in a shallow embayment) is already impacted by sedimentation, since (unlike other known beds) most rhodoliths in the San Remo bed were dead, with living rhodoliths make up only 37% of the bed, possibly because of temporary burial or increased turbidity from suspended sediments (Harvey & Bird 2008).

Heavy metals, TBT, toxicants, pathogens and pesticides

Toxicants can enter soft sediments by source input and deposition as well as resuspension of polluted sediments. Heavy metal concentrations in Western Port can be locally high, and their effect on benthos is species specific (Ahsanullah 1976; Ahsanullah *et al.* 1980). The amphipod *Allorchestes compressa* was more sensitive to Cadmium and Zinc than the polychaete *Neanthes vaalii* or the crab *Paragrapsus gaimardii* (Ahsanullah 1976). With species specific threshold existing and ecotoxicological studies showing tolerances as well as toxic effects, generalisation of heavy metal effects on soft-sediment benthos cannot be made (King *et al.* 2004; King *et al.* 2005; King *et al.* 2006). Any bioaccumulation of heavy metals in benthic organisms will affect assemblages and food webs, as benthic organisms are important prey for higher trophic levels (Stark 1998; Reish *et al.* 1999; Waring *et al.* 2006).

Species-specific responses to tributyl tin (TBT) contamination are also known (Reish *et al.* 1999). Among gammaridean amphipods, some Haustoriidae have a rapid uptake of TBT, leading to a quick death, while Phoxocephalidae had a slower uptake and show sublethal effects, including changes in their burrowing behaviour (Meador *et al.* 1993). Amphipods of these families are prominent in Western Port sediments, yet no species-specific toxicity studies have been carried out on them. The muricid snail *Lepsiella vinosa* has been severely affected by imposex in the vicinity of major ports in South Australia, and snails found in Western Port (on rocky shores) showed the phenomenon as well, with further laboratory experiments indicating that not only TBT but heavy metals and environmental stress contribute to this condition (Nias *et al.* 1993).

Organics (oil)

Oil pollution can be chronic or the result of a particular spill, and soft-sediment benthos may be affected by the oil as well as dispersants used to clear spills. Oil can penetrate into the sediment, where it can accumulate for a long time, and it can also seal the surface and prevent any exchange of oxygen or dissolved nutrients between the sediment and the water column (Kuiper *et al.* 1984; Peterson *et al.* 2003). Some bacterial breakdown of oil in sediments is possible, but this has not been investigated in any detail.

Salinity

The salinity in much of Western Port is close to fully marine, and the benthic fauna is typical of a southern Australian embayment, with truly estuarine species restricted largely to river mouths, especially Bunyip River and Lang Lang River (Coleman *et al.* 1978). Salinity is lowered during floods, whereas hypersaline conditions are associated with depressed rainfall and rising watertables in the catchment. The majority of the Western Port benthic fauna is thus probably exposed to salinity changes, so that both prolonged lower salinities or hypersaline conditions may alter the abundance or even the presence of some species. Changes in community structure, and possibly ecosystem function, are expected consequences of any significant change in salinity (Rosenberg & Möller 1979; Chainho *et al.* 2006).

Acidity

Soft sediments can become acidic if hydrodynamic patterns in wetlands have been modified, allowing the build up of organic matter in sediments that are exposed during drought. Oxidation in acid sulfate soils can lead to a significant decrease in pH, and rewetting can release heavy metals and acidity into the overlying water (Simpson *et al.* 2008). Soft sediment fauna thus may be subject to simultaneous changes in water saturation, salinity, acidity, hypoxia and heavy metal concentrations. A lower water pH affects calcification rates, causing shells of molluscs to become brittle or dissolve, and most other non-calcifying benthic macroinvertebrates are killed by a pH below 5 (Knutzen 1981; Corfield 2000). Changes in water pH from acid sulfate soils are more severe than those predicted from ocean acidification resulting from global warming (Doney *et al.* 2001). DPI (2003) considered the risk from acid sulfate soils in Western Port to be minimal.

Extraction and disturbance

Risks

Dredging and spoil disposal

Dredging has occurred in Western Port since the 1920s (Wallbrink & Hancock 2003), for port development, maintenance and deepening of harbours and shipping channels, and until 2000 for commercial fishing (see Chapter 11).

Consequences

Excavations lead to habitat loss and the destruction or burial of organisms and biogenic structures (e.g. rhodolith beds, brachiopod beds, ascidian clumps), which in turn affect biodiversity and community structure. Predicted consequences include increases in populations of scavengers as a result of dead or damaged organisms in the dredge path (Newell *et al.* 1998). Recovery can take years, and a return to the previous assemblage is not always accomplished (Thrush & Dayton 2002). The effects also depend on the scale and frequency of dredging operations.

Dredging also causes remobilisation of sediment and can increase turbidity. In addition, spoil disposal can effect benthic organisms by turbidity and smothering, similar to sediment effects described above. A monitoring program in the North Arm from 1972 to 2009 showed that, although there was no extinction of species, the 1980s – 1990s sedimentation in the North Arm channel bed resulted in decline of populations of infauna and epibenthos, and also found dredging impacts on seagrass and pier pile faunas (Watson 2009).

Bait collection (intertidal sediments)

A six-month study by Contessa and Bird (2004) found that bait pumping for ghost shrimp (Thalassinidae) caused sediment disturbance, which has been shown to reduce sediment porosity, increase the proportion of fine particles in sediments, increase surface algae and decrease organic carbon. The recovery of ghost shrimp populations was slow during the six months of the study. Wider impacts on benthic communities are to be expected if densities of the ecosystem engineering ghost shrimps change (Coleman & Williams 2002; Skilleter *et al.* 2005). In Western Port, a ban on bait pumping is suspected to have resulted in higher abundances of polychaetes within the Marine National Parks (Butler & Bird 2010).

Habitat loss and fragmentation

Risks

The quality of soft sediment habitats in Western Port is affected by drainage inputs and shoreline erosion from mangrove loss, and land reclamation leads to irreversible habitat destruction (Wallbrink & Hancock 2003). Seagrass decline has contributed to an increase in unvegetated sediments in Western Port (Chapter 10).

Consequences

Given the extent of sedimentary habitats in Western Port, the loss of this broad habitat category is unlikely. But on a smaller scale, habitat loss may occur if particular biogenic structures are destroyed. This will have consequences similar to those for sediment quality and extraction and disturbance. Urban and industrial development may result in the loss of muddier high intertidal sites, which can be a refuge for juveniles of benthic organisms and important for certain stages of their life cycle, and fragmentation could affect species that have a limited dispersal capability (Thrush *et al.* 2008a). Most benthic organisms have pelagic larvae and many are also able to enter the water column as adults, so the effects of fragmentation have to be considered in a wider context of habitat mosaic and connectivity (Eggleston *et al.* 1999).

Sea-level rise, temperature increase, UVB

Risks

The predicted rise in sea levels is undeniably a consequence of human-induced emissions that are causing climate change (Domingues *et al.* 2008). In coastal areas, ultraviolet radiation and sea temperature changes driven by climate change are also predicted to affect large areas (Halpern *et al.* 2008). For embayments and other coastal systems, predictions may have to be adjusted for site-specific geomorphology and site history, as broad-scale climate patterns can be modified by smaller-scale weather variability (Hewitt & Thrush 2009).

Consequences

Sea-level rise will change the ratio of intertidal to subtidal areas in Western Port. As intertidal areas are important foraging grounds for migratory shorebirds, their landward retreat would need to be facilitated to prevent increasing loss of intertidal area.

Higher temperatures could change the distribution patterns of species and facilitate the establishment of species from warmer regions, including invasive species from tropical or subtropical origins (see below). Temperature changes could also affect the annual recruitment pattern of benthic species, but baseline information in this regard is missing for southern temperate regions.

Some benthic organisms, e.g. nemerteans, can be UV-sensitive and are only active at night (Nordhausen 1988), but some others can prevent damage from UVB by producing pigments (Dahms & Lee 2010).

Marine pests

Risks

Western Port, like all harbours, is exposed to the risk of accidental introduction of non-native marine species through commercial and recreational shipping, as hull-fouling organisms and organisms in ballast water. Any escalation of shipping activity will increase the risk of such introductions.

Consequences

The high diversity of benthos in Western Port is unlikely to protect it from the establishment of exotic species. Eighteen exotic species have been recorded in Western Port, of which 12 were considered likely to have established self-sustaining populations — three ascidians *Ascidiella aspersa*, *Ciona intestinalis*, *Styela clava*, *Styela plicata*, two bryozoans *Bugula neritina* and *Watersipora subtorquata*, a crab *Carcinus maenas*, two algae *Codium fragile tomentosoides* and *Ulva lactuca*, two bivalves *Musculista senhousia* and *Theora lubrica*, and a toxic dinoflagellate *Alexandrium tamarense* (Parry & Cohen 2001). However, a monitoring program in the North Arm from 1972 to 2009 did not find any of the large introduced epibenthic species that are common in Port Phillip (Watson 2009). Four other exotic species are present in Western Port but not as self-sustaining populations: two bivalves *Corbula gibba* and *Crassostrea gigas*, a polychaete *Sabella spallanzanii* and a kelp *Undaria pinnatifida* (Parry &

Cohen 2001). To that list must be added the Pie Crust Crab *Cancer novaezelandiae*, native to New Zealand but known from a few specimens collected at Flinders (Museum Victoria unpubl. data). Ecological impacts from these species in Western Port are likely but have not been demonstrated.

In Port Phillip Bay recent research has failed to demonstrate any impacts on community structure because of the presence of introduced suspension feeders, at least at naturally occurring densities of those suspension feeders (Ross *et al*. 2007), although clumps of *Sabella*, which are common, affect macrofauna and geochemistry (J. Ross, M. Keough & A. Longmore, unpublished data). Nevertheless, depending on which species may arrive, changes to entire communities and their functions cannot be excluded.

Cumulative impacts

Cumulative effects of the various threats to soft sediment habitats and biota in Western Port are very likely, as several stressors are usually acting at once (Thrush *et al*. 2008b). At present the interaction and possible intensification of multiple stressors on benthic assemblages are not well understood, and temporal and spatial scales of effects of single versus multiple impacts remain to be evaluated (Thrush *et al*. 1999). Furthermore, any impacts on soft sediment organisms will affect higher trophic levels (Chapters 11 and 12) and reverberate with functional implications in the bay (Chapter 14).

Research to fill key knowledge gaps

To mitigate the threats outlined in the previous section, improvements are needed to the water quality.

The evaluation of threats in this report relies heavily on studies carried out elsewhere in Australia or overseas, and the validity of transferring that insight to Western Port, with its extensive sedimentary environments and high benthic biodiversity, is uncertain. Furthermore, the assessment in this chapter is largely based on studies carried out in Western Port several decades ago, and no recent evaluation of the full biodiversity in soft sediments has been made. Other areas in southern Australia, such as Port Phillip Bay, have seen major changes in species composition and dominance, including the prevalence of introduced species (Wilson *et al*. 1998), yet for Western Port no attempt has been made to identify the current biodiversity and compare it with previous records or nearby embayments. If Western Port proves to be as different in its macroinvertebrate biota compared to adjacent bays at is appears, it must be given special attention.

Based on the studies from the 1970s, a high diversity on species level exists in Western Port (Barnard & Drummond 1978; Coleman *et al*. 1978), more akin to tropical. Both evolutionary backgrounds as well as ecological implications of the diversity need to be understood to properly address conservation measures for the soft sediment biota of Western Port.

A further step needed to inform future threat mitigation is habitat mapping to identify areas of functional importance, based for example on the occurrence of ecosystem engineering species. Knowledge on the occurrence, density and spatial and temporal variations of biogenic structures, together with accompanying studies on associated ecosystem processes, would allow us to prepare targeted mitigation measures and reduce the effects of threats outlined above. Such insights could also allow modelling of possible effects and efficiency of mitigation measures.

With a range of environmental changes and threats occurring simultaneously, knowledge on tolerance ranges of benthic species will further support mitigation, especially towards effects of single and combined stressors.

To evaluate the benthic biodiversity in unvegetated sediments of Western Port in comparison to earlier surveys, a historical repetition of studies by Coleman *et al*. (1978) and Edgar *et al*. (1994) should be carried out; that is, revisiting the same sites and applying the same methodology. This would provide knowledge on benthic diversity and assemblages, which can be compared on various spatial (within Western Port, with Port Phillip Bay) and temporal scales (over about two and four decades). Given the environmental changes that have occurred in Western Port over those time frames and the further seagrass loss, this research would help to answer how different the unvegetated sediment assemblages in Western Port now are compared to the historic situation. A comparison with Port Phillip Bay would also enable us to assess the relevance of ecological and historic factors in changes in benthic fauna, and would provide a new assessment of marine introduced species in Western Port, which has not been surveyed for a decade.

In addition, insight from this research can contribute to answering the question 'Why is there such a high benthic biodiversity in Western Port?' It may also lead to further phylogeographic investigations to unravel the biogeographic history, especially linkages with the biota of the eastern coast.

Ecological functions

Related to research into biodiversity are investigations into ecological functions of benthic species, in order to evaluate the roles of particular species, and to enumerate their contribution to ecosystem scale processes (see also Chapter 14), such as the role of soft-sediment fauna for nutrient cycling, especially on the extensive shallow mudflats.

Experimental field and laboratory studies could enable us to understand why several species of ghost shrimps can coexist and how they differ in their ecosystem engineering properties. Similar experiments could address whether the high benthic diversity facilitates resilience towards disturbances, or whether there is redundancy in the ecological functions realised by (related or unrelated) species, all of which would inform conservation management.

Further experiments should aim to evaluate whether it is the greater habitat diversity on a smaller scale provided by ecosystem engineering species that provides so many niches and supports a higher biodiversity.

On a larger scale, soft sediment habitats need to be evaluated as one part of a mosaic of habitats, and linkages with adjacent habitats for foraging or certain life history stages need to be assessed. This would contribute to an evaluation of the importance of habitat heterogeneity and connectivity in Western Port and other coastal ecosystems.

8 Mangroves

Sabine Dittmann

Western Port supports the southernmost mangrove species in the world, Avicennia marina. Mangroves line most of the shore of the bay and are represented in the three Marine National Parks. Since European settlement there has been some loss of mangroves in the bay, especially near Hastings. Localised destruction, disturbances and changes in the sediment budget of the bay have contributed to changes in mangrove distribution.

There are no recent records of the biota associated with mangroves in Western Port, apart from fish frequenting the mangrove fringe and forest, and research is needed on the links between biota and mangrove disturbance history and patch size. Research is also needed on the functional relevance of mangrove biota for coastal ecosystems in Western Port. Habitat loss and fragmentation are serious threats to mangroves, and landward retreats are needed to prevent the loss of mangroves as the sea level rises.

Mangroves are salt-tolerant trees that grow in the intertidal region of sheltered embayments and estuaries. Only one species, *Avicennia marina*, occurs in Victoria — the southernmost extent of the genus in the world, compared to 39 mangrove species in tropical Australia (Duke 2006). Southern Australia has the largest area of temperate mangroves worldwide (Morrisey *et al.* 2010), and mangrove areas in New South Wales and South Australia are three to four times larger than in Victoria.

Those in Western Port are close to the southernmost limit of mangroves, which is in Corner Inlet. Increases and decreases in the area of temperate mangroves is subject to local geomorphology and to natural and artificial changes to catchments and hydrodynamics, as mangroves are a sink for terrigenous sediment inputs, which contributes to their potential function as an erosion buffer. Mangroves can fuel a detritus-based food web with little direct herbivory, yet rates of nutrient recycling and export as well as uptake of mangrove derived detritus vary locally. Fish as well as benthic fauna occurring in mangroves are usually found in adjacent estuarine or terrestrial systems as well, yet the microhabitat use of benthos or shelter function for juvenile fish make mangroves important habitats at least in certain life stages. The comprehensive review by (Morrisey *et al.* 2010) further highlights that temperate mangroves are far from uniform, with site and regional specific variation in ecological values and functions. Because most temperate mangrove stands are in developed countries, the potential disturbances and management options are different from tropical mangroves (Morrisey *et al.* 2010). Mangroves in South Australia have been well studied in relation to their biota and susceptibility towards pollutants, while those in New South Wales have been subject to many ecological studies. The exhaustive review by (Morrisey *et al.* 2010) considers the literature from all southern Australian mangroves and and is an important source for further reading.

Mangroves of Western Port

History of mangrove studies in Western Port

The first surveys of mangroves in Western Port were carried out soon after Europeans arrived in Victoria (Smythe 1842), yet it was not until the 1970s and 1980s that serious research efforts were made. Studies at that time focused on the distribution of mangroves in relation to tides and geomorphology (Bunt *et al.* 1985, Bird 1986), physiological adaptations and nitrogen cycling (Cain and Boon 1987, Boon and Cain 1988), seedling growth and mangrove establishment (Farrell and Ashton 1974), productivity and litterfall dynamics (Attiwill and Clough 1978, van der Valk and Attiwill 1984c) and associated algal communities (Davey and Woelkerling 1985). This was a time of intense interest in mangrove ecology throughout Australia. Although much of the research was on tropical mangroves, Western Port, where some of the southernmost mangroves in the world grow, was included in comparative studies (Bunt 1995).

This surge in research has been followed by more than 20 years of relative inactivity. Since then, research into mangroves in Western Port has been sporadic and poorly coordinated. Recent efforts examined geomorphological linkages between vegetation, surface elevation and groundwater (Rogers *et al.* 2005a, 2005b, 2006; Rogers and Saintilan 2008) and nutrient cycling and plant–nutrient interactions (Cain and Boon 1987, Boon and Cain 1988). Ecological studies have included investigations into barnacle settlement and their role for mangrove seedling survival (Satumanatpan and Keough 1999, Satumanatpan *et al.* 1999, Satumanatpan and Keough 2001) and investigations into fish assemblages in mangroves and their predation patterns (Hindell and Jenkins 2004, 2005; Smith and Hindell 2005).

8 Mangroves

Since the 1970s Western Port has also been the subject of several environmental assessments and literature reviews in relation to industrial and urban developments, and mangroves were covered in these studies (Shapiro 1975, Ross 2000). Insights gained from Western Port and related ecosystems have made it clear that mangroves, saltmarshes, mudflats and seagrass beds are an integrated whole and have to be considered as such in order to further our understanding and management of the Western Port ecosystem (Shapiro 1975, Harty 2008). More recently, the condition of mangroves and saltmarshes in Western Port has been evaluated under future threats and climate change scenarios (Victorian Saltmarsh Study 2011).

Distribution

The most extensive mangrove fringes in Western Port are those lining the northern and western mainland shores of the bay and French Island, and there are also stands along the south-eastern shores (e.g. Pioneer Bay and Bass River) and the eastern side of Phillip Island. The mangrove stands between Tyabb and Tooradin and along the northern shore of French Island have the highest conservation value (Ross 2000). The estimated area of mangrove in Western Port ranges from 12 km^2 (Shapiro 1975) to 18.23 km^2 (Victorian Saltmarsh Study 2011).

The extent and distribution of mangroves in Western Port has changed considerably over the last 170 years (Figure 8.1) (Bird and Barson 1975). Smythe (1842) found much of the shoreline of the bay covered in mangroves, apart from the clay cliffs near Koo Wee Rup and parts of Phillip Island. This original distribution is still apparent, although only 40% of the mid shoreline around the bay supported mangroves in the mid 1980s (Bird 1986). Losses of mangroves have been caused by land claim near Hastings, and mangrove fringes appear more scattered and discontinuous along the western and southern shores of French Island and at Pioneer Bay (Figure 8.1).

Special features

Western Port mangroves are White Mangroves, *Avicennia marina var. australasica*, and are among the most southerly mangrove opoulations in the world (Duke 2006, Morrisey *et al.* 2010). The stands may have originated from seed dispersal or, more likely, are relicts of a wider distribution when seas were warmer in the late Tertiary – early Pleistocene (Macnae 1966). Air temperatures in Western Port can be as low as 0°C, which *A. marina* can withstand provided there are no frosts (Macnae 1966, Shapiro 1975). At this lower limit of their distribution, the growth of White Mangroves is more stunted, although some reach heights of 2–5 m, with rather uniform appearance of the forest at single locations (van der Valk and Attiwill 1984b, Davey and Woelkerling 1985, Duke 2006). Stunted mangroves have usually a more open canopy (Saenger *et al.* 1977) (Figure 8.2). Seedlings grow slowly and have a high mortality rate in Western Port (Satumanatpan and Keough 1999), whereas seedlings in other temperate mangrove populations in Australia have a low mortality rate (Clarke and Myerscough 1993).

Mangroves are present in all three marine protected areas in Western Port, in particular in Yaringa and Churchill Island Marine National Parks (Edmunds *et al.* 2010). About one third of Yaringa MNP is mangroves and saltmarshes, mostly undisturbed and in a habitat mix with mudflats and seagrass beds. Mangroves cover a small area of French Island MNP, although they are widespread along the northern shore of the island. This park includes some of the major mangroves in Victoria with trees up to 4 m tall (Edmunds *et al.* 2010). The present boundary of this marine park leaves most of the mangrove forests outside of the protected area (Edmunds *et al.* 2010). In Churchill Island MNP mangroves are present at two locations. Other mangrove stands in Western Port, such as those at Rhyll and Sandy Point, are not in conservation reserves.

Figure 8. 1 Recent and historic distribution of mangroves in Western Port, based on Smythe (1842) and Bird (1986).
Note that Smythe included the saltmarsh plant Sclerostegia in the mangrove area (Ross 2000) and, unlike Bird, did not differentiate continuous and scattered mangroves. (Sources: Western Port Seagrass Partnership and Bird 1986.)

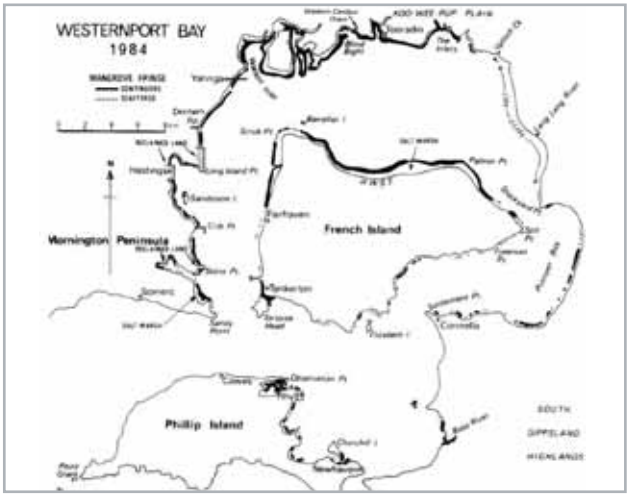


Figure 8.2 Low mangroves near Grantville. (Photo: A. Pope.)

Summary of current understanding

Spatial and temporal distribution patterns

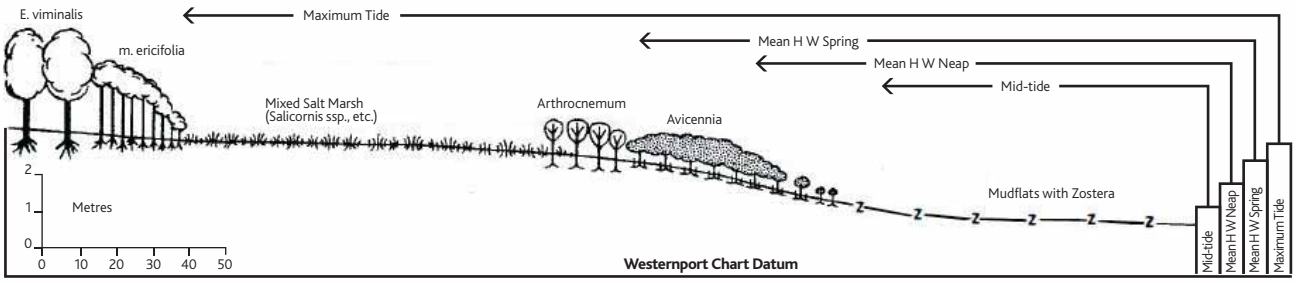
Changes in the sedimentary conditions of the bay following the end of the Holocene transgression about 6000 years ago led to muddier conditions from the erosion of clay cliffs and riverine input (Bird 1986). This spreading mud accretion then allowed colonisation by mangroves, which further affected the geomorphology of Western Port by sedimentation (Bird 1986). Bird (1986) thus viewed mangrove colonisation as a consequence of coastal deposition, and provided further indications of sediment trapping around the pneumatophores. However, the stabilisation of sediment depends on the density of pneumatophores, with a spacing of 5 cm or less affecting deposition and erosion, and is enhanced by the subsurface layer of rootlets and low energy conditions (Spenceley 1987).

The width of the mangrove fringe around Western Port varies between about 40 and 300 m (Marsden *et al.* 1979, Davey and Woelkerling 1985, Bird 1986). On the seaward side mangroves do not extend below the mid-tide level (Bunt *et al.* 1985, Bird 1986) (Figure 8.3), and on the landward side they are genrally bounded by saltmarshes, which extend to the highest spring tide level where *Melaleuca* stands commence (Bird 1971, cited in Saenger *et al.* 1977). In some areas of Western Port, mangroves are bordered on the landward side by heathy woodland, agricultural land, bluffs or cliffs (Bird 1986). Seaward of the mangrove stands, extensive mudflats and sandflats complete the intertidal habitat mix. Mangroves protect these flats from erosion by land run-off and offshore winds (Bird 1986).

Historical maps (Smythe 1842), nautical charts from 1865 and aerial photographs have allowed comparisons to be made of temporal changes in the distribution of mangroves in Western Port (Figure 8.1) (Shapiro 1975, Bird 1986, Ross 2000). The first loss of mangroves occurred in the 1840s, when mangroves were harvested and burnt to obtain barilla ash for soap production. Land claiming near Hastings and Rhyll and the construction of boat landings and piers at Stony Point, Crib Point and Denham Road led to a visible reduction in the extent of the mangrove fringe in the first aerial photographs from 1939. This became a recurring pattern in further aerial photographs in the 1970s, with continued land claiming for industrial and port developments and the removal of mangroves to create boat harbours; see Ross (2000) for review, and also Shapiro (1975). Environmental changes caused by the drainage of adjacent land have also contributed to mangrove loss.

The historical comparisons by Shapiro (1975) and (Bird 1986) also showed that there has been some recovery of mangrove stands; for example, at sites along the northern shores of Western Port where sediment conditions have become more stable again following excavations for the drainage schemes of the 1930, or where channels had been dug through the mangroves, as at Yaringa. While most of the seaward margin of mangroves in Western Port was unchanged for over a century (Bird 1986), some seaward

Figure 8.3 Cross-section of a stylised mangrove-fringed intertidal section in Western Port. (Source: Bird 1986.)



expansion occurred in embayments on French Island and in the north-west of the bay, where extensive mudflats and sediment accretion at the seaward edge of the mangroves allowed the establishment of seedlings. On the northern shores of Western Port the accretion rates were too slow for the seaward advance of mangroves (Bird 1986).

Mangroves have been encroaching into saltmarshes in Western Port, as in other wetlands of south-eastern Australia (Rogers *et al.* 2005b, 2006). This encroachment has occurred at a lower rate in Western Port than on the eastern coast of Australia, perhaps because Western Port is close to a southern extreme for mangroves, or because particular saltmarsh species in Western Port may shade seedlings and inhibit mangrove colonisation, or because of the geomorphological setting (Rogers *et al.* 2005a, 2005b, 2006). Mangrove encroachment is facilitated by a deficit in saltmarsh elevation, which can come about when land use changes in the hinterland affect sediment influx or erosion (Rogers *et al.* 2005b). The need to manage mangroves, saltmarshes and paperbark swamps together, as articulated by (Shapiro 1975), is very apparent from these recent studies.

Sediment dynamics

Bird (1986) found that the accretion of sediment, which would otherwise be mobile in the intertidal area, ranged from less than 1 cm to more than 4.6 cm within three years in mangroves near Yaringa. He concluded that mangroves in Western Port rely on a seaward supply of sediment, and that the accretion rate is subject to the frequency and duration of tidal inundation. Rates of sediment deposition and erosion in mangroves are further affected by bioturbation activities of crabs (Bird 1986, Spenceley 1987).

Sediment accretion is not correlated with elevation, which can be lower or higher than the accretion (Rogers *et al.* 2005b, Rogers and Saintilan 2008). Sediment elevation is also affected by below-ground processes such as root accumulation, decomposition of organic matter, autocompaction and sediment water storage (Rogers *et al.* 2005a,b). Depending on the geomorphological setting, drought conditions can reduce the below-ground water accumulation and lead to a drop in sediment surface elevation, thus affecting the ability of mangroves to respond to sea-level rises (Rogers *et al.* 2005a,b; Rogers and Saintilan 2008). However, the sedimentation rate (1.4 –2.5 mm per year) determined by 210 Pb activity over the last 100–150 years suggests that accretion has exceeded the local sea-level rise (0.26 mm per year) that has occurred over the past three decades (Rogers *et al.* 2005b). This sedimentation rate would not be sufficient to sustain mangroves under worst-case scenarios of sea-level rise of 2–8 mm per year predicted for this century (Morrisey *et al.* 2010).

Production, litter fall and decomposition

Mangrove productivity is usually measured by litter production, and varies considerably within temperate regions (Morrisey *et al.* 2010). The annual production of mangroves in Western Port is 840 g/m^2, with a standing crop of 23.5 kg/m^2 (Attiwill and Clough 1978). Most of the standing crop is roots (about 15 kg/m^2), and leaves and branches contribute about 7–9 kg/m^2 (Shapiro 1975).

The biomass of the entire mangrove area in Western Port has been estimated to be 283 000 tonnes (Shapiro 1975).

The annual litterfall rate was estimated at 162 g/m2 by Attiwill and Clough (1978), whereas (Bunt 1995) grouped Western Port into the climatic zone of long mild summers and cool winters, with a total annual litterfall of 436 ± 148 g/m^2 (dry weight). Leaves account for about 70% of litterfall, followed by wood and debris and then propagules. Mangroves in Western Port flower in April–May and fruit in January–February (Duke 1990).

Litterfall rates in mangrove stands vary significantly between climatic zones around Australia, depending on site-specific growth conditions (Saenger and Snedaker 1993, Bunt 1995). The litterfall rate in Western Port is at the lower end of the range (Attiwill and Clough 1978). In temperate mangroves, litterfall is larger relative to the mangrove biomass, indicating a higher carbon turnover (Saenger and Snedaker 1993). Litterfall in Western Port is seasonal, being highest between late spring and early autumn (Attiwill and Clough 1978, Duke 1990).

(Bird 1986) reported a landward accumulation of leaf litter, shells and crab skeletons in the mangroves of Western Port, but the fate of leaf litter is more varied. Based on a series of leaching and decomposition studies, van der Valk and Attiwill (1984b) calculated that 52% of the litter produced over summer is mineralised *in situ*, 5% remains refractory litter, and 43% is exported as whole leaves (25%), Dissolved organic matter (DOM) (10%) or Particulate organic matter (POM) (8%). This export is accompanied by an export of nutrients, but this is offset by the import of nutrients with sediments and seagrass litter, especially during storms (Attiwill and Clough 1978). Of the leaf litter retained in the mangroves, crab consumption of senescent leaves was estimated to be 30–50% (van der Valk and Attiwill 1984b).

Biogeochemistry

Sediments in Western Port were described as mainly muddy (Bird 1986) or silty sands with 5–8 % organic matter (Stephens 1979). Sediment characteristics of mangroves and saltmarshes differ, with a lower clay content in mangroves (Boon and Cain 1988), although Marsden *et al.* (1979) described saltmarsh sediment as muddier (55–100 % mud) than mangroves sediment (15–60% mud).

Boon and Cain (1988) compared nitrogen cycling in mangroves and saltmarshes in Western Port, and found few differences and no patterns in the nitrogen cycle, although there were differences in particular rates and with sediment depth. NH4 is the major source of inorganic nitrogen in all sediments, yet accounts for less than 0.15% of the total nitrogen in sediments, leaving most of the nitrogen in organic forms. Based on their experiments, Boon and Cain (1988) concluded that nitrogen cycling was correlated with organic matter, the total nitrogen content, and the concentration of soluble reactive phosphorus.
They considered that phosphorus not only affected the primary productivity of mangroves and saltmarshes but also the decomposition processes. Nitrogen concentrations in decomposing leaf litter in Western Port are increasing over time (van der Valk and Attiwill 1984a).

Figure 8.4 Mangrove seedling with barnacles on stem and leaves. (Photo: A. Pope.)

Shapiro (1975) estimated that most of the nutrient cycling is through the turnover of leaves and roots. Nitrogen fixation by bacteria in decomposing leaf litter, roots and sediments contributes to over 40% of the annual nitrogen demand of the mangroves (van der Valk and Attiwill 1984a). (See Chapter 14 for a comprehensive assessment of ecosystem processes in Western Port.)

Cain and Boon (1987) found that the nitrogen, chloride, sodium and water contents in sediments in mangrove stands varied little with season, whereas adjacent saltmarshes were more subject to seasonal variation because of high summer evaporation (Cain and Boon 1987). However, chloride concentrations in the leaf cell sap of *Avicennia marina* varied throughout the year (270 mmol/L in July – 701 mmol/L in March), but were lower than in saltmarsh plants. Correlations between environmental conditions and cellular osmotic potential were poor for mangroves, where short-term changes were suspected to be higher than seasonal ones (Cain and Boon 1987).

Mangrove communities

Associated algae

The macroalgae associated with mangroves in Western Port, as in all temperate mangroves, are mainly red algae, belonging to the genera *Bostrychia, Caloglossa* and *Catenella* (Saenger *et al.* 1977; Davey and Woelkerling 1980, 1985; Morrisey *et al.* 2010). Bostrychia (a mix of three species) was the most commonly occurring alga in Western Port mangrove stands, although *Catenella* contributed most to the algal biomass on pneumatophores (Davey and Woelkerling 1985). The cover of algae on mangrove pneumatophores in Western Port was greater towards the seaward edge, and closer to the sediment than on the tip of the pneumatophores, where longer emergence could dry them out (Davey and Woelkerling 1985). In recolonisation experiments on cleaned pneumatophores, Davey and Woelkerling (1985) detected differential colonisation capabilities of the three main algal groups, with *Caloglossa* being a better coloniser. The resulting algal community thus differed compared to the natural algal community on pneumatophores. *Caloglossa leprieurii* consists of several haplotypes with different adaptations in intertidal conditions, which can further complicate the analysis of their distribution (Zuccarello *et al.* 2001). The macroalgal community contributes to the primary productivity of temperate mangroves and affects the food web (see the review by Morrisey *et al.* 2010).

Barnacles

The small barnacle *Elminius covertus* is the only barnacle found commonly on mangroves in Western Port. It occurs in large numbers on pneumatophores and also on branches and leaves (Figure 8.4). The only detailed study of its ecology, by Nateekanjanalarp (1997) in the area around Rhyll, found that over two years the removal of barnacles from seedling leaves and stems did not produce an increase in the growth rate of seedlings or alter their mortality (Satumanatpan and Keough 1999). Barnacle abundance varied from shoreward to landward across mangrove stands as the result of a complex interplay between the numbers of barnacle larvae, their responses to mangrove surfaces, immersion time of different sections of the stands at high tide, and probable depletion of barnacle larval supply as water moved through forests (Satumanatpan *et al.* 1999, Satumanatpan and Keough 2001) .

Benthic invertebrates

Macnae (1966) and Saenger *et al.* (1977) listed benthic macroinvertebrates from Western Port. Their lists are a restricted subset of mangrove fauna present in tropical and subtropical mangroves. They include 22 gastropod and five bivalve species that are common on the eastern coast, but also include some species found in mangroves along the southern coasts of Australia. The number of mollusc species in Western Port is small, but the gastropods *Salinator fragilis* and *S. solida* (Amphibolidae), *Ophicardelus ornatus* (Ellobiidae) and *Pyrazus ebenius* (Potamididae) are quite abundant (Macnae 1966).

The mangrove crabs in Western Port (*Helograpsus haswellianus, Heloecius cordiformis, Parasesarma erythodactyla*) are also present in mangroves on the eastern coast (Macnae 1966, Morrisey *et al.* 2010). (Macnae 1966) also reported burrows of burrowing shrimps (*Upogebia* and *Alpheus*) in Western Port.

Polychaetes present in mangroves at Western Port include *Neanthes vaali* (Ahsanullah 1976). Other species are very likely to be present, but there are no records specifically from mangroves.

Based on studies elsewhere, the benthic invertebrates in mangroves in Western Port are likely to play a role in food webs and nutrient fluxes; for a review, see Morrisey *et al.* (2010). Most of them are grazers, surface deposit feeders or omnivores, and are eaten by fish, birds or large crabs. The abundance of leaf litter and algae may affect the occurrence and density of macrofauna, while the removal of leaf litter from temperate mangroves by crabs may be of less importance than in some tropical mangroves. Some crabs, such as *H. cordiformis*, can display preferential habitat use with regard to mounds or sediment properties. The burrowing behaviour of crabs and shrimps causes bioturbation, which affects the nutrient flux; see Morrisey *et al.* (2010) and Chapter 14.

Fish and other vertebrates

Mangroves are a structurally complex habitat where fish can find shelter and food (Chapter 11). Hindell and Jenkins (2004) collected 37 species in the mangroves of Western Port. Species richness was highest at the mangrove edge (Hindell and Jenkins 2005), hinting at differential microhabitat use and a transient fish community. Many fish frequent several habitats throughout their life history; the fish found in mangroves were mainly juveniles, while fewer but larger fish accounted for a higher biomass outside of the mangroves (Hindell and Jenkins 2004, 2005). Because the area of pneumatophores at the mangrove edge had the highest fish biodiversity and a high biomass, Hindell and Jenkins (2005) concluded that habitat fragmentation could be beneficial for fish using this forest edge but detrimental to fish resident in mangroves.

Mangroves in Western Port are also visited by land vertebrates such as Black Wallabies, Swamp Rats and various skinks (Counihan *et al.* 2003).

Major threats

Any assessment of current and future threats to mangroves in Western Port has to be seen against the background of historical disturbances outlined earlier in this chapter. Temperate mangroves often occur in isolated populations that are genetically distinct (Morrisey *et al.* 2010). Being almost at their southernmost limit, Western Port mangroves are more sensitive to environmental change (Morrisey *et al.* 2010), and because only one species is present they are more vulnerable than seagrasses or saltmarshes.

The dieback of mangroves in Western Port has been attributed to various causes (Bird and Barson 1975, Shapiro 1975, Bird 1986), which can be classified into four types:

1. Direct habitat destruction — e.g. clearing for boat access, land claiming, or historical cutting and burning to obtain ash for soap production
2. Environmental changes resulting from land use or coastline modifications — e.g. reduced salinity following freshwater diversion from the hinterland, water-logging following the construction of artificial embankments, defoliation from cattle grazing or accidental spraying with herbicide, damage from pathogenic organisms such as *Phytopthora*, and *Pythium*, damage from oil spills, toxic chemicals and other industrial pollution, and wave and current scour from ships and powered boats
3. Natural disturbance — e.g. occasional frosts, severe storms, and natural wave and current scour
4. Natural or human-induced habitat modifications — e.g. sediment mobilisation leading to a build-up of sand, loss of pneumatophores, water-logging or impeded drainage with increased salinity, or an accumulation of seagrass detritus leading to defoliation.

Habitat loss and fragmentation

Mangrove habitat can be lost directly from land claiming, clearance for industrial and port or marina developments, and other effects of urbanisation, and indirectly from changes in the sediment budget of the bay as a result of sedimentation and erosion. The scale of loss depends on the area over which these disturbances occur, but a common result is disconnected stands where once a coherent mangrove coastline existed.

The persistence of mangroves may be affected by climate change, which is changing the frequency and intensity of rainfall and causes sea-level rise. During drought, surface elevation of mangroves can drop leading to increased inundation frequency (Rogers *et al.* 2005a, Rogers *et al.* 2005b, Rogers *et al.* 2006). Any such changes will be difficult to generalise, as patterns of accretion and sediment elevation are site specific and further affected by man-made coastal modifications (Rogers *et al.* 2005a, Rogers *et al.* 2005b, Rogers *et al.* 2006, Rogers and Saintilan 2008).

Risks

The risk to mangroves from habitat loss and fragmentation is very high and the likelihood of severe impact expected to increase with the scale of mangrove affected. As a biogenic habitat, destruction of mangrove habitat threatens this ecosystem and diminishes the ecological role mangroves play for further habitats and biota in the bay.

Consequences

Loss of mangroves, as recorded in historical comparisons by (Bird 1986), was followed by erosion of saltmarshes, steepening of the shore profile, widening of gaps in mangroves with sand mobilisation and deposition, and killing off adjacent mangroves. With a loss of mangroves, sediment trapping in the bay will be reduced, resulting in a higher erosion risk.

Apart from their relevance for shoreline protection, loss or fragmentation of mangroves will also affect associated biota, such as fish communities resident in the forests or frequenting the mangrove fringe (Hindell and Jenkins 2005) (see Chapter 11). With fish of commercial value relying on mangroves at some stage in their life history (Hindell and Jenkins 2004), mangrove loss could have economic consequences for fisheries.

Benthic macrofauna in mangroves includes some species that also inhabit saltmarshes and mudflats (Morrisey *et al*. 2010), but the majority would disappear if their mangrove habitat were lost, resulting in a drop in biodiversity. Older mangrove stands in particular are structurally more complex and can provide habitat for terrestrial insects and spiders (Morrisey *et al*. 2003, Morrisey *et al*. 2010).

Further functional losses would include a decline in the export of dissolved and particulate organic matter from the mangroves into the bay proper, which in Western Port is 43% of the total litter production (van der Valk and Attiwill 1984b), with a consequential fall in productivity.

If mangroves were lost from any of the coastal embayments in the south-east their recovery would be slow because the dispersal capability of *A. marina* is poor which has resulted in disconnected populations with limited gene flow (Clarke 1993).

Water and sediment quality

The water and sediment quality in Western Port is affected by a number of urban, agricultural and industrial activities and developments (Chapter 3), but the timeframes and pathways of their impacts on mangroves can be very different. The existence of many sources that affect water and sediment quality could lead to the loss or fragmentation of mangrove habitat, and larger effects from a combination of sources could occur.

Risks

Because mangroves are a shoreline habitat, effluents and runoff of pollutants or sediments reach them before entering many other marine habitats in the bay. Sediment resuspension in unvegetated soft sediments is wave-driven and can be increased by storms, but also dredging operations, and lead to sediment input in mangroves. Substances bound to finer particles can settle in mangroves, leading to an accumulation of pollutants. Excessive sedimentation or sediment resuspension can affect the fate of pollutants in mangroves, as well as the growth and survival of the mangroves. Dieback of mangroves can also be caused by changes in water and sediment quality from embankments and other coastal developments (Shapiro 1975, Bird 1986). The likelihood of risk thus varies with the particular pathway and the agent that is changing water and sediment quality.

Consequences

Nutrient influx into Western Port can have indirect consequences for mangroves. For example, seagrass dieback leads to an excessive deposition of seagrass detritus in mangroves, which can smother their pneumatophores and seedlings or lead to defoliation (Shapiro 1975, Bird 1986).

The effects of *sediment input* on mangroves depend on quantity and particle size. They can include mangrove dieback or changes in elevation and habitat extent of mangroves and their adjacent habitats. Land use changes have led to an increased delivery of terrigenous sediment into temperate estuaries, and contributed to landward encroachment in south-eastern Australia and seaward encroachment in New Zealand (Morrisey *et al*. 2010). In relation to continued sediment accretion in Western Port, however, Bird (1986) considered mangroves to be reliant on sediment supplied from the bay. The encroachment of mangroves into either saltmarshes (landward) or mudflats (seaward) depends on changes in sediment fluxes between coast and catchment (Rogers *et al*. 2005b). Changes in sediment flux, together with loss or fragmentation of mangroves, would change the spatial mosaic of intertidal habitats in Western Port.

The effects of *heavy metals, toxicants and pathogens* have not been well studied in Western Port, but possible consequences can be derived from investigations in related ecosystems. Metal concentrations are higher in siltier sediment with high organic matter content, and metals are thus easily deposited in mangroves which provide a trap for finer particles (Harbison 1986b). Trace metals, especially copper and zinc, accumulated in mangrove mud in the vicinity of industrial developments in Port Adelaide (Harbison 1986a). Sulfate reduction in anaerobic conditions in mangrove muds retains metals as sulfides in the sediments (Harbison 1986b). The changing pH and redox conditions over tidal and diurnal cycles further affect the accumulation and availability of metals in mangroves (Harbison 1986b). If mangroves are cleared and left exposed, acid sulfate soils can develop, risking the release of acid and metals when sediments are oxidised and rewetted, with detrimental effects for biota (Cook *et al*. 2000, Corfield 2000). Toxicity tests on the polychaete *Neanthes vaali* from mangroves in Western Port showed a slightly higher toxicity for zinc than cadmium (Ahsanullah 1976).

Tributyl tin could affect molluscs (Ruiz *et al*. 1994) in the vicinity of marinas and ports, and imposex (in snails, the development of male reproductive structures in females) been found in mangrove snails in New South Wales (Roach and Wilson 2009), yet no specific indications exist from Western Port. Imposex induced by tributyl tin can quickly

lead to the breakdown of mollusc populations; and because molluscs are among the more abundant animals in the mangrove macrobenthos of Western Port, further functional effects could follow from reduced grazing.

An indication of the consequences of an *oil spill* on mangroves in Western Port can be obtained from an experimental study in South Australian mangroves. Wardrop *et al.* (1987) showed that the lighter, more aromatic oil had a higher toxicity than a heavier oil, that the effects intensified when the oil was combined with a dispersant, and that a dispersant by itself has detrimental effects. Some oil residue was still visible three years after the experiment. Defoliation occurred following the application of a 1 : 1 mix of lighter Tirrawarra crude oil with dispersant. Mangroves did not produce new leaves for about eight weeks after the simulated spill in all of the treatments.
The combination of oil plus dispersant caused greater leaf damage than crude oil alone. About 20% of pneumatophores showed discolouration and withering three months after the oil and dispersant application. Flowering and fruiting of the mangroves was not affected in the experiment.

Herbicides have been reported to cause defoliation of mangroves in Western Port following aerial spraying (Shapiro 1975, Bird 1986), and they can also affect *A. marina* seedlings and trees by accumulating in sediments following run-off (Duke 2008).

Both reduced and increased salinity have led to mangrove dieback in Western Port in the past (Shapiro 1975, Bird 1986). Salinity has physiological effects on mangroves with regard to nutritional demands, and also limits their water uptake (Krauss *et al.* 2008).

Acidity is expected to have little effect on mangrove trees because sediments in mangrove muds can be slightly acidic, but a lower pH would affect associated benthos such as molluscs by reducing the strengths of their shells.

Extraction and disturbance

Risks

Mangroves are no longer used as a resource for wood or ash in Australia, so the risk from extraction is rather low. Disturbances of varying severity can result from trampling and habitat loss.

Consequences

Past extractions of mangroves in Western Port have led to the loss of mangrove stands (Bird 1986), with consequences as described above.

Disturbance by trampling includes damage to pneumatophores and microhabitat structures of the mangrove and a decrease in biomass of the algal associations, both of which also lead to a decrease in associated gastropods (Ross 2006). The establishment of boardwalks can reduce these consequences, although benthic changes in the immediate vicinity of boardwalks have been recorded, albeit without a clear pattern (Kelaher *et al.* 1998a). Localised changes in sediment texture and microhabitat structures in the vicinity of boardwalks can also affect the density of crabs (Kelaher *et al.* 1998b). The consequences of more severe disturbances are described under 'Habitat loss and fragmentation' above.

Sea-level rise

Risks

Accelerated sea-level rise would reduce the intertidal area available as mangrove habitat, unless inland retreats are possible. The Holocene rise in sea-level led to gradual shifts in coastal habitats (Barnett *et al.* 1997), but such an adaptation could be impeded by the fortification of coastlines to protect infrastructure or prevent inundation.

Consequences

Prolonged submergence of *A. marina* seedlings affects their physiology, but recovery is possible (Sayed 1995, cited in Krauss *et al.* 2008). The capability for continued sediment accretion and maintenance of surface elevation relative to sea-level rise, together with landward retreat options, is important for mangrove adaptation to climate change induced sea-level rise (Boon *et al.* 2010). Otherwise, erosion of mangroves from the seaward side as well as backwash from seawalls could rapidly lead to the loss of mangroves. However, mangroves in sediment-rich estuaries may be the most resilient to such effects of climate change (Morrisey *et al.* 2010).

Temperature increase

Risks

With predicted warming accelerated by climate change, mangroves that are at their southernmost distribution limit in Western Port may be exposed to reduced risk of frost kill.

Consequences

Warmer conditions may benefit growth and survival of seedlings, although the effects of temperature on mangrove growth and physiology are not well understood (Krauss *et al.* 2008). Temperate mangroves would probably benefit from warmer conditions, and could extend their range (Morrisey *et al.* 2010).

Pests

Risks

The human-induced spread of marine species around the world is well documented (Carlton 2000, Bax *et al.* 2003, Occhipinti-Ambrogi and Savini 2003) and coastal embayments with frequent international shipping connections are prone to the invasion of exotic species (Hewitt *et al.* 2004). Increasingly, the relevance of regional transport is recognised for further dispersal of invasive species (Floerl *et al.* 2009). Regional transport in Australia is seen as the likely source of introduction for *Parasesarma erythodactyla* into mangroves in Spencer Gulf, South Australia (Baggalley 2009). Western Port, which is adjacent to Port Phillip Bay which now hosts a large number of exotic species (Hewitt *et al.* 2004), is at risk from introduced species, if ballast water and hull fouling from current and future shipping activities are not managed appropriately.

Consequences

The consequences of the introduction of exotic marine species are very site and species specific. Ecosystems with a high diversity can be more resilient to the establishment of introduced species (Stachowicz *et al.* 1999), but mangroves in Western Port may not have such resilience. However, the specialised niches available in mangrove stands may limit the establishment success of invasive species.

Cumulative impacts

None of the threats mentioned above occurs in isolation. Both risks and consequences will intensify from a combination of threats, and will depend on the spatial and temporal scale of the stress. Earlier studies in Western Port have shown an awareness of multiple causes of mangrove dieback (Shapiro 1975, Bird 1986) and provide a valuable background against which to assess future impacts. Mangroves in Western Port are no longer located in a pristine environment, and past stress has to be considered in combination with current and future impacts on this tidal embayment.

Research to fill key knowledge gaps

The greatest threats to mangroves in Western Port come from land use changes and any related changes to sediment deposition and nutrients and toxicants. Climate change will also present an important threat.

The greatest challenge to efficient mitigation of threats from urban and industrial developments lies in knowledge gaps about ecosystem-scale responses (Lee *et al.* 2006, Morrisey *et al.* 2010). In particular, the effects of cumulative impacts need to be evaluated. For Western Port, knowledge gaps for ecosystem-scale assessments of a complex disturbance regime include scarcity of data on benthic communities, plant–animal and other biotic interactions (above and below ground), animal–sediment interactions and food web studies. Interactions between stressors (e.g. temperature, flooding, salinity, nutrients, CO_2) on mangrove growth and recruitment also need to be investigated further to evaluate responses to changing environmental conditions (Krauss *et al.* 2008). Furthermore, from the insights we have into the functioning of temporal mangroves and the impacts of human disturbances obtained from studies in various regions, it is clear that site-specific idiosyncrasies in the functioning and disturbance response of mangroves in Western Port are poorly understood.

Biodiversity of mangroves in Western Port

One of the knowledge gaps identified in this review is the scarcity of information on mangrove-associated invertebrate fauna in Western Port, whereas unvegetated sediments are known to be inhabited by a species-rich community (Chapter 7). Research into the biodiversity of benthic and terrestrial invertebrates is needed to provide further information on linkages with the history (particularly disturbances) and patch size of particular mangroves in Western Port. Additional investigations into the life histories of selected species would contribute to the assessment of the resilience of major biodiversity components.

Ecological functions

Species have several roles in ecosystems, for trophic as well as non-trophic interactions, which include, for example, ecosystem engineering by burrowing crabs. Investigations into interactions between organisms (mangroves and associated biota) with their environment as well as with other species are essential to obtain an understanding of the ecosystem structure and functions provided by species in particular habitats. Such knowledge is essential for evaluating how any environmental changes will affect the functioning of ecosystems or parts thereof, even if they are not in the direct path of any disturbance event.

Habitat landscape

Mangroves are part of a mosaic of intertidal habitats, and the connectivity between saltmarshes, mangroves, mudflats and subtidal sediments is poorly understood. Investigations into the dependence of habitats for certain life history stages of both invertebrate and vertebrate species would increase our knowledge in the relevance of this habitat heterogeneity for the biodiversity and ecosystem scale processes.

Such investigations would need to extend beyond the biota and include assessments of exchange processes of particulate and dissolved organic matter. A better understanding is also needed of the effects of seagrass detritus on nutrient budgets, productivity and mangrove survival.

Morphodynamic patterns

In a sedimentary environment like Western Port, a more detailed understanding on the effects of sedimentation and erosion on mangrove recovery and seedling survival are needed, as well as an understanding of how mangroves affect the sediment budget at different scales. Understanding the factors that limit seedling success will be an essential component of mangrove restoration, which may be important for reducing suspended sediment loads.

According to Ross (2000), it is still debated whether mangroves in Western Port are advancing seaward or landward. With the advantage of historical maps and aerial photography, calculations of the loss or gain of mangroves in Western Port can contribute to the understanding of past changes, which would supplement future research to make stronger predictions about responses of ecosystem components to environmental changes.

9 Saltmarshes

Paul I. Boon

Photo courtesy Michael Keough.

Saltmarshes occur around much of the coast of Western Port, generally between the mangrove fringe on the seaward side and more terrestrial vegetation, such as Swamp Paperbarks and Manna Gum woodlands, on the landward side. There are about 1 000 ha of saltmarsh in Western Port, which is about the same area as there is of mangroves. The only larger coastal saltmarshes in Victoria are at Connewarre (at the mouth of the Barwon River), around Corner Inlet-Nooramunga, and in the Gippsland Lakes around Lake Wellington, Lake Reeve and Jack Smith Lake. A number of the larger saltmarshes in Western Port occur in protected areas, such as the Yaringa (980 ha), French Island(2 800 ha) and Churchill Island (670 ha) Marine National Parks. Saltmarshes in Western Port are likely to be very vulnerable to sea-level rise and other consequences of climate change, especially rising air and water temperatures. Saltmarshes have been progressively lost already, due mostly to development for agriculture and industry, around the western and northern shores of Western Port.

We identify several research gaps, including better understanding of the ways tidal inundation affects waterlogging and salinity regimes in saltmarshes, and in particular how they affect the saltmarsh plant communities which provide food and habitat for terrestrial and aquatic animals. Much more research is needed on the way that terrestrial (e.g. bats and bushbirds) and aquatic (e.g. waterbirds and shorebirds) use saltmarshes. The susceptibility or resilience of saltmarshes to threats such as nutrient enrichment, oil pollution, weed invasion (e.g. by Spartina), altered salinity and hydrological regimes, and climate change is also an important research gap.

Distribution

Coastal saltmarsh and other types of estuarine wetland in Victoria

The general perception of a saltmarsh is a coastal area intermittently inundated by only the highest tides and vegetated with sparse, low-lying, succulent plants. The definition of a saltmarsh, however, is not a trivial or easy matter, and the recent Victorian Saltmarsh Study (2011)[6] identified at least 20 descriptions that have been invoked by various researchers and management agencies.

In Victoria, native vegetation is classified and described in terms of ecological vegetation classes, or EVCs. Ecological vegetation classes are defined as one or more floristic and structural types that appear to be associated with a recognisable environmental niche and that can be characterised by their adaptive responses to ecological processes that operate at the landscape scale (DNRE 2002). Coastal saltmarsh in Western Port is classified as EVC 9 Coastal Saltmarsh Aggregate (DSE 2009). EVC 9 is described as an aggregate because it is not floristically, structurally or ecologically homogeneous, and with further analysis could be resolved into a number of smaller and more homogeneous units. As it currently stands, EVC 9 describes coastal saltmarsh as a low shrubby (to herbaceous, sedgy or grassy) type of vegetation that occurs in sheltered embayments and estuaries, on salinised coastal soils that are influenced by tides tidally. Its floristic composition variously includes shrubby dicots such as *Tecticornia* (previously *Sclerostegia*) *arbuscula*[7], *Tecticornia pergranulata* and *Tecticornia halocnemoides*, grasses such as *Austrostipa stipoides* and *Distichlis distichophylla*, and dicot herbs such as *Sarcocornia quinqueflora*.

Coastal saltmarsh in south-eastern Australia (and Western Port) often occurs alongside a wide range of other types of wetlands in coastal and estuarine settings, and it is important to differentiate it from them.

In Victoria, these other types of saline coastal wetland may include:

- Brackish Grassland (EVC 934)
- Brackish Herbland (EVC 538)
- Brackish Lignum Swamp (EVC 947)
- Brackish Sedgeland (EVC 13)
- Brackish Wetland (EVC 656)
- Estuarine Flats Grassland (EVC 914)
- Estuarine Reedbed (EVC 952)
- Estuarine Scrub (EVC 953)
- Estuarine Wetland (EVC 10)
- Mangrove Shrubland (EVC 140)
- Saline Aquatic Meadow (EVC 842)
- Sea-grass Meadow (EVC 845)
- Seasonally Inundated Sub-saline Herbland (EVC 196)
- Unvegetated (open water/bare soil/mud) (EVC 990 – 'Non Vegetation')

EVC 10 Estuarine Wetland is especially problematic. It is dominated by Sea Rush *Juncus kraussii*, and commonly forms in estuarine areas subject to freshwater seepage, just behind EVC 9. It is not included as a 'classic' saltmarsh type of wetland in Victoria, and thus is classified as a separate EVC. In other jurisdictions (e.g. New South Wales), however, *Juncus kraussii* is considered a component of 'core' intertidal saltmarsh. More broadly, a number of plant species that are listed as characteristic saltmarsh taxa in New South Wales (DECC 2009) are not generally considered to be indicative of saltmarshes in Victoria (e.g. *Baumea juncea*, *Bolboschoenus* spp., *Ficinia nodosa*, *Phragmites australis*, *Schoenoplectus* spp., *Tetragonia tetragonioides* and *Typha* spp.). These taxa are allocated to other wetland EVCs in the Victorian schema, such as EVCs 10, 13, 914 and 952.

Thus what is considered 'coastal saltmarsh' in Victoria is not completely congruent with what is considered 'coastal saltmarsh' in New South Wales, nor probably in South Australia, Tasmania and Queensland. In Queensland, for example, the recent state-wide mapping of coastal wetlands includes 'mangrove', 'samphire' and 'claypan' as

6 Some of the information in this review has been extracted from the Victorian Saltmarsh Study (Boon *et al.* 2011). *Mangroves and coastal saltmarsh of Victoria: distribution, condition, threats and management*. Institute for Sustainability and Innovation, Victoria University, Melbourne.

7 The taxonomy of *Halosarcia*, *Pachycornia*, *Sclerostegia* and *Tegicornia* have been recently revised, and the single genus *Tecticornia* now applies: see Shepherd & Wilson (2007).

map units (EPA Qld 2008) but includes, as a separate map unit, *Sporobolus virginicus* grassland associations. In Victoria this species is considered as just one of a number of different types of coastal saltmarsh (Coastal Saline Grassland, see below), but in Queensland it is given a map unit of its own.

This decision is justifiable on the basis of the (species-poor) floristics of Queensland's extensive coastal saltmarsh, but it again demonstrates that what one jurisdiction considers coastal saltmarsh is not always what another one does.

The definition adopted by the Victorian Saltmarsh Study (Boon *et al*. 2011) for coastal saltmarsh is:

'land that experiences recurrent low-energy inundation by seawater and which is vegetated by low-growing vascular plants (generally <1.5 m height), such as succulent chenopods and salt-tolerant monocots'.

This definition differentiates coastal saltmarsh from other types of saline wetland with a similar floristic composition, but which occur inland. Saline or hypersaline saltmarshes, for example, occur in the drier parts of western, north-western and north-central inland Victoria and are dominated by genera such as *Atriplex, Halosarcia, Lepilaena* and *Ruppia*, which occur also in coastal saltmarshes (Love 1981). Inland saltmarshes are variously described as EVC 708 or EVC 888 in Victoria. The definition also excludes salt-tolerant vegetation in the salt-spray or splash zone along cliffs, and instead refers only to intertidal land. It also makes specific reference to the salt-tolerant characteristics of the vegetation, which distinguishes saltmarsh vegetation from non-halophytic wetland vegetation that can occur on the coast, such as Common Reed *Phragmites australis* and *Cumbungi Typha* spp. The specific case of the mildly salt-tolerant *Juncus kraussii* was discussed above. Note that in Victoria, mangroves are allocated to EVC 140 Mangrove Shrubland. Structurally, but not floristically nor in terms of position in the landscape, the Western Port mangrove *Avicennia marina* is not that dissimilar to the woody saltmarsh shrub Tecticornia arbuscula.

In recognition that the current EVC description for coastal saltmarsh was inadequate and that the aggregate needed to be disassembled, the Victorian Saltmarsh Study (Boon *et al*. 2011) proposed a new typology for EVC 9. Doug Frood, an author of the study, recommended that the existing EVC be divided into seven EVCs. This recommendation is currently being considered by the Department of Sustainability and Environment. The seven proposed EVCs are:

- Wet Saltmarsh Herbland — Low herbland dominated by succulent to semi-succulent halophytic herbs or semi-shrubs, occupying low-lying areas of coastal saltmarsh subject to regular tidal inundation. Often very species-poor, most frequently dominated by *Sarcocornia quinqueflora*, less commonly by *Hemichroa pentandra, Selliera radicans, Samolus repens* or *Sueda australis*, and on rare occasions *Triglochin striata*.
- Wet Saltmarsh Shrubland — Shrubland dominated by halophytic species and subject to regular tidal inundation. Often very species-poor, most frequently dominated by *Tecticornia arbuscula*, much less commonly by *Atriplex paludosa*, and rarely by *Atriplex cinerea*. *Sarcocornia quinqueflora* is also frequent in wetter communities.
- Coastal Saline Grassland — Grassland dominated by rhizomatous grasses, occurring towards upper zones of coastal saltmarsh. Frequently very species poor and typically dominated by either *Distichlis distichophylla* or *Sporobolus virginicus*.
- Coastal Dry Saltmarsh — Herbland to low shrubland of upper saltmarsh, subject to relatively infrequent or rare tidal inundation. Variously dominated by *Sarcocornia blackiana, Frankenia pauciflora, Disphyma crassifolium, Angianthus preissianus* or very rarely *Sebaea albidiflora*.
- Coastal Hypersaline Saltmarsh — Low shrubland dominated by succulent chenopods (or rarely Salt Lawrencia *Lawrencia squamata*), occurring in highly hypersaline saltmarsh habitat above the zone of regular tides. Dominated by *Tecticornia pergranulata, T. halocnemoides*, or very locally *Lawrencia squamata*.
- Coastal Tussock Saltmarsh — Upper saltmarsh zones dominated by robust tussock-forming grasses or graminoids, such as *Gahnia filum* or *Austrostipa stipoides* with a range of halophytic species at lower covers).
- Saltmarsh-grass Swamp (inundation-prone saltmarsh vegetation dominated by Australian Saltmarsh-grass *Puccinellia stricta* and sometimes with *P. perlaxa*).

The significance of this proposal is twofold. First, it suggests that Victorian saltmarsh is taxonomically and structurally more complex than is implied by the existing single EVC, and certainly more complex that is often thought to be the case by the community and land managers. Teasing apart of the EVC aggregate into a number of discrete EVCs that better describe this complexity would enable better mapping of coastal saltmarsh and, importantly, enable the possible threats and losses for the different types to be assessed individually; for example, not all the different types of saltmarsh are likely to be affected in the same way by grazing. Second, clarification of what constitutes coastal saltmarsh allows it to be recognised explicitly as part of a larger mosaic of wetland types that occur in coastal settings across south-eastern Australia. An inventory that includes, for example, *Juncus kraussii* as saltmarsh (as in New South Wales) or treats *Sporobolus virginicus* as a separate subset (as in Queensland), would be difficult to reconcile across state boundaries. In other words, what is meant by 'coastal saltmarsh' and what is included and what is excluded in its mapping and inventory needs to be stated explicitly, but often is not.

The proposed new EVCs were used as mapping units in the mapping and inventory component of the Victorian Saltmarsh Study, the results of which are summarised below.

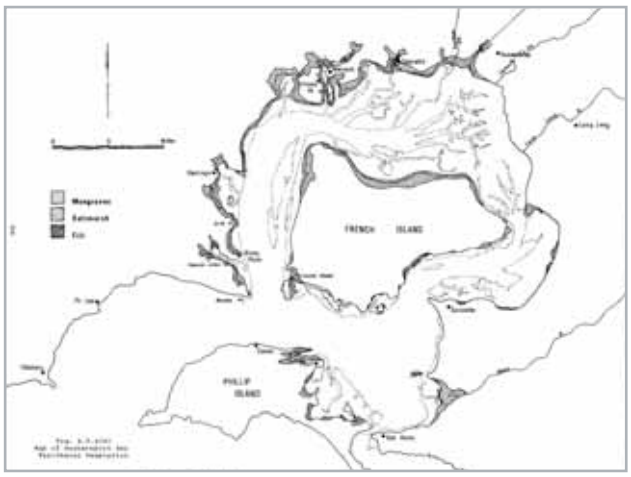
Mapping and inventory of coastal saltmarsh in Western Port

Mapping

Ross (2000) summarised the historical mapping of shorelines, mangroves and saltmarsh in Western Port, commencing with the 1842 surveys of George Smythe. A more detailed state-wide historical analysis was undertaken for the Victorian Saltmarsh Study (Boon *et al*. 2011), which also started with maps by the early Victorian surveyors.

The first 'modern' mapping of coastal saltmarsh in Western Port seems to have been undertaken as part of the Westernport Bay Environmental Study of 1973–1974 (Shapiro 1975). Peripheral vegetation of Western Port was mapped at 1 : 15 000 (from aerial photographs) and selected areas were mapped at 1 : 5000. Mapping units were mangrove and saltmarsh, i.e. saltmarsh was not divided up floristically or structurally, but instead was mapped as a single unit much like the current-day EVC 9. Figure 9.1 shows the whole-of-bay map of Shapiro (1975).

Figure 9.1 Map of peripheral vegetation in the 1975 Westernport Bay Environmental Study. (Source: Shapiro 1975.)



Not long after the Shapiro (1975) investigation, Carr (1979) undertook state-wide mapping of habitat used by the Orange-bellied Parrot *Neophema chrysogaster* — i.e. mainly coastal saltmarsh — at a scale of 1: 100 000, including Western Port (Figure 9.2). His study indicated that less than 60 km^2 of coastal saltmarsh remained in the state. Three types of saltmarsh were identified, although they were conflated somewhat in the final maps:

- *Sarcocornia (Salicornia) quinqueflora* with no *Arthrocnemum* (now *Tecticornia*) arbuscula present
- *Tecticornia arbuscula* plus *Sarcocornia quinqueflora*
- *Tecticornia halocnemoides* with or without *Sarcocornia quinqueflora* and *Tecticornia arbuscula*.

The biodiversity interactive mapsite of the Department of Sustainability and Environment shows the current modelled and partly ground-truthed distribution of EVC 9 for the entire state. Figure 9.3 shows, as an example, the current (2005 EVCs) distribution of peripheral vegetation between Tooradin and Warneet, on the northern shore of Western Port. Mangrove shrubland and coastal saltmarsh have been specifically identified, as well as areas where they could not be reliably differentiated at the scale of mapping: these areas are termed 'mosaics'.

It is acknowledged that finer-scale mapping could untangle such mosaics, which is not the case with EVC aggregates because their lack of resolution is a function of the weakness of the underlying classification system rather than low spatial resolution of the mapping. Note also that the mapping shown on the DSE interactive mapsite conflates the different sorts of saltmarsh into one EVC, EVC 9 Coastal Saltmarsh Aggregate, because the Victorian Saltmarsh Study's recommendations have yet to be accepted and implemented.

The most recent mapping of Western Port's wetland vegetation was reported in the Victorian Saltmarsh Study. This mapped the distribution of coastal saltmarsh, mangroves and a number of other estuarine wetland types across the state at a scale of 1 : 10 000. The more detailed typology for saltmarsh EVCs described above was employed, and mosaics were mapped only when detailed ground-truthing failed to differentiate individual EVC map units. Ten mapsheets were prepared to cover Western Port; Figure 9.4 shows the mapsheet for Warneet–Tooradin. The advantage of this mapping over previous attempts is not only its finer scale (1 : 10 000) and much improved resolution (compare Figure 9.3), but also the strong emphasis on ground-truthing to confirm tentative interpretations based on 1 : 5000 aerial photographs, and the mapping of different types of coastal saltmarsh rather than a single saltmarsh aggregate as used in earlier mapping (e.g. Figures 9.1–9.3). The different types of saltmarsh in the Warneet–Tooradin area are shown in Figure 9.5, which is the legend to the mapsheet shown in Figure 9.4. Figure 2.2 (page 32) also shows saltmarsh and mangrove distributions for the whole of Western Port.

In addition to the mapping of peripheral vegetation for the whole of Western Port, there has been some detailed mapping undertaken of parts of Western Port, usually as a result of consultancies. Unlike the more widely available whole-of-bay mapping, it is impossible to determine all the consultancy projects that have mapped parts of Western Port, as there are numerous client groups (Catchment Management Authorities, Parks Victoria, Melbourne Water, Coastal Boards, and various State Government agencies, etc.) and many of the reports are ephemeral and not archived.

Figure 9.2 Distribution of saltmarsh of Western Port in the report by Carr (1979). Saltmarsh is shown in green.

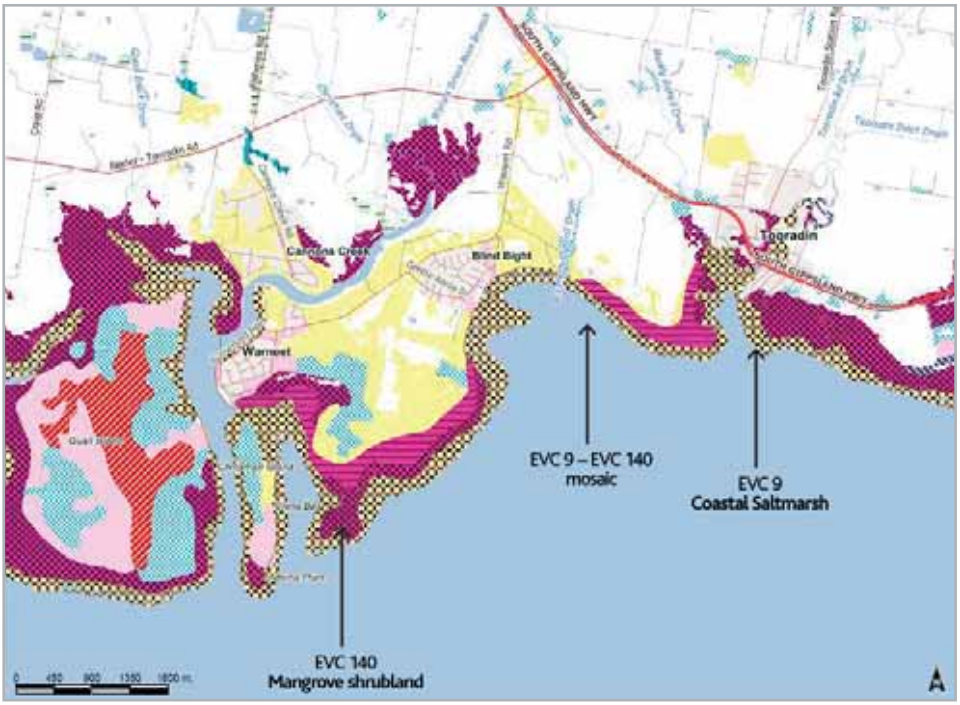


Figure 9.3 Peripheral vegetation between Tooradin and Warneet, northern shore of Western Port (EVCs as of 2005).
(Source: DSE biodiversity interactive mapsite, viewed 14 February 2011.)

Figure 9.4 Mapping of saltmarsh and mangrove in the Warneet–Tooradin area of Western Port. (Source: Boon *et al.* 2011, mapsheet 29.)

Figure 9.5 Legend to mapsheet 29 (Boon *et al.* 2011), showing the different types of saltmarsh in the Warneet–Tooradin area as mapped for the Victorian Saltmarsh Study (Boon *et al.* 2011).

Parts of the northern shore, around The Inlets near Koo Wee Rup, were mapped at a fine scale by Biosis Research (Yugovic 2008). Earlier, Yugovic and Mitchell (2006) undertook an ecological survey of Koo Wee Rup swamp. Other studies may exist, but none have been found to date. In this aspect the literature on Western Port's mangroves and coastal saltmarsh probably differs little from that on other types of wetland in Australia, which also suffer from poor availability and a lack of peer review (Boon & Brock 1994). An illustration of the importance of unpublished reports for Western Port is provided by the review by Ross (2000) of the area's mangroves and coastal saltmarsh: of the 88 reports she cited, about one half were in 'grey literature' and thus hard to obtain and subject to more variable quality control than material in the published scientific literature.

A few student theses have involved mapping Victorian mangroves and coastal saltmarsh or attempted some aspect of an inventory, including studies of Western Port. Ghent (2004), for example, compared past and present distributions of coastal saltmarsh in Port Phillip Bay, and found that about 65% of pre-European saltmarsh had been lost, mostly before 1978. However, student theses are difficult to collate, and much depends on a personal knowledge of the project work; although doctoral theses and most masters theses are routinely indexed and stored by university libraries, honours theses are not.

Inventory

As a result of the mapping undertaken by the Victorian Saltmarsh Study (2011), there is now good information on the areas of coastal saltmarsh and other estuarine wetlands around Western Port. Table 1 shows the areas in Western Port of the vegetation types mapped by that study.

Table 9.1 Areas of a range of estuarine wetland types around Western Port. (Source: Boon *et al.* 2011.)

Region	Wetland area (ha)			
	Combined saltmarsh	EVC 10 Estuarine Wetland	EVC 140 Mangrove Shrubland	EVC 196 Seasonally inundated Sub-saline Herbland
The Inlets	48	6	8	0
Western Port	1 088	58	1 230	0
Victorian total	19 212	3 227	5 177	647

The Victorian Saltmarsh Study (2011) indicated that in Victoria there were 19 212 ha (ca 192 km^2) of Coastal Saltmarsh aggregate, 5177 ha (ca 52 km^2) of Mangrove Shrubland (EVC 140), and 3227 ha (ca 32 km^2) of Estuarine Wetland (EVC 10). These figures exclude some EVCs that may occasionally be considered saltmarsh on less detailed maps, such as Seasonally Inundated Subsaline Herbland (EVC 196, ca 647 ha) and Saline Aquatic Meadow (EVC 842, no area calculated because it is an ephemeral EVC). These values are by far the most accurate to date, and supersede earlier estimates of the total area of saltmarsh in the state, for example by Bucher and Saenger (1991: 125 km^2; cited in Kelleway *et al.* 2009). According to vegetation mapping currently available from the Department of Sustainability and Environment, the area of EVC 9 Coastal Saltmarsh Aggregate is about 132 km^2.

Descriptions of Western Port saltmarshes

Vertical and spatial zonation in vegetation

There is a long history of ecological studies of saltmarshes in Australia. For example, Hamilton (1919) and Collins (1921) studied saltmarsh and mangrove vegetation in the Sydney region, Pidgeon (1940) reported on spatial zonation (which she interpreted as successional patterns) in mangroves and saltmarshes along the central coast of New South Wales, Patton (1942) studied Victorian saltmarshes, and Curtis and Somerville (1947) described coastal saltmarshes in Tasmania. The 1942 study by Patton on Western Port saltmarshes included a floristic description of vegetation in different parts of the bay and an analysis of the role played by elevation in structuring the vegetation.

For the next three decades there appears to have been little study of Western Port saltmarshes, until a number of studies were undertaken in the mid-late 1970s. The Shapiro (1975) report shows an idealised zonation of vegetation, from seagrass beds to terrestrial Manna Gum *Eucalyptus viminalis* – Swamp Paperbark *Melaleuca ericifolia* stand (Figure 9.6). A similar diagram appeared in Bird (1993). This zonation was explicitly interpreted as an ecological succession, although more recent interpretations would not necessarily support such a conclusion (see below).

Bridgewater (1975) provided what is probably the most detailed description of plant zonation within an Australian saltmarsh, using Western Port as the study site. Behind the most seaward zone of *Avicennia marina*, nine vegetation complexes were identified on the basis of floristic and structural criteria:

- introduced *Spartina*
- extensive *Salicornia* (now *Sarcocornia*) – dominated zone
- extensive Arthrocnemum (now *Tecticornia*) – dominated zone
- *Suaeda* complex
- *Puccinellia* complex
- *Juncus* complex
- *Stipa* (now *Austrostipa*) complex
- *Schoenus-Cotula* complex
- *Melaleuca* zone

In some cases, where there is a clear and simple elevational gradient, the various zones follow a consistent pattern with distance from the sea; in other cases they are intermixed into complex mosaics, the size and juxtaposition of which depends on small-scale changes in topography and drainage caused by minor depressions or raised areas (e.g. hummocks) or creeks and tidal runners. The importance of tidal inundation, freshwater inputs and climate were analysed for Sydney saltmarshes in the late 1960s and early 1970s by Clarke and Hannon (1967–1971) (Figure 9.7), but little detailed work has been undertaken in Australia since then to unravel these interactions. Some potentially relevant work was published by Raulings *et al.* (2010), but it dealt with brackish wetlands of the Gippsland Lakes rather than Western Port saltmarshes.

Figure 9.7 Importance of tidal inundation and rainfall in controlling hydrological and salinity regimes and plant performance in coastal saltmarsh. (Source: Nybakken 2001, after Clarke & Hannon 1969.)

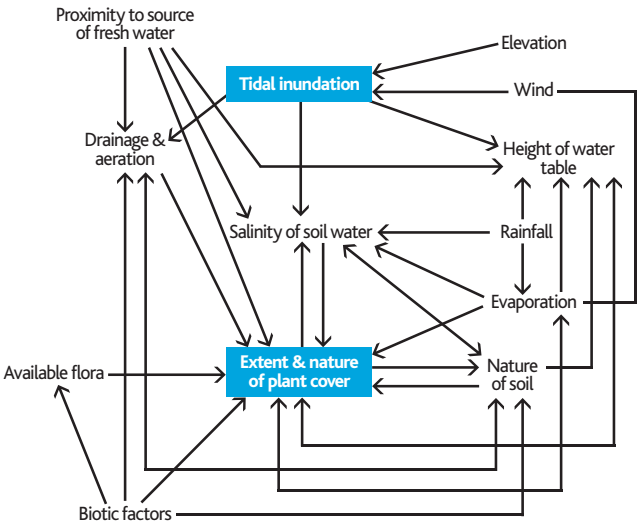
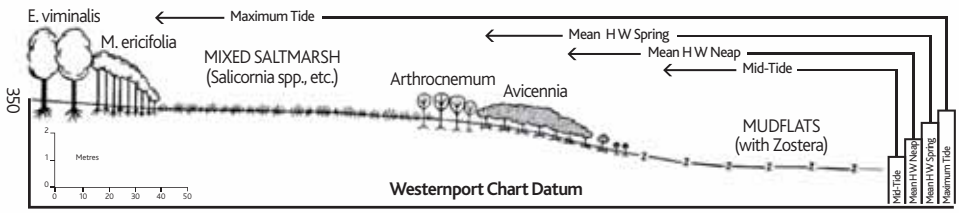


Figure 9.6 Elevational zonation of peripheral zonation in Western Port. (Source: Shapiro 1975.)



Successional change and rates of sedimentation

A clear elevational pattern is often taken to indicate successional changes in peripheral estuarine vegetation. In his monograph *The Coast of Victoria*, Bird (1993, page 196), for example, argued that 'The building of a mangrove-fringed salt marsh terrace around the northern shores of Westernport Bay during the past 6000 years was the outcome of vegetation colonising and stabilising foreshore areas as muddy sediment accreted'. Figure 9.8 shows the accompanying figure in Bird (1993) that outlined the proposed mechanism.

There are few reports of changes in surface elevation in Australian saltmarshes (e.g. see Rogers *et al.* 2005a), but the exhaustive study of Rogers *et al.* (2006) shows rates varying from –0.68 to +5.27 mm year1 in various saltmarshes of south-eastern Australia. Rates for saltmarshes and mangroves at four sites in or near Western Port are shown in Table 9.2.

Table 9.2 Mean (± standard errors) rates of change in surface elevation at four sites in Western Port. (Source: Rogers *et al.* 2006.)

Site	Vegetation type	Change in surface elevation (mm year-1)
French Island	Mangrove	-2.13 + 1.66
	Saltmarsh	5.27 + 0.96
Koo Wee Rup	Mangrove	-0.03 + 2.23
	Saltmarsh	-0.16 + 0.94
Quail Island	Mangrove	-2.60 + 2.07
	Saltmarsh	-0.68 + 1.18
Rhyll	Mangrove	0.92 + 1.87
	Saltmarsh	0.64 + 0.75

Rapid rates of sedimentation in the lowest parts of a saltmarsh imply that saltmarshes may extend laterally (i.e. seawards), as well as accreting vertically. This finding has important implications for the resilience of coastal saltmarsh to sea-level rise, and presumably is a function (at least in part) of high sediment loads in Western Port. Thus a process perceived as problematic for seagrasses — water-column turbidity and sediment deposition — may be seen as beneficial for saltmarshes.

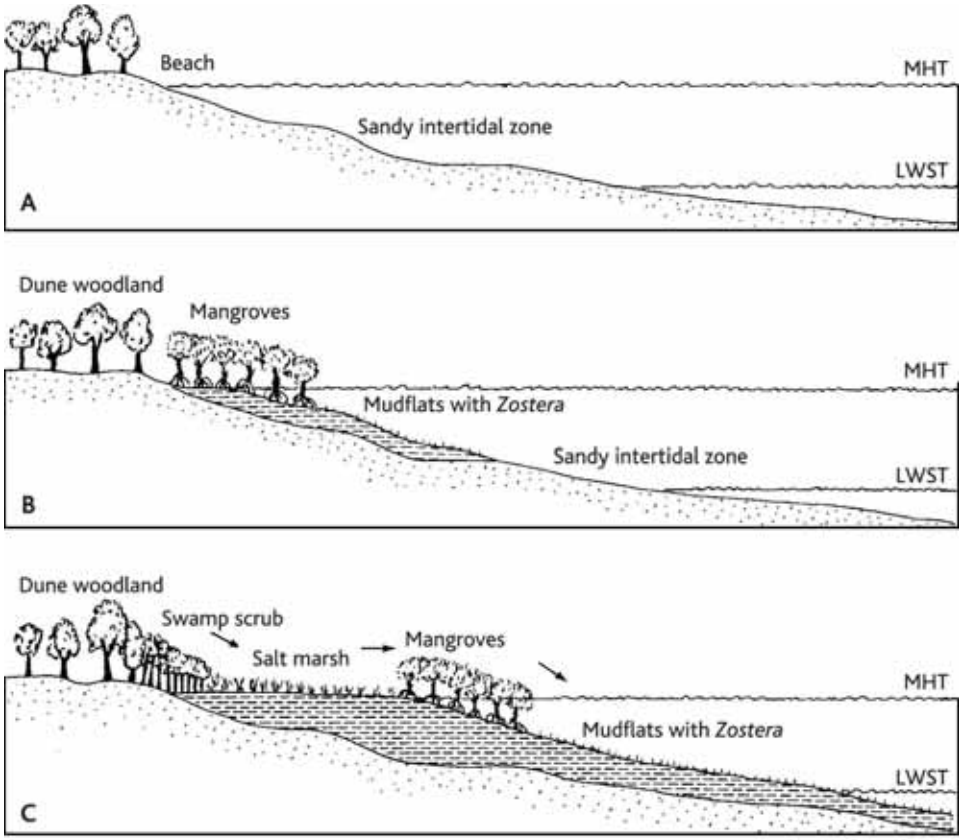


Figure 9.8 Proposed mechanism for the evolution of the mangrove-salt marsh terrace on the northern shores of Western Port. (A) the sandy coast at the end of the Late Quaternary marine transgression; (B) with Holocene mud accretion, a mangrove fringe begins to spread seawards; and (C) as the muddy terrace is built up to mean high tide level, mangroves are displaced by saltmarsh, backed by swamp scrub vegetation. (Source: Bird 1993.)

The widely held view of saltmarshes and mangroves as land builders, however, may not be as valid as is often assumed. Adam (1990) argued that saltmarshes were better seen as taking advantage of sites where sediment deposition was occurring already. In this view, the capacity of saltmarsh vegetation to stabilise sediment against subsequent erosion is the critical process, rather than any putative land-building ability. The process was explained by Morrisey (1995, pages 206–207):

'Saltmarshes begin to form when sediment deposited by rivers or the sea… accumulates to heights above the average level of neap high tides. Under these circumstances, plants start to colonise the sediment, which their roots bind and stabilise. The aerial parts of the plants also retard the movement of water over the sediment, causing sediment to be deposited at an increasing rate. Thus, the mud- and sand-flat becomes higher until eventually it is no longer flooded by even the highest tides.'

Floristics

There is scattered information on the floristics of Victorian saltmarshes, some of which refers specifically to Western Port (e.g. Bridgewater 1975, 1982; Bridgewater & Kaeshagen 1979; Bridgewater *et al.* 1981). Saenger *et al.* (1977) provided the most exhaustive list of Australian saltmarsh plant species, but this list is now considered to be seriously flawed. King *et al.* (1990), for example, argued that it was incomplete because criteria for including species in the upper saltmarsh were inconsistently applied. Moreover, it employed a tripartite geographic split of Victorian taxa into east, central and west areas. This split fails to acknowledge that the central Victorian coast includes rainshadow and non-rainshadow elements, so that it compounds saltmarsh vegetation in the 'dry' saltmarshes of the western shore of Port Phillip Bay with the 'wet' saltmarshes of Western Port; see Barson & Calder (1981) for a description of 'dry' and 'wet' Victorian saltmarshes.

An updated floristic list of saltmarsh plants has been prepared by the Victorian Saltmarsh Study (2011), but it does not allow the selective identification of taxa that occur in Western Port. The state-wide flora sums to 140 indigenous taxa known to occur in Victorian coastal saltmarsh, plus another 118 exotic species. Although Victorian coastal saltmarsh contains in absolute terms a low number of plant species, it is floristically more diverse than coastal saltmarsh in more northerly parts of the country. This is particularly important for Western Port, and it could represent one of the most floristically diverse coastal saltmarshes in the world.

A pattern of decreasing floristic diversity in saltmarshes with decreasing latitude has been commented on by a number of reviewers (e.g. Turner *et al.* 2004; Saintilan 2009a, b; Saintilan *et al.* 2009) and is not limited to Australia (Adam 1994). Specht and Specht (1999), for example, identified about 30 species of plants in Australia saltmarshes at 40oS, compared with fewer than 10 species at latitudes of 10–20oS (Figure 9.9). Similarly, Saenger *et al.* (1977) identified only seven species (belonging to four angiosperm families) for saltmarshes in tropical Australia, but 36 species (belonging to 16 angiosperm families) for Victorian coastal saltmarshes at a latitude of about 35oS. The reasons for such strong latitudinal patterns are not well understood (Adam 1994), but the analysis published recently by Saintilan (2009a) hints strongly at temperature as a controlling factor. He reported that mean minimum temperature explained nearly 80% of the variability in saltmarsh species diversity across Australian bioregions, and suggested that germination and growth of several species may be inhibited by high temperatures. If this is the case, climate change could have serious impacts on Western Port saltmarshes via changes to temperature regimes as well as changes to inundation and salinity regimes.

Exotic taxa

It is often assumed — mostly incorrectly — that, because of their harsh physico-chemical conditions, coastal saltmarshes are immune to weed invasions (e.g. EPA 1996; Laegdsgaard 2006). King *et al.* (1990), for example, argued that while the upper saltmarsh provided habitat for many alien plant species, the potential for them to spread was limited. In New South Wales, coastal saltmarshes have been invaded by a number of aggressive weeds, including **Juncus acutus* and **Cortaderia selloana* (Adam 1994). In northern New South Wales and southern Queensland, **Baccharis halimifolia* is an important weed of coastal saltmarsh.

A report to the Environment Protection Authority of Victoria (EPA 1996) concluded that Western Port saltmarshes 'supported a very low weed problem'. This conclusion may not be correct, and at the very least needs to be validated. In Victoria 118 exotic plant species have been recorded in coastal saltmarsh (Boon *et al.* 2011), including a large number of highly problematic weed species in the upper parts of the saltmarsh, such as **Lophopyrum ponticum*, **Parapholis incurva*, **Hordeum marinum* and **Juncus acutus*. Other common exotic species include **Atriplex prostrata*, **Parapholis strigosa* and **Hordeum marinum*. The very lowest levels of Victorian coastal saltmarsh can be invaded by **Spartina anglica* and **Spartina × townsendii*. One or the other **Spartina* species is known to occur in Western Port, especially in the northern parts around The Inlets and in the western parts around the Bass River, where it is considered a very serious weed and is subject to a vigorous control program by Parks Victoria and Melbourne Water. **Spartina* was considered to pose the greatest direct threat to the wetland ecosystem of Western Port in the management plan for the Western Port Ramsar site (DSE 2003), but it is not clear how this assessment came about.

Boston (1981) provided a detailed history of the introduction of **Spartina* into Australia. What makes **Spartina* infestations different from many other coastal-verge weed problems is that the plant was almost always deliberately, not accidentally, introduced into Australian estuaries. Although in some cases the plants were introduced by government agencies, many of the introductions seem to have been made without quarantine clearance or government sanction (Boston 1981). Williamson (1996) provided an overview of **Spartina* introductions in Victoria and his review has yet to be updated, although more recent maps on its distribution across the state, including in Western Port, are available in the Victorian Saltmarsh Study (Boon *et al.* 2011).

Photo courtesy Michael Keough.

A number of student theses have addressed some aspect of **Spartina* in Western Port. Cowling (2001) compared macrofauna in four habitats near the Bass River: unvegetated mudflats, native saltmarsh, and two forms of **Spartina anglica* infestation. She found that the macrofaunal diversity was not depleted in *Spartina* infestations, and was increased in the **Spartina* that grew in slightly elevated mounds (as opposed to extensive flat swards). In a set of transplant experiments, she showed that the native pulminate gastropod *Ophicardelus ornatus*, which was abundant only in mounded *Spartina*, grew more slowly when moved to nearby mudflats. Conversely, the gastropod *Salinator fragilis*, which was abundant on unvegetated mudflat, did not survive when confined to flat expanses of **Spartina sward*.

Hamilton (2001) reported on some aspects of **Spartina anglica* in Western Port. He found that plants were fertile (hence **Spartina anglica* not **Spartina* × *townsendii*), had spread into a band about 3 km along either side of the Bass River and, on the basis of transplant experiments, could grow out onto the open mudflats of the embayment.

Sumby (2001) reported that there was no difference in the depth of the oxic zone, particle size, organic-matter content or algal pigments (chlorophyll *a* and phaeophytin) between estuarine mudflats in Western Port and areas vegetated with **Spartina*. The macrofauna of both habitats was dominated by a single species of polychaete worm, *Nephtys australiensis*.

It appears that few studies have been undertaken on **Spartina* in Western Port, and that little is known of its environmental impacts. Kriwoken and Hedge (2000) analysed the impacts of **Spartina anglica* on Tasmanian estuaries, and noted the variety of responses of different interest groups to infestations. In the Tamar River the plant is welcomed by some residents and agencies because it was claimed that '*Spartina* infestations significantly improve the navigability of shipping channels by stabilising sediments' (Kriwoken and Hedge 2000, page 575). Some residents preferred the 'green meadows' of **Spartina* over the original brown mudflats. Conversely, others considered it a serious nuisance because it limited public access to the water for recreation and competed for space with aquaculture, especially beds of Pacific Oysters *Crassostrea gigas*. Whether similar views would be held in relation to Victorian systems is unclear, as a significant difference is that saltmarsh in Victoria is often fronted by green bands of mangroves, whereas in Tasmania there are mudflats on the seaward side of saltmarshes.

Overseas studies have shown a wide range of impacts. Callaway and Josselyn (1992) examined the ecological impacts of introduced *Spartina alterniflora* (from the eastern coast) on estuaries of the western coast of the USA (Table 9.3). In many ways the invasion of coastal marshes by **Spartina* species in Victoria is mimicked by the invasion of some North American Spartina marshes by the introduced *Phragmites australis*; see Weis and Weis (2003) for an analysis of the ecological impacts of *Phragmites* infestations in the northern hemisphere.

Table 9.3 Potential ecological effects of *Spartina alterniflora* on estuaries of the west coast of the USA. (Source: modified from Callaway & Josselyn 1992.)

Potential impact	Likely mechanisms
Competitive replacement of native taxa	Higher seed production and germination; more rapid clonal growth
Increased rate of sedimentation	Greater stem densities; larger and more rigid stems
Impacts on food-web structure	Changes in quality and quantity of detritus
Decreased benthic algal productivity	Shading under dense *Spartina* canopy
Impacts on upper salt marsh	Increased production of wrack and deposition in upper marsh
Impacts on habitat quality	Greater stem densities
Impacts on benthic fauna	Greater root/rhizome densities; colonisation of subtidal zones
Loss of foraging areas for shorebirds and waders	Colonisation of bare mud flats; colonisation of subtidal zones

Special features

Size

With a combined area of just over 1000 ha (Table 9.1), there is approximately as much saltmarsh around Western Port as there is mangrove shrubland. The only larger areas of coastal saltmarsh in Victoria are at Connewarre (at the mouth of the Barwon River), around Corner Inlet – Nooramunga, and in the Gippsland Lakes around Lake Wellington, Lake Reeve and Jack Smith Lake.

Ecological intactness

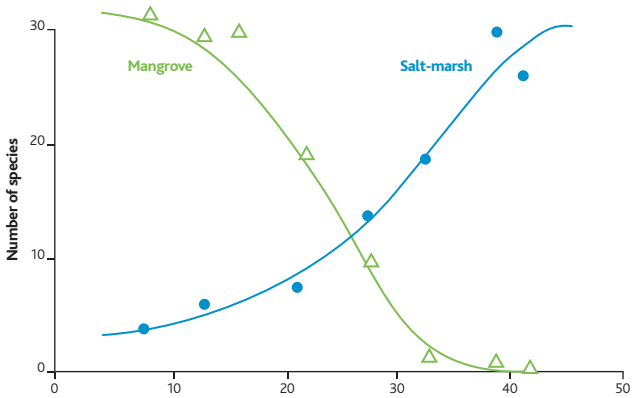
As noted below, much of the original (i.e. pre-European) extent of saltmarsh has been preserved in Western Port (ca 85%), notwithstanding some substantial losses around the Hastings foreshore and marina, and industrial development at nearby Long Point and at The Inlets. The Inlets are part of Koo Wee Rup swamp, and their surrounding Swamp Paperbark-dominated Swamp Scrub and Estuarine Scrub has been almost entirely cleared for agriculture (East 1935; Roberts 1985). A similar point about the relative ecological intactness of the saltmarshes of Western Port was made 15 years ago in a report to EPA Victoria (EPA 1996).

Floristic diversity

Detailed information on floristics is not available, but it is likely that Western Port saltmarshes are floristically diverse, certainly in comparison with coastal saltmarshes at more northerly latitudes in Australia and even more so with northern hemisphere saltmarshes (which are often largely grass-dominated). There are two lines of evidence that support this conjecture.

First, the 192 km^2 of coastal saltmarsh that occur in Victoria represents <1% of the total area of coastal saltmarsh in Australia, but more than 50% of the species of plants found nationally in coastal saltmarshes are found in Victoria (Saintilan 2009a,b). In saltmarshes around Sydney, for example, most contain four plant species or fewer (Morrisey 1995). More southerly coastal saltmarsh almost always supports more plant species than more northerly saltmarshes (Figure 9.9).

Figure 9.9 Relationship between species richness in saltmarshes and mangroves with latitude along the coastline of eastern Australia. (Source: Specht & Specht 1999.)



Second, we repeat the conclusions reached by Opie *et al.* (1984), cited in EPA (1996), that 'Western Port saltmarshes were considered to be very significant in Australia...for a number of reasons' (extensive, floristically rich and relatively undisturbed) and represented 'some of the most significant stands in south-eastern Australia; and are of national importance'.

Conservation status and listed species of plants

In Victoria, native vegetation is allocated a bioregional conservation status (BCS), on the basis of the depletion from its pre-European extent, its rarity, and the intensity of current threats. The highest rating, other than 'presumed extinct', is 'endangered', followed by 'vulnerable', 'depleted', 'rare', and finally 'least concern'. Currently coastal saltmarsh is rated as 'least concern' because at the time of assessment more than 50% of the pre-European extent was thought to remain in the state and the vegetation type was thought to be subject to little to no degradation over a majority of the remaining area. Figure 9.10 shows the BCS for the area between Tooradin and Warneet (compare Figure 9.3).

The Victorian Saltmarsh Study (Boon *et al.* 2011), however, recommended that the BCS for coastal saltmarsh be upgraded to reflect more recent information on the scale of loss since European settlement, and the range and intensity of threats. In the Gippsland Plain bioregion, the Study rated Coastal Dry Saltmarsh as endangered, Coastal Hypersaline Saltmarsh and Coastal Tussock Saltmarsh as vulnerable, Wet Saltmarsh Herbland and Wet Saltmarsh Shrubland as depleted, and Coastal Saline Grassland as rare (Boon *et al.* 2011, Table 6.3). These proposed changes, if accepted, would represent a significant upgrading of the BCS for coastal saltmarsh.

Coastal saltmarsh in New South Wales has, since 1994, been listed as an endangered ecological community under the New South Wales *Threatened Species Conservation Act 1995*.

Special mention should be made of Salt Lawrencia Lawrencia spicata and Yellow Sea-lavender *Limonium australe*, both of which are listed as 'rare' in Victoria and occur in Western Port saltmarshes (DSE 2003).

Conjunction with extensive mangrove shrublands

The saltmarshes of Western Port directly abut extensive mangrove shrublands. Because of the contrary latitudinal variation in mangroves and saltmarshes (Figure 9.9), mangroves are floristically more diverse in the tropics and saltmarshes more diverse in temperate regions. Intimate juxtapositions of mangroves and saltmarsh are not uncommon on a global scale – several parts of the world, including the Gulf Coast of the USA, central Florida, many parts of northern, eastern and southern Australia (e.g. Spencer Gulf in South Australia and in many places along the coast of New South Wales), as well as parts of New Zealand and southern Japan, have an obvious intertidal zone in which mangroves and saltmarshes intermingle (Chapman 1977) – but it is not common to find them so well developed as in Western Port. Moreover, it is not clear

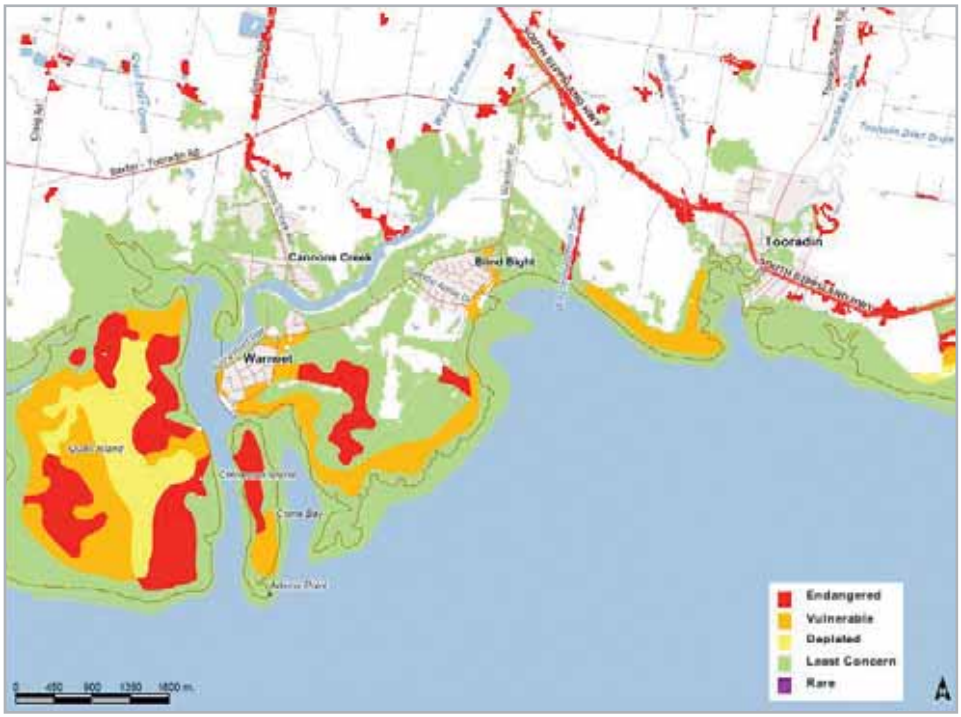


Figure 9.10 Current bioregional conservation status (BCS) for coastal saltmarsh between Tooradin and Warneet, northern shore of Western Port. (Source: DSE biodiversity interactive mapsite, viewed 14 February 2011.)

whether the other locations in Australia where the two plant communities occur adjacently have the same saltmarsh floristic diversity as in Western Port; certainly this is not the case for Northern Hemisphere and northern Australian examples.

Location in protected areas

Coastal saltmarsh (called 'intertidal marsh' in Ramsar nomenclature) is included as one of the three important marine vegetation components (the others being seagrasses and mangroves) in the Western Port Ramsar site (DSE 2003). Note that the fringing wetlands immediately behind the saltmarsh zone, e.g. Swamp Scrub, are not included in the Ramsar site.

There are three Marine National Parks in Western Port — Yaringa (980 ha), French Island (2800 ha)[8] and Churchill Island (670 ha) Marine National Parks — and all contain coastal saltmarsh (Parks Victoria 2007a). Many of the areas of shoreline around Western Port that contain coastal saltmarsh are in Coastal Reserves (e.g. from east of Tyabb to Tooradin, the north-eastern shoreline, and the shoreline near Corinella) and parts are in nature conservation reserves (e.g. around Quail Island and near Bass River).

Support of waterbirds and other fauna

Chapter 12 of this report provides a detailed summary of what is known about the marine mammals and birds of Western Port. As noted in that review, there are good long-term records of waterbird numbers in Western Port (since at least 1973), thanks to the efforts of groups such as the Bird Observers' Club and the Australasian Wader Studies Group. In 2003 the Department of Sustainability and Environment reported that the waterbird record for Western Port was the second-longest series of counts of waterbirds at any coastal wetland in Australia (DSE 2003). Western Port is highly significant for waterbirds in that it supports 29 species listed under JAMBA and 31 under CAMBA (DSE 2003). Not all of these make exclusive use of coastal saltmarsh, but when they are feeding on Western Port's extensive mudflats it is likely that some of their food originated in mangrove and saltmarsh habitats, as discussed below.

It is known that Victorian coastal saltmarsh provides over-wintering habitat for a large number of bird species, including the Eastern Curlew *Numenius madagascariensis*, Common Sandpiper *Actitis hypoleucos*, Red-necked Stint *Calidris ruficollis*, Common Greenshank *Tringa nebularia*, Marsh Sandpiper *Tringa stagnatilis*, Double-banded Plover *Charadrius bicinctus*, Sharp-tailed Sandpiper *Calidris acuminata* and Latham's Snipe *Gallinago hardwickii*. The Blue-winged Parrot *Neophema chrysostoma* and the critically endangered Orange-bellied Parrot *Neophema chrysogaster* feed in saltmarshes along the southern Australian coast. A number of colonial-breeding bird species also use coastal saltmarsh, including the White Ibis *Threskiornis molucca*, Straw-necked Ibis *Threskiornis spinicollis* and Cattle Egret *Ardea ibis*, especially when inland wetlands were in drought. Occasionally large numbers of Black Swans *Cygnus atratus*, Chestnut Teals *Anas castanea* and Australasian Shelducks *Tadorna tadoranoides* feed and roost in coastal saltmarsh. Coastal saltmarsh provides habitat for a number of species of rare or endangered birds. Examples in south-eastern Australia include Lewins Rail *Rallus pectoralis pectoralis*, Royal Spoonbill *Platalea regia*, Great Egret *Ardea alba*, and Orange-bellied Parrot (Spencer *et al.* 2009). Given the paucity of knowledge about the fauna of Western Port's saltmarshes, however, it remains to be discovered which of these species makes use of them for habitat or food.

8 French Island National Park includes areas 150 m seaward of the high-tide mark.

Orange-bellied Parrots deserve special mention because of their critically endangered status in Victoria and their demonstrated heavy dependence on coastal saltmarsh (see also Chapter 12). They are a winter migrant to south-eastern Australia from southern Tasmania (and are one of only two migratory parrot species) and are generally present in Victoria from April to September. The Victorian range of the species extends from east of Corner Inlet in Gippsland to the South Australian border. It is believed that fewer than 50 individuals survive in the wild, making this the most critically endangered bird species in Victoria. Although Orange-bellied Parrots can forage in nearby pastures, coastal saltmarsh is their principal over-wintering foraging habitat in Victoria (Yugovic 1984). It supports many of the key species of food plants needed by the parrots, including Beaded Glasswort *Sarcocornia quinqeflora*, Shrubby Glasswort *Tecticornia arbuscula*, Austral Sea-blite *Suaeda australis*, and other less commonly used species such as Grey Glasswort *Tecticornia halocnemoides* and Southern Sea Heath *Frankenia pauciflora*.

Ecological connectedness with other aquatic systems

Coastal saltmarsh in Western Port cannot be considered in isolation to the other habitats on its landward and seaward sides, especially the extensive mudflats in the middle of the bay. Mazumder *et al.* (2006) analysed gut contents to show that itinerant fish leaving a saltmarsh in Botany Bay (Glassfish *Ambassis jacksoniensis*, Flat-tail Mullet *Liza argentea* and Blue Eye *Pseudomugil signifer*) had fed on crab larvae while they were in saltmarsh, which they moved into during flood tides. More recently, Plattell and Freewater (2009) reported a similar importance of crab zoeae as food for small fish in saltmarshes in the Brisbane Water estuary on the central coast of New South Wales. A similar situation probably occurs also in the saltmarshes of Western Port, the inference being that there are important biological linkages between saltmarshes, mangrove and planktonic environments in Western Port. The research needed to elucidate such interactions, however, has not been undertaken and it remains a significant knowledge gap (see discussion below).

Major threats

Laegdsgaard (2006) provided a review of factors adversely affecting coastal saltmarsh, but her analysis had a strong New South Wales focus and it is debatable whether it can be applied validily to Victoria (e.g. in relation to susceptibility to weed invasions). The Western Port Ramsar site management plan (DSE 2003) identified a number of threats to the Ramsar values of Western Port, including altered water regimes, salinity, pollution (nutrients, heavy metals, and sediments and turbidity), shipping (dredging, oil and marine pests), pest plants and animals (for saltmarshes, most notably **Spartina* and Spiny Rush **Juncus acutus*), resource utilisation, recreation, port development, and erosion.

The Victorian Saltmarsh Study (Boon *et al.* 2011) identifies the following as potential threats to coastal saltmarsh across the state:

- land-claim, infilling, habitat destruction and fragmentation
- fire
- mangrove encroachment
- excessive freshwater inputs (e.g. from stormwater)
- nutrient enrichment and eutrophication
- toxicants
  - oil pollution
  - polycyclic aromatic hydrocarbons
  - halogenated hydrocarbons
  - heavy metals
- acid sulfate soils (potential and active)
- introduced plants (e.g. agricultural weeds in upper saltmarsh; **Spartina* in lower levels and in mangroves)

- grazing
  - domestic stock (e.g. sheep, cattle)
  - feral animals (e.g. European Rabbit *Oryctolagus cuniculus*, Brown Hare *Lepus capensis*, Sambar Deer *Cervus unicolor*, Hog Deer *Cervus porcinus*, and goats *Caprea hircus*, which graze on coastal saltmarsh on French Island.
  - exotic invertebrates (e.g. *Cochlicella barbara, Cornu aspersum, Theba pisana, Cernuella virgata* and *Cernuella vestita*).
- inappropriate mosquito control
  - runnelling
  - insecticides
- recreation
  - disturbance (bait digging, trampling, noise etc)
  - inappropriate infrastructure (e.g. bridges)
  - vehicle access
- inappropriate rehabilitation
  - plants not native to the site
  - plants not occurring naturally in saltmarshes (especially woody plants)
  - modification towards freshwater systems.

A subsequent overview, based on an analysis of the degree and cause of degradation of coastal saltmarsh across the state undertaken during ground truthing of the mapping-inventory component of the Study, identified four threats as particularly important in Western Port (Boon *et al*. 2011, Table 5.7):

- land-claim, infilling, habitat destruction and fragmentation
- vehicle access
- stock grazing — see also comment by Bridgewater (1975) on the effect of stock grazing on upper-level saltmarshes around Western Port
- climate change and sea-level rise.

The two for which we have substantial information (land-claim and climate change) are discussed below. We currently have little information on are vehicle access and stock grazing. During our inspections of saltmarshes across the state, however, it was apparent that many have been used for 'recreational' car use. Parts of saltmarshes at Yaringa and Hastings in Western Port, as well as Connewarre near Geelong and the Lake Wellington saltmarshes near Sale, are criss-crossed with tracks made by 4WDs and motorbikes. There seems to have been little study of the impacts of such disturbances on saltmarsh ecology in Victoria.

Saltmarshes in Victoria are also often subject to cattle grazing. Site visits undertaken during the ground-truthing component of state-wide mapping suggested that grazing access was common across Western Port saltmarshes (Boon *et al*. 2011). The management plan for the Western Port Ramsar site (DSE 2003) specifically identified grazing as damaging to coastal vegetation in the site around Western Port. There is an abundant literature on the adverse impacts of grazing on freshwater wetlands (e.g. Spencer *et al*. 1998; Jensen & Healey 2003; Stanton & O'Sullivan 2006), but little on the impacts on Australian brackish wetlands or saltmarshes.

Land-claim

The term 'reclamation' is often applied to the infilling of wetlands for human use. It is an unfortunate usage (the noted eucalypt taxonomist L.A.S. Johnson in 1973 called it 'a profoundly dishonest word': see Benson *et al*. 1996) as it implies the regaining of land that was originally 'ours'. Some authors (e.g. Strong & Ayres 2009) have enclosed the word in quotation marks to indicate its inappropriateness. The Victorian Saltmarsh Study (2011) proposed that the term 'land-claim' was better suited to what has been euphemistically termed 'reclamation' in the past. Quite independently, the same word was coined by Thomsen *et al*. (2009) to describe the process of loss of New Zealand coastal marshes.

Chapter 3 of this report addressed in broad terms the types of land-use changes and associated loss of habitats that have occurred in Western Port. In the Koo Wee Rup area bordering the northern shore of Western Port, large areas of saltmarsh, mangrove and Swamp Paperbark wetlands were 'reclaimed' for agriculture at the end of the 19th and beginning of the 20th centuries, as described enthusiastically by East (1935) and Roberts (1985). Yugovic and Mitchell (2006) reported on historical changes to the Koo Wee Rup swamp since European colonisation. Bird (1980a,b) reported that much of the mangrove fringe of Western Port has been reduced by clearing, land claim, drainage and dieback. Changes to the shoreline since the early surveys of 1842 and 1865 were evident in 1939 (Bird 1980a,b), and changes have continued apace since then. A boat channel, for example, was cut into mangroves and saltmarsh in 1967 at Yaringa (Bird 1971). Figure 9.11 shows an extreme example of land-claim in Western Port: parts of the original coastal saltmarsh at Hastings were originally used as a Council rubbish tip and then converted into a sports ground. The management plan for the Western Port Ramsar site (DSE 2003) outlined the forces prompting port development in Western Port and likely areas that could be lost as a result, particularly in the Hastings region where large areas of coastal saltmarsh (and mangroves) are found.

Saintilan and Williams (2000) reviewed the records of saltmarsh loss in eastern Australia and concluded that, on the basis of 28 published surveys employing historical aerial photographs, there had been a widespread decline of saltmarsh from estuaries since the 1930s–1940s. Quantitative data on the extent of loss were not available for saltmarsh in Victoria, and almost all of the surveys were for saltmarshes in New South Wales and Queensland. In one of the few Victorian studies, Ghent (2004) compared past and present distributions of coastal saltmarsh in Port Phillip Bay and found that about 65% of pre-European saltmarsh had been lost, mostly before 1978. Gullan (2008) estimated that about 30% of Victorian coastal saltmarsh had been permanently cleared for coastal or marine development.

The Victorian Saltmarsh Study (2011) estimated the areas of mangroves, saltmarsh and estuarine wetland lost in Victoria since European colonisation on the basis of a comparison of modern-day distributions with historical surveyors' maps and contemporary *in situ* evidence of loss.

Figure 9.11 Former coastal saltmarsh at Hastings which is now a sports ground, and previously used as a Council rubbish tip. (Photograph: Paul Boon.)

Some areas of the Victorian coast have suffered substantial losses since the 1840s, most notably Corner Inlet and parts of the Nooramunga coast, Anderson Inlet, Shallow Inlet and Port Phillip Bay, where more than 50% of the original extent had been 'reclaimed'. Other areas, however, have retained most of their pre-European area of intertidal wetland, and in some cases the area has increased. Dr Steve Sinclair, who was largely responsible for this part of the project, estimated that 45% of the pre-European area of saltmarsh in The Inlets had been lost since European arrival, along with 15% from Western Port bay, 5% from Phillip Island, and 10% from the Rhyll coast (Boon *et al.* 2011, Table 6.1). In contrast, the Lang Lang coast of Western Port showed an increase in the extent of coastal saltmarsh since European settlement. At that time, this section of the coast contained virtually no saltmarsh or mangroves, and was fringed instead with a dense stand of Swamp Paperbark. The coast at this time was probably cliffed and eroding (Yugovic and Mitchell 2006). The drainage of the hinterland and the construction of sea walls created a species-poor Wet Saltmarsh Herbland.

Climate change and sea-level rise

Existing reviews of likely impacts of climate change on coastal systems say little specifically about coastal saltmarsh (e.g. Voice *et al.* 2006, DECC 2009, Steffen *et al.* 2009). Even so, it can be predicted that climate change will have many effects on coastal saltmarsh. The likely impacts can be divided into two main types:

- *direct impacts* resulting from higher temperatures and increased ambient CO_2 concentrations
- *indirect impacts* resulting from a rise in mean sea level and increased incidence of extreme events such as storm surges.

Temperature will exert a direct influence via mechanisms such as:

- phenology (the timing of onset of different phases of a plant's or animal's development, e.g. flowering, seed germination, and establishment of seedlings)
- allocation of resources to above-ground and below-ground components (e.g. shoots and leaves versus roots and rhizomes in plants)
- allocation of resources to reproductive versus maintenance activities (e.g. investment in seeds by plants, success of reproduction in animals)
- patterns of life history and longevity (e.g. shortened life spans because of heat stress or drought)
- variations in competitive or other interactions between organisms (Bonan 2002).

Increased temperatures may have consequences also for the spread of exotic taxa in mangroves and coastal saltmarsh. Loebl *et al.* (2006), for example, attributed the recent spread of *Spartina anglica* in parts of the Wadden Sea in the Netherlands to increasing temperature. The sensitivity of native saltmarsh plants to temperature in south-eastern Australia, as reported by Saintilan (2009a), has been noted already.

Altered CO_2 concentrations will have direct impacts on coastal wetlands because of the different photosynthetic mechanisms used by various saltmarsh taxa. Plants fix atmospheric CO_2 in different ways using different metabolic pathways:

- C_3 photosynthesis, the pathway used by most plants for photosynthesis
- C_4 photosynthesis, notably common in warm-season grasses, and advantageous in warmer climates and under water stress
- CAM photosynthesis, which occurs in relatively few taxa but is strongly advantageous under extreme drought stress. It is often associated with succulence.

C_3 plants (most saltmarsh taxa in south-eastern Australia) have high rates of photorespiration and a variable photosynthetic capacity (Bonan 2002). C_4 plants (e.g. grasses such as *Spartina* and *Distichlis*) show little photorespiration and at full sunlight can be twice as productive as C_3 plants. They also use less water to achieve the same rate of primary production as C_3 plants. Altered CO_2 concentrations, therefore, can be expected to have a suite of impacts on the competitive relationships of different saltmarsh plant species. On this basis, Adam (2008) concluded that it was almost certain that there would be changes in the balance between C_3 and C_4 plants in coastal saltmarshes as a result of climate change.

As noted in Chapter 3, the best-available science suggests that, under a 'business as usual' scenario, mean sea levels are likely to increase by 0.8 m by 2100 (Victorian Coastal Council 2008). It is expected that the magnitude of sea-level rise will vary along the coast, depending for example on tidal restrictions and other site-specific factors (see Chapter 4). Western Port, accordingly, has been subject to a number of detailed investigations of the possible impacts rises in mean sea level, extreme events and storm surges (WPGA 2008, Boon *et al*. 2010). Mean sea-level rises of 0.17 m and 0.49 m are projected for Western Port by 2030 and 2070, respectively (WPGA 2008). These are relatively small values when compared with the heightened sea levels that are expect to arise from extreme events that are considered likely under even the most reasonable climate-change scenarios. Storm tides at Cowes (on Phillip Island), for example, could reach 2.29 m by 2030 and 2.74 m by 2070. Not only will storm surges be higher than those currently experienced, but they will occur more often. Storm surges with a current return interval of 100 years would have a new average return interval of only 40 or as little as 6 years by 2030, and 20 years and perhaps as low as 1 year by 2070. In other words, what is currently a severe storm that occurs only once a century could become an annual event by 2070. Linked with the increase in the severity and frequency of storm surges is a projected increase in extreme rainfall and extreme winds (WPGA 2008).

Boon *et al*. (2010) modelled the expected impact on distributions of mangroves and saltmarshes in Western Port with an expected 0.8 m rise in mean sea levels, as is currently used for planning in Victoria (VCS 2008). A simple model based on bathymetry, lidar, altitude limits of plant distributions, and relative wave exposure index demonstrated the broad regions where saltmarsh could migrate with increased sea level, such as the coastal plains around Tooradin, against those where it was impossible because of steep hinterland terrain, such as the San Remo coast. Not considered was population growth in the region, which is likely to further limit any capacity for landward migration of coastal wetlands in currently 'available' areas. Other physical factors that could influence the response of Western Port biota to sea-level rises and storm surges were discussed in Chapters 3 and 4.

Knowledge gaps

Pervasive lack on investment/interest in coastal saltmarsh

The saltmarshes of Western Port suffer from massive knowledge gaps, as indeed can be applied to coastal saltmarsh across Australia more generally. Until the publication in 2009 of *Australian Saltmarsh Ecology* (Saintilan 2009c), the most recent text with substantive sections on Australian coastal saltmarsh was almost 20 years old: Adam's (1990) *Saltmarsh Ecology*. Even so, *Australian Saltmarsh Ecology* has a strong emphasis on New South Wales and southern Queensland, and little is said of Victorian saltmarshes. It is likely that this focus simply reflects the relative amounts of research undertaken on coastal saltmarsh in New South Wales and southern Queensland compared with Victoria and the rest of Australia.

The lack of research investment — and possibly interest — in coastal saltmarsh is indicated also by the presence of separate entries for mangroves (Bridgewater 1999) and seagrasses (Walker 1999) — but not for saltmarsh — in the introductory volume of *Flora of Australia* (Orchard 1999). Maybe they are just not 'attractive' enough, especially in comparison with charismatic ecosystems such as wet forests and even mangroves.

Western Port saltmarshes were the subject of detailed study in the mid 1970s and early 1980s (e.g. Bird 1971; Bridgewater 1975, Bridgewater *et al*. 1981). Little has been published on their ecology in the intervening three decades: the most recent publications I could locate were three of my own — Boon and Cain (1988) and Cain and Boon (1987) on nitrogen–salinity relations of the vegetation, and Boon *et al*. (1997) on the use of d13C and d15N to determine the relative importance of mangroves, saltmarsh and seagrasses as food for burrowing callianassid shrimp in intertidal mudflats on the northern shore of Western Port — and Crinall and Hindell (2004) on the use of Western Port saltmarshes by small fish. Bird (1986) referred to Western Port mangroves, and Vollebergh and Congdon (1986) examined the submerged macrophytes *Ruppia* and *Lepilaena* in saltmarsh pools. Little or nothing has been published on saltmarsh fauna (compare with mangrove fauna: see Chapter 8) in the intervening 15 years to challenge the conclusions reached in a 1996 report to EPA Victoria (EPA 1996) that 'Little is known of the fauna associated with the saltmarshes in Western Port'.

Crucial knowledge gaps

1. Factors controlling the distribution of different plant taxa in coastal saltmarsh, including their relationship with elevation, sedimentation/erosion, and tidal inundation.
   - This information is needed to determine how saltmarsh plants will respond to climate change, especially sea-level rise, and altered rates of sedimentation or coastal erosion that arise from altered catchment practices or run-off. Parks Victoria (2007b), for example, noted that an important hazard to the Yaringa and French Island Marine National Parks was 'lack of knowledge about the way changes in hydrology (including the rate of siltation) impact on flora and fauna…'.
2. Role played by coastal saltmarsh in providing habitat and food (i.e. organic carbon) for saltmarsh fauna, including invertebrates.
   - This information is needed to determine whether coastal saltmarsh provides valuable food resources or habitat for the animals that live in these wetlands, including possibly rare, threatened or endangered species (see also Parks Victoria 2007b).
3. Interactions between saltmarsh and coastal waters in terms of nutrient import/export and fluxes of biota (e.g. crab larvae).
   - This information is needed to determine whether coastal saltmarsh supports non-saltmarsh animals, e.g. by exporting food either as detritus or as living organisms, or supports primary production in adjacent waterways, e.g. by exporting nutrients used by phytoplankton or benthic algal mats.
4. Quantitative descriptions of changes in the extent and distribution of coastal saltmarsh (and mangroves), including studies from the 1840s using surveyors' maps and following World War 2 using aerial photography.
   - This information is crucial for quantifying the loss of saltmarsh that has occurred over recent decades, and thus whether further revisions to the community's biodiversity conservation status are required. It would also demonstrate areas that may benefit from stronger saltmarsh protection or rehabilitation. The management plan for the Western Port Ramsar site (DSE 2003) identified wetland rehabilitation as a recommended activity.
5. Responses of saltmarsh biota to commonplace perturbations such as nutrient enrichment, altered freshwater discharge, hydrocarbon pollution, and grazing.
   - This information is needed to understand how sensitive or resilient coastal saltmarsh is to different stresses, and thus allow managers to build general resilience so that saltmarshes can better withstand future stresses such as climate change and weed infestations (see also Parks Victoria 2007b).

Research to fill knowledge gaps

It is important not to be too prescriptive when describing the types of research needed to fill these knowledge gaps, as different researchers could well devise different approaches to tackle each one. But as a beginning, the following are the types of research directions that could be used to fill the specific knowledge gaps:

1. Factors controlling the distribution of different plant taxa in coastal saltmarsh, including their relationship with elevation, sedimentation/erosion, and tidal inundation:
   - Vegetation profiles along elevational and tidal gradients in saltmarshes around Western Port.
   - Correlation of vegetation profiles with hydrological data (e.g. wetting and drying regimes) and edaphic data (e.g. soil salinity, nutrient status).
   - Quantification of sedimentation/erosion rates.
   - Complementary laboratory/greenhouse experiments on salinity and water regime requirements of saltmarsh plant species.
2. Role played by coastal saltmarsh in providing habitat and food (i.e. organic carbon) for saltmarsh fauna, including invertebrates:
   - Differentiation of the range of habitats provided by coastal saltmarsh (e.g. is the habitat provided by Wet Saltmarsh Herbland equivalent to that provided by Wet Saltmarsh Shrubland for different faunal groups?)
   - Quantitative description of patterns of animal use of different types of coastal saltmarsh.
   - Measurement of the rates of primary production of different saltmarsh plants, including algae (e.g. microphytobenthos on saltmarsh mud flats and saline pools).
   - Analysis (gut analysis and stable isotope analysis) of the structure of saltmarsh food webs.
3. Interactions between saltmarsh and coastal waters in terms of nutrient import/export and fluxes of biota (e.g. crab larvae):
   - Overlaps strongly with the research directions noted about for habitat provision, but with the inclusion of measurements of the export and import of biological (i.e. living), detrital (dead biological) and inorganic (nutrient) fluxes. It will require a careful selection of sites so that these processes can be studied.

Photo courtesy Michael Keough.

4. Quantitative descriptions of changes in the extent and distribution of coastal saltmarsh (and mangroves), including studies from the 1840s using surveyors' maps, and after World War 2 using aerial photography:
   - Two earlier reports have used historical surveys to determine the losses of Western Port peripheral vegetation (Ross 2000, Boon *et al*. 2011). These analyses could be repeated with the specific intention of quantifying losses, gains and floristic changes around different parts of the Western Port coast.
   - The potentially rich resource of historical aerial photographs has not been mined. Similar analyses have proven highly useful in the past for Gippsland wetlands (e.g. Boon *et al*. 2008) and could be applied to Western Port.
5. Responses of saltmarsh biota to commonplace perturbations such as nutrient enrichment, altered freshwater discharge, hydrocarbon pollution, and grazing:
   - Field, glasshouse and laboratory experiments designed to determine the effect of the most important potential pollutants (nutrients, hydrocarbons, fresh water) on saltmarsh vegetation and fauna.
   - Field trials to determine the impact of grazing by sheep and cattle on saltmarshes. The long-term grazing trials in the Victorian Alps could be used as a model.

10 Seagrasses

D I Walker

Seagrasses are unusual aquatic flowering plants that have an important function as ecosystem engineers and are involved in sediment movements, nutrient and energy transfer, and the provision of habitat for a diversity of animals. In Western Port, the most important seagrasses are Zostera *species, which grow on intertidal flats and subtidally in many areas in the northern and eastern parts of the bay. The south-western segment of Western Port has areas of* Amphibolis antarctica*, which may be ecologically important. There was extensive loss of seagrasses in the 1970s, followed by some recovery, but large areas have lost seagrass and not recovered, and recovery has been poor in areas where water quality is poor.*

It is generally agreed that a greater cover of seagrass is desirable, but there are several knowledge gaps preventing us from identifying the best way to achieve this. First, and most practically, we do now know which species of Zostera *is/are present in Western Port, so we do not know the extent to which we can make use of earlier work or use results from Port Phillip Bay in setting management strategies for Western Port. We know that turbidity limits the areas in which seagrasses can grow in Western Port, but we are not sure whether the limits come just from suspended sediments or a combination of sediments and nutrients. Understanding the limits to seagrass would allow us to determine how much water quality needs to improve in Western Port, and which aspects of water quality should be targeted.*

Even if water quality improves, we do not know how seagrasses recolonise suitable habitat, and an important research need is to understand these processes in Western Port.

Temperate seagrass meadows as a habitat are symbolic of many interactions between humans and coastal environments. They are widely acknowledged as critical habitats, playing a major role in sediment stabilisation and a similarly important role in nutrient and energy transfer. Perhaps most significantly, they form an important habitat structure that supports a wide diversity of animals and, in particular, acts as a nursery area for many fish of commercial and recreational importance. Many people value this habitat without any detailed knowledge or direct experience of the biological diversity of seagrass habitats.

At the same time as they are valued, these habitats are threatened by a wide range of human activities, and have declined substantially in many parts of the world. They are affected by habitat clearing, excess nutrients and changes to sediments (see Dennison 2009).

The global decline in seagrasses has led to many reviews, and to very large research programs in areas such as Chesapeake Bay on the eastern coast of North America. Major book-length reviews have been produced by Shepherd *et al.* (1989) and Larkum *et al.* (2006), with a recent update by Dennison (2009). These reviews have included detailed case studies in various parts of Australia, including Western Port. On a more local scale, the Victorian Government has initiated a research program examining resilience of seagrasses in Port Phillip Bay (www.dse.vic.gov.au/coasts-and-marine/marine), including the establishment of water quality thresholds and information on connectivity and resilience. This program was preceded by a formal review of current knowledge of seagrasses in Victoria (Warry & Hindell 2009) and an expert panel (including several of the authors of this review) convened by the Victorian Coastal Council. This information was consolidated into a set of research priorities by an independent experts group (including two authors of this review). These priorities were advertised widely, and an international selection panel recommended two research programs for funding. These programs will address environmental thresholds for seagrasses, including nutrients and sediments, and a connectivity and resilience program.

Because of the volume of past reviews, we have not re-summarised this information here, and the importance of seagrasses to Western Port requires no additional emphasis, given the considerable efforts of organisations such as the Western Port Seagrass Partnership.

In this chapter, we focus on information specific to Western Port, and in particular on knowledge gaps.

Distribution of seagrasses in Western Port

Background

There is some taxonomic confusion about the status of *Heterozostera*. *Heterozostera* has now been sunk into *Zostera* (Jacobs and Les 2009), and in this document will be referred to under that name only. Previous research in Western Port on the biology of *Heterozostera tasmanica* may actually have been on either *Zostera tasmanica* or *Zostera nigricaulis*, but this is impossible to determine because the species are difficult to separate. We have assumed that the research was on the more common *Zostera tasmanica*. Whether *Zostera nigricaulis* behaves differently, and which of the species was studied in Western Port by Bulthuis (1981), are largely unknown.

Seagrasses are important primary producers and providers of habitat, and are widespread within Western Port (Figure 10.2). Four species of seagrass predominate in the bay:

- *Amphibolis antarctica* in the oceanic Western Entrance segment
- *Zostera capricornii* (formerly *Z. muelleri*, Jacobs and Les 2009) on the intertidal mud flats
- *Zostera tasmanica / Zostera nigricaulis* (Figure 10.3) mainly in the shallow subtidal areas and on the lower intertidal mudflats. No recent mapping is available to define where these two species occur, but Z. *nigricaulis* is more common elsewhere in Australia, in deeper more stable conditions where the long black stems are not broken under high energy conditions.

10 Seagrasses

Figure 10.1 Subtidal *Zostera* bed. (Photo: M. Keough.)

Figure 10.2 *Zostera nigricaulis*. (Photo: M. Keough.)

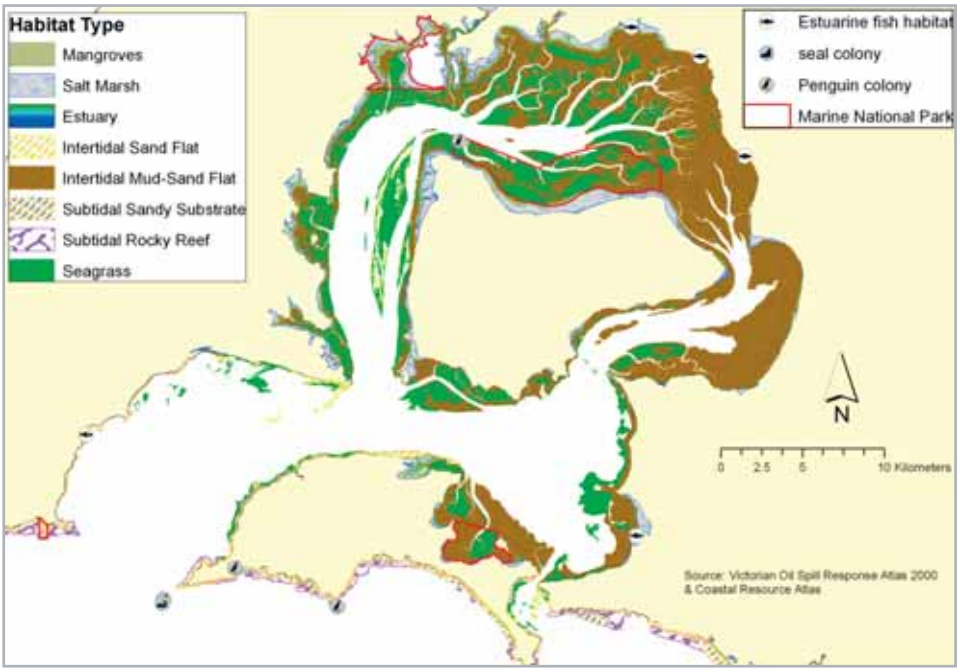


Figure 10.3 Marine habitat map of Western Port showing the extensive intertidal flats that dominate the ecosystem.

The small seagrass Halophila australis also occurs in Western Port, but its distribution is patchy and, compared to *Zostera*, is usually in deeper, darker water.

In the early 1970s Western Port had 250 km^2 of seagrass meadows, covering 37% of the bay and dominated by *Zostera tasmanica* (135 km^2, but see earlier note on taxonomy), *Zostera capricornii* (40 km^2) and *Amphibolis antarctica* (20 km^2) (Bulthuis, 1981; see Table 10.1).

Z. tasmanica dominated the muddy intertidal banks and dendritic channels in the north and eastern regions of the bay, while *Z. capricornii* grew higher in the littoral zone and *A. antarctica* dominated the well-flushed, sandy and exposed southern sections of Western Port (Bulthuis, 1981; Figure 10.4). *Caulerpa cactoides*, a coenocytic green macroalga, was also abundant in the subtidal regions in the eastern region of Western Port (48 km^2) (Bulthuis 1981).

Between the mid 1970s and 1984 the seagrass cover in Western Port fell from 250 km^2 to 72 km^2 (Shepherd *et al.* 1989; Figure 10.5). The greatest losses were in the intertidal banks of the northern and eastern sections of Western Port, and *Zostera tasmanica* was the main species lost (Shepherd *et al.* 1989). The subtidal seagrass meadows in the dendritic channels and in the south-western section of the bay survived much better (Shepherd *et al.* 1989).

Mapping from 1994 (Stephens 1995; Figure 10.6) identified a subsequent increase in the cover of seagrass and macroalgae, with the total area having increased from approximately 59 km^2 in 1983–84 to 93 km^2 in 1994. In 1999 a further increase was observed in Western Port seagrass and macroalgae, increasing to 154.5 km^2.

When Western Port was remapped in 1999–2000, a total area of 154.5 km^2 of seagrass and macroalgae was found (Blake and Ball, 2001). Of this area, 129.7 km^2 (84%) was either seagrass or a mixture of seagrass and algae. The dominant vegetation (in terms of area) was 'Dense *Zostera*/*Heterozostera* with Algae', comprising 43.2 km^2 (28% of the total vegetation mapped). (The two seagrass species could not be distinguished on aerial imagery.) The broad category 'Undefined Algae' covered 24.9 km^2 (16%), and '*Amphibolis* with Macroalgae' covered 20 km^2 (13%).

Comparisons of seagrass areas between different years need to be made with caution, as mapping techniques and field verification methods varied between times and teams.

A qualitative assessment of aerial photography from 1956 to 1999 was conducted for four sites around Western Port to identify patterns of seagrass change (Blake and Ball 2001). This showed a pattern of decline commencing in the late 1970s and continuing through the 1980s, followed by a recovery in the late 1990s. For each site the greatest area of seagrass cover occurred in 1974 or earlier, and the lowest area occurred between 1985 and 1990. This pattern is consistent with the observations of published seagrass studies undertaken in Western Port at the time of the decline.

Table 10.1 shows the marked decrease in seagrass and macroalgae cover between 1973–74 and 1983–84. Over this 10-year period, approximately 70% of the total area of seagrass and macroalgae was lost in Western Port. By 1994 the total area had recovered somewhat, and a further increase was observed in 1999 when the total area covered by seagrass and macroalgae in Western Port was 154.5 km^2.

Table 10.1 Trends in seagrass and macroalgae for Western Port (from Blake & Ball 2001)

Year vegetation mapped	Source	Total area of vegetation (km^2)	Approx. area of vegetation in the Western Entrance (km^2)	Total area of vegetation minus Western Entrance (km^2)
1973–74	Bulthuis 1981	251	21	230
1983–84	Bulthuis 1984	72	13	59
1994	Stephens 1995	113	20	93
1999	MAFRI 1999	154.5	23	131.5

A more detailed analysis of localised aerial photography (Blake and Ball 2001) showed that the data were largely consistent between studies and that patterns varied across Western Port (Figure 10.7 and Table 10.2), with declines during the 1970s and 1980s with some regrowth in the late 1990s.

The most recent maps show some localised increases in some areas of Western Port but declines around Yaringa and in the Corinella segment (Figure 10.8), which are areas of poor water quality where there has been no seagrass recovery or continued declines.

Table 10.2 Summary of changes to seagrass, derived from analysis of aerial photography. (Source: Blake and Ball 2001.)

Aerial photographs					
Year & format	Scale	Corinella	South of Stony Point	Rhyll	Point Leo
1956 B&W	1 : 12 000	■	■ *	■	■
1960 B&W	1 : 15 840	■	■	■	■
1970 B&W	1 : 50 000	■ ˅	■	■ *	■
1973 Colour	1 : 15 000	■	■	■	■
1974 Colour	1 : 15 000	■ ˄ *	■ ˅	■ ˅	■ *
1978 Colour	1 : 10 000	■	■	■ ˄	■
1979 B&W	1 : 25 000	■	■ ˅	■	■ ˅
1981 B&W	1 : 30 000	■	■	■	■ =
1984 Colour	1 : 10 000	■	■	■ ˅	■ =
1985 B&W	1 : 25 000	■ ˅ #	■ ˅	■	■
1988 Colour	1 : 25 000	■ ˄	■	■	■
1989 Colour	1 : 25 000	■	■ ˅ #	■ ˅ #	■
1990 Colour	1 : 15 000	■	■	■	■ ˅ #
1990–2000 Colour	1 : 20 000	■ ˄	■ ˄	■ ˄	■ =

■ Sparse cover (relative to other photographs for the same site)
■ Medium cover (relative to other photographs for the same site)
■ Dense cover (relative to other photographs for the same site)
■ No photograph available for the site and year
˄ Indicates an increase in seagrass cover from previous photograph
˅ Indicates a decrease in seagrass cover from previous photograph
= Indicates that only a small change in distribution had occurred since the previous photograph
# Indicates the year(s) with the lowest seagrass cover for each site
* Indicates the year(s) with the greatest seagrass cover for each site

10 Seagrasses

Figure 10.4 Distribution of seagrass and macroalgae in Western Port in 1974. (Source: Blake and Ball 2001.)

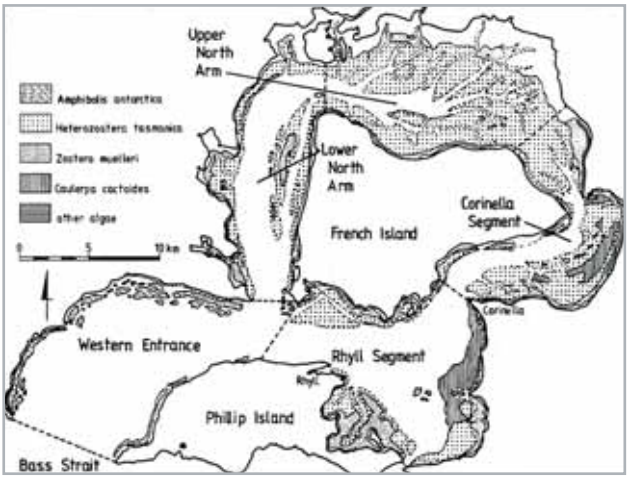


Figure 10.5 Distribution of seagrass and macroalgae in Western Port in 1984. (Source: Blake and Ball 2001.)

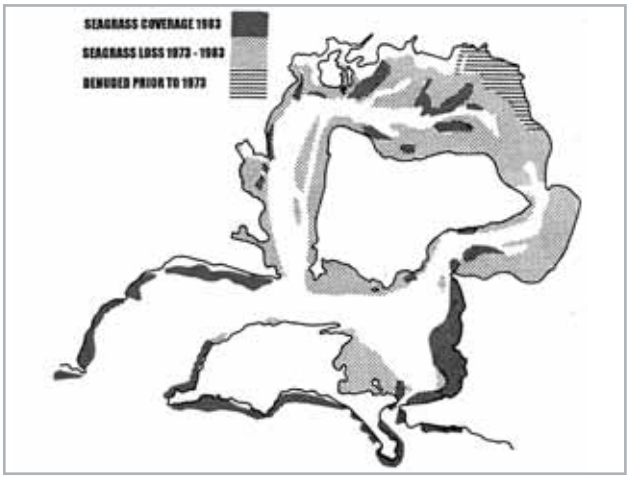


Figure 10.6 Seagrass distribution in Western Port in 1994. (Source: Blake and Ball 2001.)

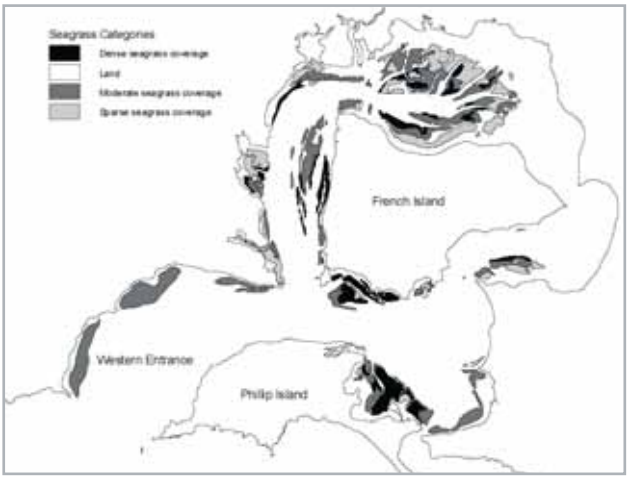
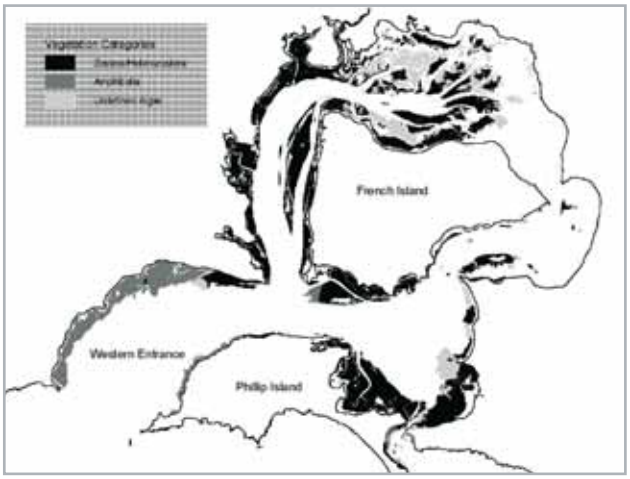


Figure 10.7 Distribution of seagrass and macroalgae in Western Port in 1999. (Source: Blake and Ball 2001.)



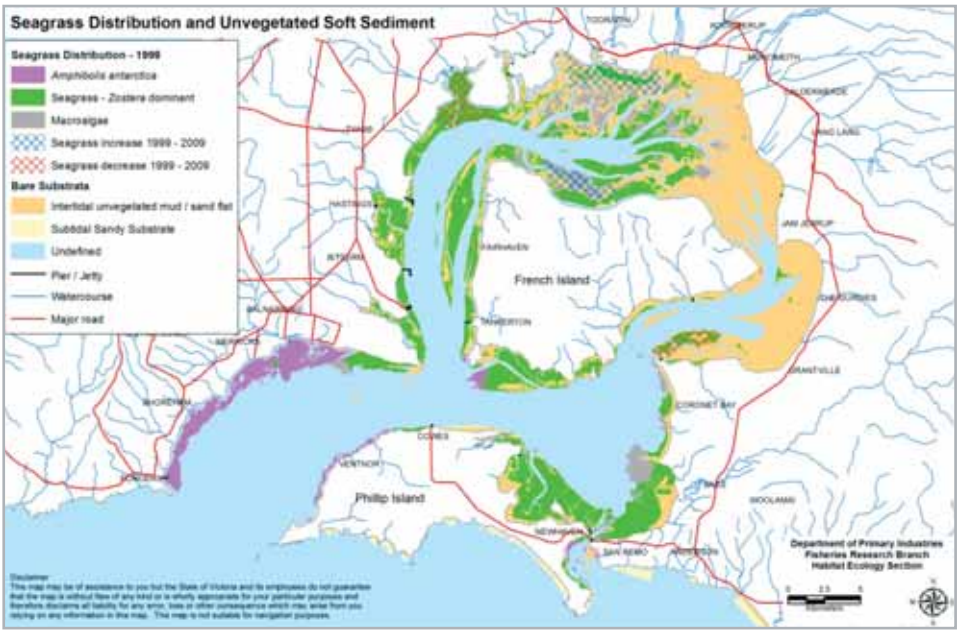


Figure 10.8 Seagrass distributions in 1999, with changes between 1999 and 2009 superimposed, to show losses around Yaringa and the Corinella segment, along with some net increase in the northeastern sections.

Summary of current understanding

Seagrasses rely on light for photosynthesis, and are important primary producers. Seagrasses act as a nursery and refuge for many marine organisms, including seahorses and seadragons (Figure 10.9) and juvenile whiting (Figure 10.10). Leaves have extensive epiphytes, both algae and invertebrates. These epiphytes are a major food source for shellfish, crustaceans and fish. A few animals such as garfish, leatherjackets and swans (Figure 10.11) are adapted to eating and digesting seagrass leaves (see also Chapter 11).

Process of loss

Large areas of seagrasses were lost in Western Port during the 1970s and 1980s because of physical smothering of leaves in shallow water and a consequent reduction in light reaching the seagrass leaves (Bulthuis and Woelkerling 1983b). The leaves developed epiphytic communities (Figure 10.12) of pennate diatoms, filamentous green and encrusting coralline algae, which increased the amount of trapped fine sediment. Within the dendritic tidal channels the water movement was great enough to flush the fine sediments off the leaves of the seagrass. This led to a counter-intuitive situation in which the seagrass in shallow water died but the seagrass growing deeper within the channels survived. The deposited fine sediment raised the bed height of the intertidal flats, thereby reducing water depth and increasing the desiccation and temperature stress for seagrass that might otherwise have regrown in regions of seagrass loss. *Zostera tasmanica* has a higher light requirement at higher temperatures, and is more likely to be negatively affected by reductions in light during the warmer months (Bulthuis, 1983).

Campbell and Miler (2002), Campbell *et al.* (2003) and Miller *et al.* (2005) used chlorophyll fluorescence to examine responses of *Zostera tasmanica* and *Zostera capricornii*, as well as documenting light climate and morphology. They found high photosynthetic activity and efficiency in Z. capricornii compared with *Z. tasmanica* at midday, a response consistent with an adaptation of *Z. capricornii* to high light conditions. Their results were consistent with the restricted distribution of *Z. capricornii* to depths less than 2 m and the ability of *Z. tasmanica* to grow to depths of 8 m throughout Western Port (Campbell *et al.* 2003).

Significantly higher leaf tissue nitrogen content and nitrogen : phosphorus ratios for *Z. tasmanica* were found at Charing Cross and Crib Point compared with Newhaven (Campbell and Miller 2002). Epiphyte nitrogen contents were similar across all sites, but nitrogen : phosphorus ratios were higher at the Charing Cross compared with Newhaven. Tissue phosphorus in seagrass leaves and epiphytes was similar at all sites. The lower tissue nitrogen of *Z. tasmanica* at Newhaven was consistent with the values obtained by Bulthuis and Woelkerling (1981) and suggests that nitrogen is limited at this site. In contrast, tissue nitrogen was higher at the more northern sites and was greater than that reported by Bulthuis and Woelkerling (1981). Campbell and Miller (2002) suggested that nitrogen availability at these latter sites has increased over the past 20 years, and that nitrogen availability and uptake is in excess of growth requirements. Low tissue phosphorus values also did not match those reported by Bulthuis and Woelkerling (1981), who found no phosphorus limitation in Western Port sediments.

Measures of internodes and internodal length indicated that Newhaven and Crib Point populations had older living shoots and relatively low mortality, while plants in the upper North Arm had short nodal lengths and fewer internodes typical of young plants with possibly low growth rates. Plants at Charing Cross form sparse meadows and have small shoots with rapid turnover, suggesting a chronic stress response to poor water quality (Miller *et al.* 2005).

Ecological consequences

The loss of seagrass from intertidal mudflats resulted in an increased flow of suspended solids and nutrients into the water of Western Port (Bulthuis *et al.* 1984). Increased turbidity is likely to have inhibited seagrass growth and prevented recovery as has been reported throughout the world (Waycott *et al.* 2009).

Current status

The mudbanks where the major seagrass losses occurred were still unsuitable for *Zostera tasmanica*, in the late 1990s, when transplants died within weeks from desiccation and smothering (Shepherd *et al.* 1998). A survey in 1994 found that 5000 ha of seagrass had regrown, mostly in the south-eastern section of the inlet, but that there had been little recovery in the north-eastern region (Stephens 1995).

The role of excess epiphyte, macroalgal or phytoplankton growth in shading seagrass leaves and negatively affecting seagrass health is generally agreed to be a prevalent mechanism in seagrass decline worldwide (Walker and McComb 1992, Walker *et al.* 2006, Duarte *et al.* 2008). Morris *et al.* (2007) carried out nutrient addition experiments at three sites in Western Port. The addition of NPK fertiliser increased the ash-free dry weight of seagrass leaves and loose algae at two of the three sites studied. There was also an increase in gammarid amphipod densities at the Crib Point site. The authors concluded that Western Port seagrass habitat was sensitive to increased loads of nutrients within the water column with the Blind Bight region most at risk.

Shepherd *et al.* (2009) reported on a long-term analysis of algae in Western Port. The algal assemblage on Crawfish Rock in northern Western Port was surveyed in 1967–1971 and in 2002–2006. During the 1980s, water quality declined following the large-scale seagrass loss. In 1971 Crawfish Rock had a rich algal flora with 138 recorded species, including 97 species of Rhodophyta. The biomass and cover of canopy and understorey species were measured at sites of strong and slight current on a depth gradient. In 1971, fucoid or laminarian canopy species were dominant from about 1–8 m depth, and an algal understorey extended from the intertidal zone to 12–13 m depth. By 2002–2006 the canopy species extended to only 3 m depth and the algal understorey to about 4 m depth, and 66% of the algal species had disappeared, although a few additional species were present. Persistent, sediment-tolerant species included several phaeophycean canopy species, some chlorophytes (*Caulerpa* spp.) and a few rhodophytes. These findings suggest that a long term shift in light climate in Western Port had taken place, with reduced light availability, increased sedimentation, and unfavourable conditions for photosynthetic organisms.

Figure 10.9 Weedy Seadragon in *Amphibolis* seagrass.
(Photo: M. Keough.)

Figure 10.10 King George Whiting in a seagrass meadow.
(Photo: © Peter Macreadie.)

Major threats

Risks

The major risks to seagrasses – suspended sediments and nutrients – are well known for Western Port from the historical changes that have occurred and the substantial international literature. Globally, most seagrass declines are associated with changing land use and poor water quality (Dennison 2009). Suspended sediments have been seen as an important risk since the 1970's, and high suspended sediment levels have persisted in eastern parts of Western Port. There is some uncertainty about the nutrient status of Western Port (see Chapter 14), with particular questions about Watsons Creek and in the Corinella segment. We have argued elsewhere that work be done to clarify questions about nutrients. It is worth noting that in the most recent seagrass mapping, there appeared to be some decline around Yaringa and little evidence of recovery in the Corinella segment. Both of these are consistent with nutrient enrichment, but these areas, particularly Corinella, also have high suspended sediment loads.

In addition to the effects of nutrients and current sources of suspended sediments, climate change may bring additional risks. In particular, increased temperatures caused by climate change may lead to burning off of exposed seagrasses at low tide, altered physiology of shallow subtidal seagrasses, which could lead to further loss, lack of bank stability and further reduction in light availability.

Seagrasses are also sensitive to a range of toxicants. For Western Port, this risk is unknown (Chapter 3), with little information about the extent to which toxicants spread from their entry points or their bioavailability to seagrass.

Consequences

The previous loss of seagrass in Western Port provides a clear picture of the consequences of further seagrass declines, including increased sediment instability, loss of bank stability, decreased light, and eventual loss of habitat and associated biota following seagrass loss.

Research that can fill key knowledge gaps

The international scientific literature provides clear indications of what is required to reduce seagrass losses or encourage recovery. For Western Port, improvements in water quality are necessary. While a range of actions can be identified, prioritising those actions requires additional information:

- Seagrass taxonomy and genetics
- Absence of a developed nutrient budget for Western Port, and the lack of measures of denitrification
- Present nutrient status of Westernport seagrasses, as results available are over 10 years old
- Accurate estimate of nutrient and sediment levels needed for healthy seagrass
- Factors determining recolonisation by major seagrass species in Western Port
- Accurate assessment of risks posed by toxicants
- Relative importance of different sediment sources (see Chapter 4)

Studies of the biology, reproductive strategies, and environmental tolerances (light, temperature, salinity, and nutrients) of *Zostera* species are needed to understand their resilience to variables such as light reduction, climate change, increased sedimentation and freshwater run-off, to allow us to predict environmental impacts.

Responses to some individual variables in Western Port were published by Bulthuis and Woelkerling (1983a) for *Heterozostera tasmanica* and for *Zostera capricornii* by Clough and Attiwell (1980), but a more advanced approach to modelling the combined effects of variables is now required.

The identities of seagrass species need to be confirmed because the taxonomy has changed since the work in the 1980s, and the identities of the material referred to then are unclear. This research could be linked to Port Phillip Bay research currently underway.

Figure 10.11 Swans grazing on *Zostera muelleri* at Coronet Bay. (Photo: T. Ealey, WPSP.)

Figure 10.12 *Zostera nigricaulis* showing smothering by epiphytic algae. (Photo Mick Keough.)

One of the features that distinguishes seagrasses from other angiosperms is their ability to reproduce while submerged in saline water. Studies of seagrass reproduction and phenology help us understand the contribution of reproduction to the population dynamics of seagrasses. Quantifying flowering, fruiting and seed production is essential in understanding the dispersal and recruitment characteristics of each species, especially for seagrasses such as *Zostera* where seed production is critical to colonisation processes. Studies of reproduction also allow us to delineate seagrass demography and the effects on recruitment, which informs modelling of seagrass population dynamics. Determining flowering frequency, sex ratios and reproductive success also allows the mating system of plants to be determined (Waycott and Sampson 1997), enhancing our understanding of their genetic structure. Reproductive biology may also be critical in the re-establishment of declining seagrass populations and in targeting the best species for use in revegetation (Orth *et al*. 1994).

Seagrasses are a major primary producer in Western Port. In order to identify and manage threats to the bay's seagrass communities, we need to know:

- What is the current state of the Westernport seagrass communities?
- What are the key threats to their long-term survival, productivity and recruitment?
- What are the environmental thresholds for healthy seagrass communities?
- How can the resilience/capacity of seagrass communities be maintained and restored?

Research is required to provide accurate up-to-date and reliable information on local (i.e. Western Port) seagrass biology, including:

i. a synthesis of the current state of the Western Port seagrass communities
ii. new studies of the biology, reproductive strategies, and environmental tolerances of *Zostera* species, addressing the combined effects of different variables
iii. a determination of the present nutrient status of *Zostera* spp., a potential indicator of effects of eutrophication
iv. an assessment of toxicants as a potential hazard for all primary producers, invertebrates and fish communities, to provide baseline data for potential toxic effects.

Justification for project

- There is little understanding of the environmental tolerances of seagrass species in Western Port, particularly in relation to the effects of climate change.
- These projects will provide an understanding of the environmental thresholds of seagrass species, which is needed for developing management strategies (including the off-site management of threats).

Likely benefits for the management of Western Port

- To predict the resilience of the Western Port seagrass community in terms of anthropogenic inputs from land and the direct use of aquatic resources.
- Interpretation of causal pathways to allow predictions by managers of future changes.

11 Fish

Greg Jenkins

Figure 11.1. Elephant fish.
(Photograph © Bill Boyle/OceanwideImages.com)

Western Port has a high diversity and productivity of fish, especially small fish (including juveniles of important fishing species) associated with the extensive seagrass beds. It is an important habitat for pelagic species such as Australian Anchovy and for a number of species of conservation significance, and is a breeding habitat for species such as the Elephant Fish and School Shark. Western Port also supports a very significant recreational fishery. The greatest threat to fish in the bay is loss of habitat, in particular seagrass habitat. Other threats include poor water quality (affecting eggs and larvae), overfishing, and climate change impacts such as increased water temperature.

We identify several research gaps, including research on the relationships between fish and less-studied habitats such as Amphibolis seagrass meadows, studies of eggs and larvae to identify spawning areas and to determine the sensitivity of early life history stages to toxicants, studies of the water quality requirements of estuarine fish, and the continuation and extension of recreational fishing monitoring to ensure the sustainable management of this increasingly important fishery.

Fish around Western Port

Western Port is home to a diverse and abundant array of fish species, primarily because of the extent and diversity of habitats available. The bay is highly productive in terms of small fish species — both bottom-living species associated with habitat such as seagrass meadows (Edgar and Shaw 1995a), and small pelagic fish that school in large numbers (Hoedt *et al.* 1995). These small fish are important for ecosystem function in terms of providing food for higher-order predators such as larger fish (Hoedt and Dimmlich 1994), seabirds and marine mammals (see Chapter 12). Western Port is also home to a number of fish species of conservation significance; the only Victorian records of one species (the Pale Mangrove Goby, *Mugilogobius platynotus*) come from this bay (Hindell and Jenkins 2004, 2005). Western Port is a key breeding area for some species such as Elephant Fish *Callorhinchus milii* (Figure 11.1) (Braccini *et al.* 2008), School Shark *Galeorhinus australis* (Stevens and West 1997) and Australian Anchovy *Engraulis australis* (Hoedt and Dimmlich 1995), and a nursery area for other species such as King George Whiting *Sillaginodes punctatus*, Yellow-eye Mullet *Aldrichetta forsteri* and Australian salmons *Arripis* spp. (Robertson 1978, 1980; Edgar and Shaw 1995a). Although the commercial fishery in Western Port has declined in recent years (DPI 2010), there is an increasingly important recreational fishery of high economic value (Ryan *et al.* 2009). Fish populations in Western Port are highly dynamic and show strong responses to changes in habitat characteristics and water quality (MacDonald 1992, Jenkins *et al.* 1993). Therefore the management of catchment inputs, water quality and habitat characteristics is essential to the ongoing biodiversity and sustainability of the fish fauna of Western Port.

Distribution

The primary factor that determines where fish live in Western Port is the distribution of habitats. Characteristic fish assemblages tend to be related to individual habitats (Edgar and Shaw 1995a, Hindell and Jenkins 2004). This means that fish species that are associated with seagrass, for example, are spread widely throughout the bay, as are pelagic species living in the water column and species associated with unvegetated sediment habitats (Edgar and Shaw 1995a, Hoedt *et al.* 1995).

The distribution of species that enter Western Port from Bass Strait, either as drifting larvae or migrating juveniles or adults, can be biased towards the southern part of the bay. For example, Edgar and Shaw (1995a) found higher fish species richness in seagrass at a site near the entrance, and attributed this to the settlement of juveniles of species normally associated with coastal reefs. Older juveniles and adults of species migrating in from Bass Strait, such as Snapper *Chrysophrys auratus* (Ryan *et al.* 2009), Australian salmons (Hoedt and Dimmlich 1994, Edgar and Shaw 1995a) and Australian Anchovy (Hoedt and Dimmlich 1995, Hoedt *et al.* 1995), also tend to be more common in the southern part of the bay (although in some studies the sampling was biased to the southern area). A higher abundance of these species in the southern part of Western Port may partly reflect higher water quality (e.g. decreased turbidity) (see Chapter 4).

Some species are found in specific localities in Western Port. For example, breeding of Elephant Fish in the soft sediments of the south-eastern embayment plain of the bay is highly predictable and has resulted in heavy targeting by recreational anglers (Braccini *et al.* 2008). Estuarine species are only found in the vicinity of freshwater inputs to the bay and sometimes mainly in specific estuaries. For example, Black Bream *Acanthopagrus butcheri* is abundant in Merricks Creek, while Estuary Perch *Macquaria colonorum* is abundant in the Bass River estuary (Warry and Reich 2010). A significant colony of Weedy Seadragons, *Phyllopteryx taeniolatus*, is found in the Flinders pier area associated with *Amphibolis* seagrass (Stewart *et al.* 2007).

Some species make use of different habitats, depending on their stage of development. For example, juvenile Rock Flathead P*latycephalus laevigatus* tend to be associated with unvegetated sediment while adults are associated with seagrass (Edgar and Shaw 1995a). For some species that are pelagic as adults, such as Australian salmons and Yellow-eye Mullet, juveniles may be associated with shallow seagrass meadows or mudflats (Robertson 1978; 1980).

Figure 11.2. Spotted pipefish. (Photograph: Peter Macreadie.)

Figure 11.3. Weedy seadragon. (Photograph: John Mercer.)

Figure 11.4. Snapper. (Photograph: Daniel Grixti.)

Special features

Conservation significance

A number of fish species in Western Port of conservation concern have been listed under various Acts or placed on internationally recognised lists of threatened species, including:

- Commonwealth *Environment Protection and Biodiversity Conservation Act 1999* ('EPBC Act')
- Victorian *Flora and Fauna Guarantee Act 1988* ('FFG Act')
- International Union for the Conservation of Nature and Natural Resources (IUCN) Red List 2003
- Convention on the International Trade in Endangered Species (CITES), Appendix 2
- Victorian *Fisheries Act 1995*

These species include:

- Pipefish (Figure 11.2) and seahorses, Syngnathidae (EPBC Act, CITES, IUCN Red List (some); Fisheries Act)
- Pale Mangrove Goby (FFG Act)
- Australian Grayling *Prototroctes maraena* (EPBC Act; IUCN Red List; FFG Act)
- School Shark (EPBC 'conservation dependent'; IUCN Red List)
- Great White Shark *Carcharodon carcharias* (EPBC Act; IUCN Red List; FFG Act; Fisheries Act).

Iconic species

The Weedy Seadragon (Figure 11.3), as well as belonging to the syngnathid group of listed species, is also the marine emblem for Victoria.

Recreational and commercial fishing

The importance of commercial fishing in Western Port has decreased in the last decade, firstly with the commercial fishing licence buy-back program that started in 2000, followed by the banning of netting in Western Port in December 2007. The latter action was part of the creation of a 'recreational fishing haven', although commercial long-lining is still allowable in Western Port. Over the past 30 years important commercial species have included King George Whiting, Rock Flathead, Gummy Shark *Mustelus antarcticus*, Australian salmons, Southern Sea Garfish *Hyporhamphus melanochir*, Southern Calamari *Sepioteuthis australis*, Elephant Fish, and Yellow-eye Mullet, although by 2009–10 the commercial catch from Western Port was negligible (DPI 2010).

In contrast to commercial fishing, recreational fishing in Western Port is very popular, with important species including Snapper (Figure 11.4), King George Whiting, Elephant Fish, Gummy Shark, Australian salmons, Rock Flathead, Sand Flathead *Platycephalus bassensis*, Black Bream and Estuary Perch. The recreational effort is likely to increase with population expansion and the introduction of new technologies. For example, the estimated catch of Snapper from Western Port increased from approximately 3000 fish in 2000–01 to 150 000 fish in 2006–07, partly because of these changes and partly because of increased availability as a result of recruitment (Ryan *et al.* 2009). The recreational catch of Elephant Fish in Western Port was estimated to be similar to the entire commercial catch for south-eastern Australia (Braccini *et al.* 2008).

Figure 11.5. Mussel aquaculture.
(Photograph: John Mercer.)

Figure 11.6 Australian anchovies.
(Photograph: Greg Parry.)

Aquaculture

A large aquaculture fisheries reserve was established off Flinders (DPI 2005) after initial environmental baseline surveys (McKinnon *et al.* 2004). The reserve has been used for low-level growing-out of Blue Mussels *Mytilus galloprovincialis* (Figure 11.5), primarily seeded with spat from the Beaumaris aquaculture reserve or more recently from spat cultured at Queensliff by the Fisheries Research Branch, Department of Primary Industries (J. Mercer, Fisheries Victoria, pers. comm.). Blue Mussel culture is likely to continue at a low level only until reserves in Port Phillip are fully utilised, because of high maintenance costs (J. Mercer, pers. comm.). There has also been small-scale experimentation with sea ranching of Blacklip Abalone, *Haliotis rubra*, in cages (J. Mercer, pers. comm.).

A small hatchery where Blue Mussels and abalone have been cultured is located on Phillip Island to the north of the San Remo bridge (J. Mercer, pers. comm.).

Ecosystem function

Clupeoid fishes (Figure 11.6) in Western Port (e.g. Australian Anchovy, Pilchard *Sardinops sagax*, Sandy Sprat *Hyperlophus vittatus*) are small, schooling pelagic fishes that are of low value to fishing but are important components of the food chain that leads to higher order predators such as larger pelagic fish (Hoedt and Dimmlich 1994, Edgar and Shaw 1995b), and potentially seabirds and marine mammals (see Chapter 12).

Summary of current understanding

Fish diversity and abundance

General

As part of the original Westernport Environmental Study, fish were sampled with demersal otter trawls and a 182 m beach seine (Shapiro 1975). Most sites were in the southern half of Western Port, and particular habitats were not targeted other than having a 'hard and flat substratum' (Shapiro 1975). Aggregate indices such as total biomass and cumulative diversity were reported, but not data on species composition or individual species abundances or biomass (Shapiro 1975). Fish diversity in otter trawls was higher in the Rhyll segment than in the western entrance segment or lower north arm (Shapiro 1975). Fish biomass was highest in the lower north arm, intermediate in the Rhyll segment and lowest near the western entrance (Shapiro 1975). The results of the beach seining tended to follow a similar pattern to the otter trawling (Shapiro 1975).

Seagrass habitat

The first major survey of fish in seagrass habitat was carried out at Crib Point in the mid 1970s as part of the Western Port Bay Environmental Study (Robertson 1978). This occurred at about the time of major seagrass loss in Western Port, although Crib Point was less affected than other areas (see Chapter 10). The sampling was conducted on an intertidal mudflat where meadows of *Zostera tasmanica* were covered with pooled water at low tide but unvegetated areas were exposed (Robertson 1978). Samples were collected with either a large (50 × 1 m, 1.27 cm mesh) or small (10 × 1.5 m, 1 mm mesh) seine net (Robertson 1978).

The dominant fish species were either residents — including gobies (Gobiidae), weedfish (Clinidae), Cobbler *Gymnapistes marmoratus* and Greenback Flounder *Rhombosolea tapirina* — or tidal transients — including hardyheads (Athernidae) and Smooth Toadfish T*etractenos glaber* (Robertson 1980). King George Whiting and Yellow-eye Mullet were resident as young juveniles but tidal transient as older juveniles (Robertson 1980). Permanent residents and Smooth Toadfish were more active at night, while Hardyheads, Yellow-eye Mullet and King George Whiting were more active during the day (Robertson 1980).

A broader-scale survey of fish in seagrasses (and unvegetated intertidal flats and channels) was undertaken at sites at Tooradin, Peck Point (French Island) and Rhyll in 1989–90, with additional sampling at Cowes Bank and Loelia Shoal (Edgar *et al.* 1993, Edgar and Shaw 1995a). Sampling was conducted with a seine net (15 × 3 m, 1 mm mesh) and gill nets (50 × 3 m, 64 mm and 108 mm mesh panels) (Edgar *et al.* 1993, Edgar and Shaw 1995a). Seagrass habitats in Western Port supported about twice as many fish species as nearby unvegetated intertidal flats, and four times as many as deeper, unvegetated channel habitat (Edgar and Shaw 1995a). Most fish in seagrass were small (< 10 g weight) and were mainly gobies, pipefish and weedfish (Edgar and Shaw 1995a). The production of small fishes in seagrass was over twice that in unvegetated habitat (Edgar and Shaw 1995a). Larger piscivorous fishes were found in similar abundances in seagrass and unvegetated habitats (Edgar and Shaw 1995a). The abundance of commercially important fishes tended to be higher in seagrass than in unvegetated habitat, and species included Sixspine Leatherjacket *Meuschenia freycineti*, Southern Sea Garfish, Australian Anchovy and Pilchard.

The pattern of fish abundance, and to a lesser extent fish production, was strongly seasonal, with highest levels in summer and a consistent decline through autumn and winter (Edgar and Shaw 1995a). This variation was more pronounced in seagrass than in unvegetated habitats (Edgar and Shaw 1995a) and was consistent with higher seagrass biomass, and higher macro-epifaunal and macrocrustacean production over summer (Edgar *et al.* 1994).

Seagrass beds were sampled at three sites in the north and three in the west of Western Port, as well as one at Rhyll on Phillip Island, in winter 2004 (Hindell *et al.* 2004). The beds of *Zostera nigricaulis* were near the edge of channels in the shallow subtidal zone (Hindell *et al.* 2004). Samples were collected with a seine net (10 × 2.5 m, 1 mm mesh). Most fish were small (< 10 cm) sedentary species such as gobies and pipefish (Hindell *et al.* 2004). The Widebody Pipefish *Stigmatopora nigra* was the most abundant species, while other common species included the Pygmy Squid *Idiosepius notoides*, Spotted Pipefish *Stigmatopora argus* and Halfbridled Goby *Arenigobius frenatus*. Five species of commercial importance were collected — King George Whiting (Figure 11.7), Sixspine Leatherjacket, Australian Anchovy, Grass Whiting *Haletta semifasciata* and Southern Calamari (Hindell *et al.* 2004) — and many of these fish were juveniles (Hindell *et al.* 2004). Species richness in seagrass was higher in closer proximity to mangroves (Hindell *et al.* 2004).

In parallel with sampling natural seagrass beds, Hindell *et al.* (2004) used artificial seagrass units (ASUs, 2 × 1 m as per Jenkins *et al.* (1998)) to sample fish in the upper intertidal area (near the mangrove zone or sandy beach) and the lower intertidal area (approximately 200 m offshore from mangroves or sandy beach) at Wooleys Beach near Crib Point. ASUs minimise the effects of variable seagrass cover on zonation patterns (Hindell *et al.* 2004). Fish abundances did not differ between shore types at the lower intertidal areas, but closer to shore, they were higher adjacent to mangroves than sandy beach (Hindell *et al.* 2004). Species richness showed a similar pattern, with more species adjacent to mangroves than sandy beach (Hindell *et al.* 2004). The species contributing most to these differences were gobies, which were also very common in mangroves (Hindell and Jenkins 2004, 2005). Widebody and Spotted Pipefish were only collected offshore, but the opposite was true for the Smooth Toadfish (Hindell *et al.* 2004). Yellow-eye Mullet juveniles were only collected close to mangroves (Hindell *et al.* 2004).

Mangrove habitat

A major study on fish in mangrove habitats in Western Port was conducted in 2002 (Hindell and Jenkins 2004). Sampling was conducted seasonally in both mangrove and intertidal mudflat habitat at sites located at Warneet, Hastings and Newhaven, and parallel sampling was conducted in Corner Inlet (Hindell and Jenkins 2004). Samples were collected with a beach seine (10 × 2 m, 1 mm mesh), fyke nets (6 mm mesh), and gill nets (35 × 1.5 m; 2.5, 3.8, 5.0, 6.3, 7.6 cm mesh panels). Patterns of association of fish and the two habitats were inconsistent, varying with sampling gear as well as spatially and temporally (Hindell and Jenkins 2004). The number of species collected was slightly higher in unvegetated mudflat habitat (39 versus 37), but 70% of the individual fish were collected in mangrove habitat (Hindell and Jenkins 2004). Most species were found in both habitats, and five species were found only in mangrove and six species only on mudflats (Hindell and Jenkins 2004). Unlike tropical mangrove systems, very few species were resident within mangroves (Hindell and Jenkins 2004).

Some fish collected in mangroves, such as gobies, hardyheads and clupeoids, were also common in seagrass habitat in Western Port (Edgar and Shaw 1995a). Conversely, some fish common in seagrass in Western Port, such as pipefish, weedfish and leatherjackets (Edgar and Shaw 1995a) were not collected in mangroves (Hindell and Jenkins 2004). Mangroves were characterised by greater numbers of small and juvenile fish compared to unvegetated mudflats, but there was little difference in the abundances of older juvenile and adult fish (Hindell and Jenkins 2004), a similar pattern to that found in seagrass (Edgar and Shaw 1995a). As in seagrass (based on seine net sampling), fish abundance and species richness in mangroves in Western Port tended to be lowest in winter (Hindell and Jenkins 2004).

One problem in sampling fish in mangroves is that traditional sampling techniques such as seining, fyke netting and gill netting can only be used along the seaward fringe of the mangrove forest, because of the hard structure of the habitat (Hindell and Jenkins 2004). To overcome this problem, Hindell and Jenkins (2005) used pop nets to sample within the mangrove forest, along the seaward edge of the forest, and on the adjacent mudflat at Wooleys Beach. They found that fish assemblages varied strongly with zone: the mangrove forest was dominated by gobiids and juvenile atherinnids, but the edge and mudflat were characterised by juvenile King George Whiting, Smooth Toadfish, and different goby species (Hindell and Jenkins 2005). Fish abundance was highest in the mangrove forest, whereas species richness was highest at the mangrove edge (Hindell and Jenkins 2005). The most conspicuous species sampled primarily from the mangrove forests, the Pale Mangrove Goby, had not been recorded in previous studies in Western Port, and may be one of the few truly resident species in this mangrove system.

A broad-scale survey of fish in mangroves in Western Port using pop nets was undertaken between May and September 2004 (Hindell *et al*. 2004). Three sites in each of the western, northern and southern (Phillip Island) regions of the bay were sampled (Hindell *et al*. 2004). Samples were taken from within the forest and at the edge of the forest at each site (Hindell *et al*. 2004). As found previously, gobies were the most common fish in mangroves, with six species represented (Hindell *et al*. 2004). Smooth Toadfish, Prickly Toadfish Contusus brevicaudus, and Pikehead Hardyhead Kestratherina esox were also collected, and Sandy Sprat was very abundant at one site (Hindell *et al*. 2004). Patterns of zonation were not as clear as in previous studies, possibly because of differences in sampling methods or the fact that sampling was undertaken in winter rather than summer (Hindell *et al*. 2004).

Pelagic habitat

Pelagic fish were sampled in the south-western region of Western Port in the early 1990s using a combination of trolling for predators (and associated dietary analysis of prey) and seine netting (85 × 3 m; 19 mm mesh wings, 12 mm mesh cod-end) near shore (Hoedt and Dimmlich 1994, Hoedt *et al*. 1995). Pelagic fish caught by trolling along a 65 km transect were primarily East Australian Salmon Arripis trutta and West Australian Salmon *A. truttaceus*, but occasional Snook *Sphyraena novaehollandiae*, Barracouta *Thysites atun*, and Jack Mackerel *Trachurus declivis* were also caught (Hoedt *et al*. 1995). Small pelagic fish collected from gut contents and also in seine net samples were primarily Australian Anchovy, Pilchard and Sandy Sprat. This sampling was conducted before the major Pilchard die-back that affected southern Australia in the mid 1990s (Jones *et al*. 1997).

Sampling of pelagic eggs and larvae of fish has been limited in Western Port, and confined to the southern region. Hoedt and Dimmlich (1995) sampled stations in the south-west of Western Port, as well as off the coast of Cape Schank and Phillip Island, using oblique tows of a 300 micron mesh plankton net. Anchovy eggs and larvae were collected within Western Port and also along the adjacent coast, but Pilchard eggs and larvae were collected only along the coast and in the entrance areas (Hoedt and Dimmlich 1995). In November anchovy eggs and larvae were mainly offshore, becoming much more common inside Western Port in January (Hoedt and Dimmlich 1995).

Acevedo *et al*. (2010) sampled three stations within Western Port (one at the San Remo entrance and two in the south-eastern embayment plain) as well as transects off the coast near Wonthaggi. Samples were collected with surface and near-bottom tows of a 500 micron mesh plankton net (Acevedo *et al*. 2010). Common larvae in Western Port were Australian Anchovy in summer, goby larvae in spring and summer, triplefin (Trypterygiidae) larvae in spring – early summer, and clingfish and shore eel (Gobiesocidae) larvae and Tasmanian Blenny *Parablennius tasmanianus* larvae in spring (Acevedo *et al*. 2010). Many of these larvae belong to species of small fish that live in reef or algal habitats. Apart from Australian Anchovy, larvae of other economically important species such as King George Whiting, Southern Sea Garfish, Grass Whiting and Pilchard were also collected (Acevedo *et al*. 2010).

Unvegetated sediment habitat

In addition to seagrass, Edgar and Shaw (1995a) sampled unvegetated intertidal mudflat and subtidal channel habitat, as well as one subtidal site in the south-eastern embayment plain. Species characteristic of unvegetated intertidal mudflats were the Eastern Bluespot Goby *Pseudogobius* sp., the Tamar Goby *Afurcagobius tamarensis*, Greenback Flounder and Longsnout Flounder *Ammotretis rostratus* (Edgar and Shaw 1995a). Juvenile Rock Flathead were also found on unvegetated mudflat areas (Edgar and Shaw 1995a). Sand Flathead were common in both unvegetated mudflat and channel habitat, while species of stingaree *Urolophus* spp. were most common in channel habitat (Edgar and Shaw 1995a). Elephant Fish were caught in both mudflat and channel habitat, but were most abundant at a silty substrate site within the south-eastern embayment plain (Edgar and Shaw 1995a).

In their study of mangroves and unvegetated mudflats in Western Port and Corner Inlet, Hindell and Jenkins (2004) found that Yellow-eye Mullet, Smooth Toadfish, Silverfish *Leptatherina presbyteroides*, and Longfin Goby *Favonigobius lateralis*, were common species on intertidal mudflat habitat. Apart from Yellow-eye Mullet, juveniles of other commercial species collected included Greenback Flounder, Longsnout Flounder and King George Whiting (Hindell and Jenkins 2004). Further sampling of intertidal mudflats with pop nets in Western Port resulted in the collection of a similar suite of species, but also included the Halfbridled Goby (Hindell *et al*. 2004).

Sampling of subtidal channels and embayment plains with a small beam-trawl was undertaken as part of a PhD thesis on the biology of Red Cod *Pseudophycis bachus* (J. Kemp, unpublished data). Species that occurred frequently in samples included Sand Flathead, Yank Flathead *Platycephalus speculator*, Snapper, Elephant Fish, Red Cod, species of stingaree, Sandy Sprat and Spiky Globefish *Diodon nicthemerus*.

Reef habitat

No information is available on the fish associated with reefs within Western Port. Surveys carried out using underwater visual transects on subtidal reefs near The Nobbies and Cape Woolamai may give an indication of the fish likely to be commonly associated with reefs near the entrance areas (Gilmour *et al.* 2006). These fish assemblages primarily consisted of Bluethroat Wrasse *Notolabrus tetricus*, Purple Wrasse *Notolabrus fucicola*, Herring Cale *Odax cyanomelas* and Sea Sweep *Scorpis aequipinnis* (Gilmour *et al.* 2006). Other common species included the Scalyfin Parma victoriae, Long-finned Pike *Dinolestes lewini*, Zebra Fish *Girella zebra*, and Magpie Perch *Cheilodactylus nigripes* (Gilmour *et al.* 2006). As in Port Phillip Bay (Jung *et al.* 2010), fish assemblages associated with reefs inside Western Port may vary with gradients of environmental factors such as exposure, currents and turbidity.

Estuarine habitat

There is a current program sampling fish in seven estuaries entering Western Port as part of the development of an index of estuarine condition (Warry and Reich 2010). Samples were mainly collected with mesh and fyke nets (Warry and Reich 2010). Common species in mesh nets were West Australian Salmon, Yellow-eye Mullet and Sea Mullet, Black Bream and Estuary Perch (Warry and Reich 2010). Common species in fyke nets were Flathead Gudgeon *Phylipnodon grandiceps*, species of gobies, Smooth Toadfish, and Short-fin Eel *Anguilla australis* (Warry and Reich 2010).

Distinct differences were apparent in the fish assemblages associated with different estuaries entering Western Port (Warry and Reich 2010). For example, significant numbers of Black Bream were only collected in Merricks Creek, and significant numbers of Estuary Perch were collected only in Bass River (Warry and Reich 2010).

Feeding and trophic ecology

Diet analysis

The diet of most fish species, primarily demersal fish, from seagrass and unvegetated habitat in Western Port was dominated by benthic crustaceans (Edgar and Shaw 1995b). The next largest group, including pipefish, hardyheads and clupeoids, ate mainly planktonic crustaceans (Edgar and Shaw 1995b). Another significant group of species, including Yellow-eye Mullet, leatherjackets and gobies, ate algae, sessile animals and seagrass (Edgar and Shaw 1995b). Fish predators were primarily pelagic fish (e.g. Australian salmon and Tailor *Pomatomus saltator*) feeding on small schooling fish, and Rock and Sand Flathead feeding on small demersal fish (Edgar and Shaw 1995b). A few species ate mostly molluscs or polychaetes (Edgar and Shaw 1995b). Among the elasmobranchs, Elephant Fish ate mainly benthic molluscs, and Gummy Sharks ate fish and cephalopods.

The diet of fish in seagrass and unvegetated habitats tended to depend on fish size rather than taxonomic relationships (Edgar and Shaw 1995b). Among crustacean feeders, the diet tended to change with increasing fish size from copepods to amphipods to crabs and shrimps, meaning that larger species showed an ontogenetic change in diet with growth (Edgar and Shaw 1995b). The exception to this was Yellow-eye Mullet, which tended to feed on smaller prey for a given body size compared to other species (Edgar and Shaw 1995b).

Declining condition and increasing mortality rates of fishes in autumn, with a concurrent declining crustacean production, indicates a strong linkage between crustacean production and small fish production (Edgar and Shaw 1995a, Edgar and Shaw 1995b).

Fish species on the intertidal and shallow subtidal seagrass flats at Crib Point had little dietary overlap (Robertson 1980). An exception was Smooth Toadfish and juvenile King George Whiting, which both fed predominantly on Ghost Shrimps *Callianassa australiensis* over summer and early autumn (Robertson 1980). Production of Ghost Shrimp over summer may have been limiting for these fish species (Robertson 1980). The diet of Southern Sea Garfish was highly unusual in that large quantities of seagrass were ingested in daytime but at night the diet was dominated by benthic crustaceans, particularly amphipods (Robertson and Klumpp 1983, Edgar and Shaw 1995b). The emergence of benthic crustaceans into the water column at night (Robertson and Howard 1978) apparently led to a switch in diet to the animal prey to obtain a sufficient overall intake of protein (Klumpp and Nichols 1983, Robertson and Klumpp 1983).

Stable isotope analysis

In addition to direct dietary analysis, significant information on trophic relationships, integrated over a longer time period, can be obtained by studying stable isotopes. This is particularly valuable for determining the ultimate plant source in the food chain, and also the trophic level of fish. Longmore *et al.* (2002) analysed a range of fish species (mainly commercially important ones) from Western Port for stable isotopes of carbon and nitrogen. On the basis of average stable isotope values for each species, the commercial fish were classified into three groups:

- pelagic piscivores (e.g. Australian salmons) with mixed algae as the most important ultimate source of primary production
- pelagic/benthic feeders (e.g. Southern Sea Garfish, Yellow-eye Mullet and Sand Flathead) dependent on *Amphibolis* and *Zostera* epiphytes, zooplankton and green algal detrital webs
- primarily benthic feeders (flounders, Grass Whiting, King George Whiting and Rock Flathead) ultimately dependent on *Zostera* and *Amphibolis* seagrasses.

Photo courtesy Julian Finn.

Changes in $\delta^{13}C$ with length for Southern Sea Garfish and Sand Flathead were consistent with an increasing dependence by older fish on seagrass/benthic microalgae, while changes in $\delta^{15}N$ with length for King George Whiting were consistent with a change in trophic level within the same food web (Longmore *et al.* 2002). Seagrass and/or seagrass epiphytes made significant contributions to the food supply for seven of the eight commercial fish studied, and this was the first study to identify *Amphibolis* as a significant contributor to fish food webs in Western Port (Longmore *et al.* 2002). Conversely, mangrove and saltmarsh made a very minor contribution to fish trophic webs in the bay (Longmore *et al.* 2002).

Hindell *et al.* (2004) used stable isotopes of C and N to investigate the plant basis of the diets of 20 species of commercially and recreationally important fish from four locations in each of Western Port and Corner Inlet. Sampling was conducted in winter and summer (Hindell *et al.* 2004). Seagrass, macroalgae, seagrass epipytes, phytoplankton and benthic microalgae made a similar moderate contribution to fish diets, but mangrove–saltmarsh made a very small contribution (Hindell *et al.* 2004). The Rock Flathead was the only species for which seagrass had the greatest contribution to the diet (Hindell *et al.* 2004). Further analysis of King George Whiting, however, showed that the diet of this species was clearly supported by seagrass (Hindell *et al.* 2004). West Australian Salmon had the highest trophic position, while Southern Sea Garfish had a low trophic position (Hindell *et al.* 2004).

Species of importance to recreational and commercial fishing

King George Whiting

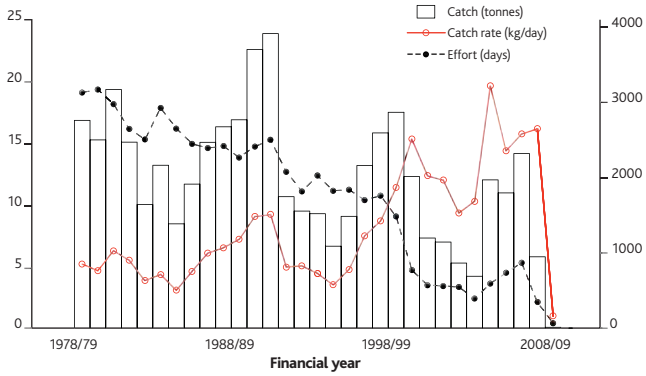
Early studies on King George Whiting focused on basic biology such as age, growth and reproductive state of subadults (Gilmour 1969). Robertson (1977) collected juvenile (0+, 1+ and occasional 2+ age) whiting on the intertidal and shallow subtidal flats at Crib Point. The smallest individuals (post-larvae approximately 20 mm in length) were first collected in seagrass beds in September (Robertson 1977). Young 0+ age whiting were mainly found in areas of dense seagrass, while older fish were found in lightly grassed or unvegetated areas (Robertson 1977). The diet of small 0+ age whiting consisted mainly of harpacticoid copepods, mysids and amphipods, while older juveniles mainly fed on Ghost Shrimps and polychaetes (Robertson 1977).

Otolith microstructure was used to estimate the hatching dates and larval duration of post-larvae collected from seagrass beds at Crib Point, Corinella and Rhyll in Western Port (Jenkins *et al.* 2000). Based on counts of daily increments, the hatching dates ranged from early May to early July, with a median hatching date of May 29 (Jenkins *et al.* 2000). The mean larval duration was 128 days and the mean date of arrival in Western Port was October 7 (Jenkins *et al.* 2000). This information was used in reverse hydrodynamic modelling to estimate that the spawning area for post-larvae arriving in Western Port was from approximately Cape Otway in Victoria to Cape Jaffa in South Australia (Jenkins *et al.* 2000).

The annual commercial King George Whiting catch in Western Port from 2000 to the closure of netting in 2007 ranged between 4 and 14 tonnes (compared with 54 to 95 tonnes in Port Phillip) (DPI 2008). The recreational catch of King George Whiting in Western Port, estimated from off-site telephone surveys in 2000–01 and 2006–07,

was approximately 65 tonnes, mostly from the western and south-eastern areas of the bay (S. Conron, pers. comm.). King George Whiting had shown an increase in CPUE over the decade preceding the closure of netting, indicating increasing abundance of the species (Figure 11.8).

Figure 11.8 Commercial catch, effort, and catch rate (CPUE) of King George whiting in Western Port for the period 1978/79 to 2008/09 (NB. Commercial netting was banned in late 2007).



Snapper

Unlike Port Phillip Bay, Western Port does not appear to be a nursery area for Snapper (Hamer and Jenkins 2004). Intensive sampling with a small beam-trawl (250 hauls) over four years caught only twelve 0+ age Snapper (Hamer and Jenkins 2004), even though adult Snapper are commonly caught by recreational fishers in Western Port over the spawning season (Hamer and Jenkins 2004).

The commercial catch of Snapper in Western Port has always been very small (in the order of a few tonnes), and the majority of Victoria's commercial Snapper catch (approximately 100 tonnes) has come from Port Phillip Bay (DPI 2008). In contrast, the recreational catch of snapper in Western Port is very significant; for example, in 2006–07 the estimated recreational catch was approximately 150 tonnes, compared with 250 tonnes in Port Phillip (Ryan *et al.* 2009).

Australian salmons

Juvenile West Australian Salmon approximately 6 cm in length were collected on the tidal flats at Crib Point in late winter – spring (Robertson 1982). These fish were thought to have been spawned three to six months earlier in Western Australia (Robertson 1982). Juvenile salmon grew over the summer period to approximately 14 cm by autumn, when they left the area (Robertson 1982). This species fed on a range of prey, including fish (e.g. gobies, weedfish, hardyheads), and crustaceans (e.g. amphipods, mysids, shrimp and Ghost Prawns) (Robertson 1982).

Subadult West Australian Salmon ranging from 19 to 34 cm in length were found to feed on pelagic clupeoid fish (Hoedt and Dimmlich 1994). Adult anchovies were the dominant prey in spring – summer, but in late summer – autumn the diet was mainly juvenile clupeoids (mainly anchovies and pilchards) and in late autumn – early winter there was a significant contribution of Sandy Sprat to the diet (Hoedt and Dimmlich 1994). The arrival of west Australian Salmon subadults in Western Port in spring and their gradual departure from the bay in late autumn and winter may be related to the seasonal inshore – offshore migrations of prey species (Hoedt and Dimmlich 1994).

Juvenile East Australian Salmon were first collected at Crib Point in December at a size of approximately 7 cm, most likely after spawning off New South Wales (Robertson 1982). These fish attained a length of approximately 12 cm by the time they left the tidal flat in the following winter. The diet differed from that of the western species in that insects and zooplankton (crab larvae and epitokous polychaetes) were included in the diet in addition to fish and epibenthic crustaceans (Robertson 1982).

Rock Flathead

Rock Flathead have been collected throughout Western Port, with adults occurring in seagrass habitat but small juveniles tending to occur in sand habitat (Edgar and Shaw 1995a). The diet was primarily benthic crustaceans, with a significant contribution of benthic fish (Edgar and Shaw 1995b). The annual commercial catch from 2000 until the closure of netting in Western Port in 2007 ranged between 7 and 11 tonnes (DPI 2008).

Gummy Shark

Edgar and Shaw (1995a) collected Gummy Sharks mainly from one site on the southern side of French Island, where they were evenly distributed across seagrass, unvegetated and channel habitats. Juvenile Gummy Sharks were relatively common in the Rhyll segment of Western Port, indicating that this was an important pupping area (Stevens and West 1997). Gummy Sharks in Western Port fed primarily on benthic decapod crustaceans (Edgar and Shaw 1995b). The annual commercial catch in the 10 years up to 2007–08 ranged between 3 and 9 tonnes (DPI 2008).

Elephant Fish

The Rhyll segment of Western Port appears to be a major breeding area for Elephant Fish (Braccini *et al.* 2008). Adult fish (8–20 years of age) of both sexes are found in mature breeding condition from about February to May (Braccini *et al.* 2008). The highest concentrations of adults are found near Bird Rock (French Island), Coronet Bay and Rhyll (Braccini *et al.* 2008). Females lay eggs in unvegetated muddy sediments, and nearby seagrass beds are likely to be important habitat for neonates (Braccini *et al.* 2008), although it is not clear whether all of the Rhyll segment is important, or if there are particular areas that account for much of the breeding activity (Dr T. Walker, DPI, pers. comm.).

An intensive, localised recreational fishery has developed to target these breeding aggregations (Braccini *et al.* 2008). In 2008 the estimated recreational catch from Western Port was approximately 14 000 fish weighing 45 tonnes, mostly mature females (Braccini *et al.* 2008). It was estimated that the recreational catch for Western Port is of a similar order to the entire commercial catch from the whole of south-eastern Australia (Braccini *et al.* 2008).

Species of Conservation significance

Syngnathids

The most common syngnathids in seagrass on an intertidal mudflat at Rhyll were the Hairy Pipefish *Urocampus carinirostrus* and Port Phillip Pipefish *Vanacampus phillipi* (Howard and Koehn 1985). The Half-banded Pipefish *Mitotichthys semistriatus* was less common, and the Widebody Pipefish was rare (Howard and Koehn 1985). In contrast, the dominant pipefish species collected by Edgar and Shaw (1995a) in seagrass beds was the Widebody Pipefish, and there were moderate numbers of Pugnose Pipefish *Pugnaso curtirostris*, Port Phillip pipefish, Hairy Pipefish, and Spotted Pipefish. In samples from seagrass adjacent to channels, Widebody Pipefish were also the most common syngnathid, and there were moderate numbers of Spotted Pipefish (Hindell *et al.* 2004). Differences in the dominance patterns of pipefish among these studies probably relates to habitat preferences; Widebody Pipefish and Spotted Pipefish are found primarily in subtidal seagrass, whereas species such as Hairy Pipefish are most common in intertidal seagrass (Jenkins *et al.* 1997).

Brooding males of two pipefish species were present over the warmer part of the year: Hairy Pipefish for six months and Port Phillip Pipefish for nine months (Howard and Koehn 1985). Juveniles of both species recruited mainly in summer and early autumn, and both species were annual (Howard and Koehn 1985).

Behavioural observations showed that Port Phillip Pipefish and Half-banded Pipefish orientated themselves horizontally and were relatively strong swimmers (Howard and Koehn 1985). In contrast, Hairy Pipefish and Widebody Pipefish were more sedentary, attaching themselves to vegetation by means of a prehensile tail, thereby mimicking seagrass leaves in movement, orientation and colour (Howard and Koehn 1985). These species were visually orientating sit-and-wait predators, mainly consuming planktonic or epibenthic copepods and small epibenthic amphipods (Howard and Koehn 1985).

Very low numbers of seahorses *Hippocampus* spp. have been collected in Western Port, from both seagrass and unvegetated habitat (Robertson 1978, Edgar and Shaw 1995a), although a significant colony of Weedy Seadragons was observed near Flinders Pier (Stewart *et al.* 2007). Seadragons were mainly associated with the interface between *Amphibolis antarctica* seagrass and unvegetated sand, possibly feeding on dense swarms of krill observed in this habitat (Stewart *et al.* 2007).

Pale Mangrove Goby

The Pale Mangrove Goby has been collected from mangrove habitat at sites around Western Port, including French Island and Phillip Island (Hindell and Jenkins 2004, 2005, Raadik and Hindell 2008, Warry and Reich 2010). This species has not been recorded from mangroves in Corner Inlet (Jenkins and Hatton 2005), Port Phillip Bay (Raadik and Hindell 2008) or the Barwon estuary (Smith and Hindell 2005), suggesting that Western Port may have the only population of the species in Victoria.

Pale Mangrove Gobies live almost exclusively within mangrove forests, so they must be able to withstand periods of exposure as the tide falls. It is uncertain where the gobies go when this occurs, but it is suspected that they bury themselves in the mud, seek refuge in crab burrows, or shelter in small puddles of water beneath mangrove trees (Raadik and Hindell 2008).

Australian Grayling

Eggs and larvae of Australian Graylings were collected from Bunyip River, mostly at the most downstream site (Koo Wee Rup), from May to July (Koster and Dawson 2010). Spawning may be triggered by an increased freshwater flow and decreased water temperature (Koster and Dawson 2010).

A period of marine residency for larvae and young juveniles of Australian Grayling from Bunyip River has been confirmed by otolith microchemistry (Crook *et al.* 2006). The most likely life-history model is that larvae drift downstream into Western Port, or possibly offshore, and return upstream as young juveniles in spring (Crook *et al.* 2006).

School Shark

In a survey of Port Phillip Bay, Western Port and Corner Inlet, the Rhyll segment of Western Port had the highest catch per unit effort of School Shark pups, indicating that the area may be an important nursery for the species (Stevens and West 1997). Small numbers of School Sharks were also collected on unvegetated and channel habitat in this area, and were found to have a diet of fish and cephalopods (Edgar and Shaw 1995b).

Species important to ecosystem function

Clupeoids

Seasonal variations in catches indicate that adult clupeoids are temporary inhabitants in Western Port, migrating into the bay between October and December and leaving between February and June (Hoedt *et al.* 1995). Juvenile Australian Anchovies and Pilchards were common in catches between February and April, indicating that Western Port serves as a nursery area for both species (Hoedt *et al.* 1995). The sizes of adult anchovies and pilchards collected in Western Port were at the lower end of the known range, and these probably represent a single age-group of young adult fish (Hoedt *et al.* 1995). The presence of anchovy eggs in summer indicates that adults were migrating into Western Port to spawn (Hoedt and Dimmlich 1995). The mean density of anchovy eggs in Western Port differed markedly between the two spawning seasons, suggesting that the number of adult fish spawning there can vary between years (Hoedt and Dimmlich 1995).

Major threats

Habitat loss and fragmentation

Risks

There is a high risk of *Zostera* seagrass loss in Western Port (See chapter 10), given the precedent of a major loss of seagrass in the mid 1970s (Shepherd *et al.* 1989). Although some recovery of seagrass cover has been recorded (Blake and Ball 2001), the system would be susceptible to further losses caused by decreased light through factors such as increased sedimentation or nutrients from urbanisation, catchment and coastal development (Chapter 10), or climate change effects such as increased storm activity and sea level rise in areas of coastal hardening (Connolly 2009). Climate change could also lead to increased desiccation and ultraviolet radiation (Connolly 2009). Seagrass loss is also often associated with increased fragmentation of habitat.

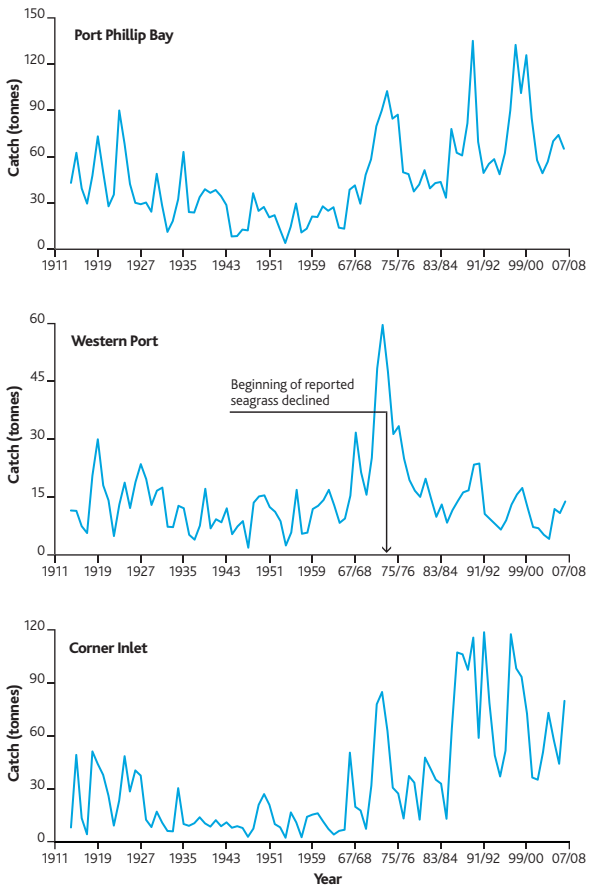
Unlike *Zostera*, the cover of *Amphibolis antarctica* has remained reasonably stable over time (Blake and Ball 2001), possibly reflecting the more consolidated nature of the sediments and clearer water associated with the entrance region. The risk of loss of this habitat therefore appears to be lower than for *Zostera*.

Significant historical loss of mangroves has occurred in Western Port (see Chapter 8), but there are also localised areas of increase, sometimes at the expense of saltmarsh cover (Rogers *et al.* 2005). There is a significant risk of further losses and fragmentation of mangroves, for example caused by increasing coastal development (Chapter 8).

Consequences

The consequences of *Zostera* seagrass loss for fish in Western Port is severe. Edgar and Shaw (1995a) estimated that there was a decline of approximately 630 tonnes in small fish production per year after the seagrass loss in 1973. Falling catches of two commercial species, the Sixspine Leatherjacket and Blue Rock Whiting, were clearly associated with seagrass loss (Edgar and Shaw 1995a). A decline in King George Whiting catches after this period that has also been linked to the seagrass loss (MacDonald 1992, Jenkins *et al.* 1993). Although catches of King George Whiting in Western Port showed similar decadal variability to Port Phillip Bay, the catch trend in Western Port has been downward while catches in Port Phillip and Corner Inlet have increased (Jenkins 2005) (Figure 11.9). These trends were not related to variation in fishing effort (Jenkins 2005). Although older juvenile King George Whiting are often found in unvegetated habitat near seagrass (Edgar and Shaw 1995a), newly settled individuals appear to utilise seagrass habitat (Robertson 1977, Jenkins *et al.* 1997), and, based on stable isotopes, seagrass appears to form the plant basis of the diet of older juveniles (Hindell *et al.* 2004). Loss of *Zostera* would also have severe consequences for important non-economic species, in particular a number of syngnathid species that are closely associated with *Zostera* habitat.

Figure 11.9. Historical catch of King George whiting in (A) Port Phillip Bay, (B) Western Port, (C) Corner Inlet.



At the scale of Western Port as a whole, *Zostera* loss is likely to mean decreased detritus exported from seagrass to other habitats (decreasing productivity) and increased turbidity from resuspension of fine sediments (Edgar and Shaw 1995a). These changes are likely to affect fish production in seagrass as well as unvegetated and pelagic habitats (Edgar and Shaw 1995a).

The consequences of potential *Amphibolis* loss are unknown because of a lack of information on fish in this habitat. It is likely that *Amphibolis* is an important spawning habitat for Southern Calamari (Jenkins 2007; Steer and Moltschaniwskyj 2007), and it may be an important habitat for Weedy Seadragons in Western Port (Stewart *et al.* 2007) although quantitative studies are required. *Amphibolis* also appears to have an important role as the plant basis of the food chain for a number of fish species in Western Port (Longmore *et al.* 2002).

The consequences of mangrove loss to the overall fish community in Western Port would be minor given the lack of distinction between fish assemblages in mangroves and adjacent unvegetated habitat, and the low apparent contribution of mangroves to food chains leading to important fish species (Longmore *et al.* 2002; Hindell and Jenkins 2004; Hindell *et al.* 2004). An exception would be the Pale Mangrove Goby, a species of conservation significance for which the consequence of mangrove loss would be moderate to severe (Hindell and Jenkins 2004, 2005).

Fragmentation of seagrass or mangrove habitat may have consequences other than habitat loss (although the two processes usually occur simultaneously). For example, recent studies on Widebody Pipefish have shown that the preferred habitat is near the edge of seagrass beds, most likely related to the distribution of food resources (Macreadie *et al.* 2010). For this species, therefore, increased fragmentation of seagrass resulting in a greater proportion of edge habitat may be beneficial (Macreadie *et al.* 2009). Differences in the fish assemblage structure among mangrove zones suggest that overall fish biodiversity may increase in fragmented mangrove landscapes as the edge-to-area ratio is increased, but this could detrimentally affect the abundance and species richness of resident mangrove fishes (Hindell and Jenkins 2005).

Water quality

Risks

The highest risk to fish in Western Port in terms of decreased water quality, particularly increased nutrients and sediments, is the secondary effect of seagrass habitat loss (Chapter 10). There are also, however, moderate to high direct risks for fish.

Elevated suspended sediments in Western Port can arise from sediment loads in catchment inputs, and from the resuspension of sediments already in Western Port. The risk of both erosion and resuspension may increase under climate change because of the predicted increase in the frequency of intense storm and rainfall activity (Connolly 2009). Expanding urbanisation and potential port development in the area could also lead to an increase in activities that resuspend sediments.

Elevated levels of suspended sediments have been shown to cause increased mortality of fish eggs and larvae in laboratory experiments (Jenkins and McKinnon 2006). This can also affect physiological processes in fish, such as respiration by clogging gill structures of fish and aquaculture species such as Blue Mussels (Jenkins and McKinnon 2006). Elevated suspended sediment levels and associated turbidity may also affect the behaviour and foraging success of visual predators such as large pelagic fish (e.g. Australian salmons, Tailor), which may in turn affect distribution patterns of these species in Western Port.

Water quality may also be affected by a range of contaminants including heavy metals, organics, pesticides, herbicides and toxicants. These can come from agricultural, industrial and urban sources, and are likely to become more problematic as Western Port and its catchment becomes increasingly developed. The early life stages of fish (eggs, larvae and young juveniles) are the most susceptible to the effects of contaminants, although other effects can occur, such as a deleterious effect of DDT accumulation on reproductive development (Jenkins and McKinnon 2006).

Other risks in relation to water quality include changes to salinity and dissolved oxygen that may particularly affect estuarine habitats. Factors such as lack of freshwater inflow and possible estuarine mouth closure can lead to a lack of mixing and decreased dissolved oxygen in bottom waters (Nicholson *et al.* 2008).

Consequences

The direct consequences of increased suspended sediments would be moderate, most likely leading to some increased mortality of eggs and larvae, decreased physiological performance in fish, and some behavioural and associated distributional changes. These consequences would be unlikely to have a major affect on the viability of fish populations in Western Port. Increased suspended sediments could have moderate consequences for production of Blue Mussels at the Flinders AFR, but the production is already low. The consequences of contaminants for fish may be moderate or higher for species that spawn in the northern section of Western Port where more direct exposure may occur.

The greatest direct consequence of decreased water quality on fish is likely to occur in estuarine habitats. For example, the highest catch rates of Black Bream in Western Port estuaries were in Merricks Creek, but this system was closed off from Western Port and was hypersaline (Warry and Reich 2010). The condition of Black Bream collected from this system was relatively poor (F. Warry, Arthur Rylah Institute, pers. comm.).

Extraction and disturbance

Risks

The risk of overfishing in Western Port has been reduced significantly by the cessation of commercial netting. Offsetting this reduced commercial effort, however, is an increasing recreational angling effort because of an expanding population in the Western Port area and the introduction of better and cheaper of technologies such as echo sounders and GPS navigation systems. The risk of overfishing in the recreational fishery is presently managed using a number of controls including legal minimum length and bag limits. A further risk for a fishery dominated by recreational angling is that monitoring of the catch is much more difficult to implement. While catch per unit effort (CPUE) trends are relatively straightforward to estimate for commercial fisheries based on logbook returns, monitoring CPUE trends for recreational fisheries is more complex.

The total recreational catch can be estimated on a periodic basis, and trends in variables such as size–age distribution of the fished population can be monitored (Ryan *et al.* 2009). There is also ongoing monitoring of CPUE in the recreational fishery for key species in Western Port based on creel surveys and angler diary returns (S. Conron, Pers. Comm.). Fisheries-independent estimates of fish populations based on scientific surveys are particularly important in largely recreational fisheries (Rotherham *et al.* 2007).

Consequences

It is well known that the consequences of overfishing can be severe, including the economic and even biological extinction of a species (Pauly *et al.* 2003). The consequences of overfishing in Western Port will depend on the species in question. For species such as Snapper, King George Whiting and Australian salmons that do not spawn within Western Port, the consequences of fishing may not be as great as for species that breed within the bay. The consequences of overfishing on Elephant Fish in particular may be severe, as the breeding aggregations are targeted intensely by recreational anglers, resulting in a catch equivalent to the entire commercial catch of south-eastern Australia (Braccini *et al.* 2008). The consequences of overfishing may also be high for Gummy Sharks and School Sharks, for which Western Port is an important breeding area.

Temperature increase

Risks

Air temperature is often strongly related to water temperature in estuaries, bays and inlets, and is expected to increase under climate change. CSIRO climate modelling suggests that under medium emissions the best estimate is for a temperature increase of 0.6–1 °C by 2030, rising to 1.5–2 °C by 2070, with the greatest increase (2–2.5 °C by 2070) occurring in summer (CSIRO/BoM 2007). Sea surface temperature off the coast of Victoria under medium emissions is predicted to increase by 0.3 to 1 °C by 2030, rising to 0.6 to 2 oC by 2070 (CSIRO/BoM 2007). The distribution of fish populations, the growth of fish, and the seasonal timing of reproduction is strongly influenced by temperature. Increased temperature, therefore, represents a significant risk to fish populations in Western Port (Jenkins 2010).

Consequences

The consequences of increased temperature are likely to be strongly negative for species in the northern part of their distribution range. For example, Sand Flathead in Victoria are in the northern part of their distribution range, and the population in Port Phillip Bay has declined markedly over the past two decades, corresponding to a period of increasing water temperatures within the bay (Jenkins 2010). Another possible consequence is that species presently not found in Western Port will establish within the bay as temperatures increase, particularly with the anticipated strengthening of the East Australian current (CSIRO/BoM 2007). This would have major consequences for the assemblage structure and ecology of fish in the bay.

Research that can fill key knowledge gaps

Water quality (directly and indirectly, through effects on fish habitat), extraction (largely through recreational fishing), and climate change pose the major risks to fish. Mitigating these threats will require several pieces of scientific information, including fundamental ecological information, particularly about the links between fish and particular habitats, the direct link between aspects of water quality and fish eggs and larvae, and information about life cycles, particularly breeding. Continued collection of information about the nature of recreational fishing is also needed, because this data will assist in determining the status of important fish stocks for ongoing management.

Information about linkages between fish and some important habitats within Western Port is lacking. The importance of *Amphibolis* seagrass beds in Western Port is still largely unknown, although evidence from elsewhere suggests that this seagrass is likely to be an important spawning habitat for Southern Calamari. There is also anecdotal evidence that this seagrass may form an important habitat for the Weedy Seadragon. Quantitative data on fish relationships with *Amphibolis* in Western Port (or elsewhere in Victoria) are also lacking. There is an indication that *Amphibolis* is important in the food chain of many fish species, but data for individual species are lacking (Longmore *et al.* 2002). Thus, predicting the possible threats to fish of potential *Amphibolis* loss and managing accordingly is not possible at present.

Information is also lacking on relationships between reef and algal habitats within Western Port. Although not extensive, there are some important reef habitats within Western Port for which information on fish is lacking (Chapter 13). There are also some reasonably extensive beds of the algae, especially *Caulerpa*, on the eastern side of Western Port (Blake and Ball 2001, J. Kemp pers.comm.) that could provide important habitat for fish. Finally, within the channel and south-eastern embayment plain areas there are isolates of habitat created by sedentary species such as bryozoans and ascidians that may be important habitat for fish (Dr P. Hamer, Fisheries Victoria, pers. comm.).

There is a lack of information on the estuarine and marine phases of Australian Grayling (Koster and Dawson 2010), which is classified as a threatened species. Larvae would be expected to drift downstream to Western Port in autumn, with a return migration of small juveniles in spring. The environmental requirements of these stages in the marine environment are unknown, as is the extent of dispersal of larvae. Therefore threats and corresponding management for mitigation are difficult to determine.

There is no information on the importance of the northern area of Western Port as a spawning area for fish. This area is the most likely to have reduced water quality, and fish eggs and larvae are the developmental stages most susceptible to the effects of poor water quality. There is also no information on the tolerances of fish species (particularly the more vulnerable egg and larval stages) to reduced water quality and to increased temperature associated with climate change. More information on potential impacts on fish eggs and larvae from water quality and climate change is required to inform catchment management.

Photo courtesy Julian Finn.

Information on the distribution, abundance and life history of a number of important fish species is presently lacking in Western Port, including the southern Calamari and the weedy seadragon. An example is the lack of recruitment of juvenile Snapper in Western Port (Hamer and Jenkins 2004); at present it is not known whether this is because large Snapper do not spawn in Western Port or whether the survival of early life stages is low. It would also be helpful to identify any areas that are particularly important for breeding of elephant fish and sharks.

There is still only limited knowledge of the size of the recreational catch and the trend in recreational fishing effort and CPUE in Western Port. The recent ban on commercial net fishing means that reliable commercial CPUE data is no longer available to use as an estimate of population abundance trends for commonly caught species. There is presently no fisheries-independent monitoring of trends in fish populations in Western Port.

One of the important knowledge gaps that can be addressed by research is the relationship between fish and habitats such as *Amphibolis*, *Caulerpa*, subtidal reef and sedentary animal isolates. For some of these habitats, traditional sampling techniques such as seine nets and trawl nets would not be suitable. For example, while seine nets work effectively in *Zostera* habitats where bottom contours are relatively flat, they are unsuitable for sampling fish in *Amphibolis* or subtidal reef habitat where bottom contours can be very uneven and there can be considerable wave surge. Underwater stereo video is a modern technique that is increasingly used for sampling these types of habitats. Stereo cameras allow the length of fish to be easily measured, and cameras can be baited to increase the sample size of fish in view (Murphy and Jenkins 2009). For example, this sampling would provide information on associations association between *Amphibolis* habitat and species such as Southern Calamari and Weedy Seadragon in southern Western Port. This method may be less effective in some parts of northern Western Port because of reduced water clarity, although acoustic techniques can be used to image fish in turbid conditions (Murphy and Jenkins 2009).

Fish egg and larval sampling in Western Port has been very limited and has only occurred in the southern part of the bay. A regular (approximately monthly) sampling program of one to three years duration would provide information on the species that use the bay as a spawning area, as well as the times of year and locations where spawnings occur. This would indicate which species have vulnerable young stages in the water column that could be affected by poor water quality and other factors. The presence or absence of Snapper eggs and larvae would indicate whether the lack of juveniles in Western Port is caused by a lack of Snapper spawning or by poor survival of eggs and larvae (Hamer and Jenkins 2004). Sampling in the northern section of the bay may also provide information on the larval stages of Australian Grayling that would be drifting into the bay over winter months (Koster and Dawson 2010). Egg and larval sampling could be supported by laboratory experiments on the vulnerability of eggs and larvae of important species to exposure to varying levels of water quality parameters such as suspended sediments, contaminants, temperature, salinity and UVB.

In terms of extraction, existing fishery monitoring programs, including creel surveys and angler diaries to estimate CPUE trends, should continue. Fisheries-independent monitoring of important fish species in Western Port could also be implemented to track population changes. For example, the current annual monitoring of post-larval King George Whiting numbers in Port Phillip Bay could be extended to Western Port. This has been found to be correlated with the catch of subadult King George Whiting in Port Phillip Bay approximately three years later (Jenkins 2010). A range of methods could potentially be used for fishery independent monitoring including beam trawls, seine nets and mesh nets, as well as scientific angling targeting juveniles of species such as snapper.

12 Birds and Marine Mammals

Peter Dann

Australian Pied Oystercatcher.
Photo courtesy Annette Hatten.

Western Port is of international significance for aquatic birds. Its importance for birds is reflected in the abundance and diversity of species, the breeding populations of some species in the bay or nearby (some unusually large), its importance as a drought refuge for waterbirds and its use as a non-breeding area for migrant shorebirds from the northern hemisphere and New Zealand.

It makes a significant contribution to Australia's obligations under a suite of international treaties and agreements including the Ramsar Convention for wetland conservation, the Bonn Convention for wildlife conservation, China-Australia Migratory Bird Agreement, Japan-Australia Migratory Bird Agreement, Republic of Korea-Australia Migratory Bird Agreement and the Shorebird Reserve Network for the East Asian-Australasian flyway. It is also designated as part of a global network of Birdlife International's important bird areas. In some contrast, although a variety of marine mammals have been reported in Western Port, it appears to have relatively little importance as marine mammal habitat.

The greatest threats to birds in Western Port are loss of habitat, reductions in food supply through extraction (particularly fish-eating birds) and seagrass loss (most species) and high levels of disturbance from human recreational activity (shorebirds). Habitat loss includes loss of feeding areas and roosting sites through sea-level rise. The greatest threat to the commonest marine mammal in Western Port, the Australian Fur Seal, is sea-level rise greatly reducing the size of their breeding colony.

We identify several research gaps, including better understandings of the decline in fish-eating birds, the relative significance of shorebird and waterbird intertidal feeding areas, the factors involved in roost selection in shorebirds including the role of human disturbance and the effects of sea-level rise on shorebirds and waterbirds.

Birds

Distribution

Western Port has a diverse aquatic avifauna, a reflection of the diversity and productivity of habitats in the region. There are representatives from across the migratory spectrum: from resident species which spend their whole annual cycle in the bay through to those that breed at the far reaches of the northern hemisphere and migrate to and from Western Port each year.

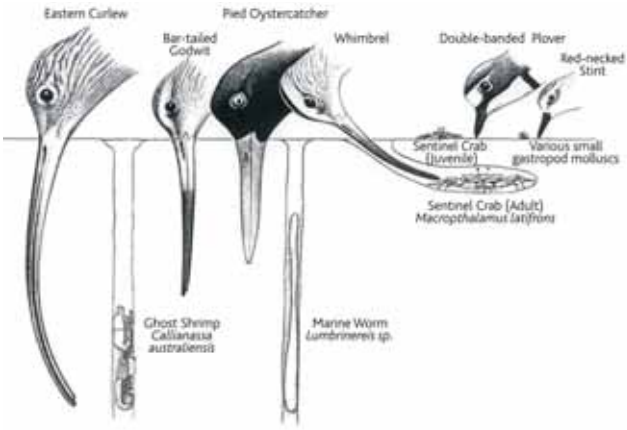
Considerable attention has been given to the distribution and abundance of aquatic birds in Western Port (Loyn 1978, Corrick 1981, Lowe 1982a, Lane 1987, Dann 1993,1994, Dann *et al.* 1994, Loyn *et al.* 1994, Dann *et al.* 2001, Loyn *et al.* 2001, Dann *et al.* 2003, Chambers and Loyn 2006, Dennett and Loyn 2009, Hansen *et al.* in prep.) and earlier information on the fauna of the whole catchment has been summarised by Andrew *et al.* (1984). The shorebirds and waterbirds of Western Port are of special scientific interest because it is one of very few Australian sites where these groups have been counted systematically for over 35 years (Loyn 1975, 1978, Dann *et al.* 1994, Loyn *et al.* 1994, Heislers et al. 2003, Dennett and Loyn 2009). This important work began in 1973 as part of the Westernport Bay Environmental Study (Shapiro 1975) and has been continued as a project of the Bird Observation and Conservation Australia organisation (BOCA), formerly known as the Bird Observer's Club. It is now the longest time series available in Australia for birds frequenting a coastal bay.

In this review the aquatic birds are divided into three groups (shorebirds, waterbirds and seabirds), an artificial classification based partly on phylogeny and partly on foraging guilds. The shorebirds, also known as waders, are made up of species largely from the families Scolapacidae and Charadiidae that forage commonly in intertidal areas in Western Port and breed both locally and elsewhere in Australia and overseas. 'Waterbirds', as used here, comprise a mixed group of species that make up the complement of birds that feed in intertidal areas that are not shorebirds. The group includes ducks, swans, ibises, herons, spoonbills and grebes. 'Seabirds' comprise those species that feed in the marine water column and are, with the exception of gulls, largely piscivorous, and include gannets, terns, cormorants, shearwaters and penguins.

Shorebirds

The 27 000 ha of intertidal mudflat in Western Port is an important habitat for migratory and resident shorebirds (Figure 12.2), being ranked third among shorebird sites in Victoria and among the top 20 in Australia in terms of numbers (Lane 1987, Dann 1994). The average number of migratory Palaearctic shorebirds in Western Port from 1973 to 2002 was 12 100 and, for Australasian shorebirds, was 1420 (Heislers et al. 2003). The abundances of six shorebird species in Western Port meet the criteria for international importance, i.e. maximum counts > 1% of estimated global populations (Watkins 1993) — Eastern Curlew *Numenius madagascariensis*, Common Greenshank *Tringa nebularia*, Red-necked Stint *Calidris ruficollis*, Curlew Sandpiper *Calidris ferruginea*, Double-banded Plover *Charadrius bicinctus* and Pied Oystercatcher *Haematopus longirostris*. Western Port =is also nationally important for the Pacific Golden Plover *Pluvialis fulva* (Watkins 1993). At times it also supports relatively high proportions of the estimated Victorian populations of the Whimbrel *Numenius phaeopus*, Grey-tailed Tattler *Tringa brevipes* and Masked Lapwing *Vanellus miles* in coastal Victoria (Emison *et al.* 1987, Dann 1994).

Figure 12.2 Some of the shorebird guild in Western Port and their feeding depths in the sediment (Dann 1987).



The importance of the bay to shorebirds is recognised internationally by its inclusion in the Shorebird Reserve Network for the East Asian – Australasian Flyway and its designation as one of Birdlife International's Important Bird Areas. In addition, most of the migratory species in the bay are listed under Australia's international migratory bird agreements with Japan, China or South Korea. Twenty-nine species listed under the Japan–Australia Migratory Birds Agreement (JAMBA) and 31 bird species listed under the China–Australia Migratory Birds Agreement (CAMBA) regularly occur in the Western Port Ramsar site (DSE 2003).

Many shorebird species have declined in abundance over the last 30 years, but several species, notably the Pied Oyster-catcher and Red-necked Avocet *Recurvirostra novaehollandiae*, have increased (Heislers et al. 2003, Dennett and Loyn 2009, Hansen *et al.* in prep.). On Phillip Island, the southern boundary of the area being considered here, the intensively managed breeding population of Hooded Plovers is the only one in the world known to be increasing (Dann unpubl. data).

Distribution of shorebirds

The distribution of shorebird roosting sites is more or less evenly spaced around the shorelines of Western Port (see Figure 12.3). Many of the roosts act in tandem, perhaps as alternatives when birds are disturbed at one roost, and most are associated with extensive and adjacent intertidal feeding areas. The roosts are usually undisturbed sites with clear views of approaching terrestrial and aerial predators, and close to intertidal feeding areas with long exposure times. Hansen *et al.* (in prep.) have ranked the roost sites and found that Bunyip River – Yallock Creek had the highest bird abundance of all sites, followed by Barrallier Island.

Figure 12.3 High tide roost sites covered in bird counts in Western Port 1973–2011. The arrows show typical movements of birds from high-tide roosts to feeding areas. (Source: Heislers *et al.* 2003.)

The important feeding areas of shorebirds are not as well known as their roosting sites. Andrew *et al.* (1984) provided maps of feeding areas of shorebirds in Western Port, classified on the basis of primary (national) and secondary (state) importance (Andrew *et al.* 1984, Figure 4). Hansen *et al.* (in prep.) have added more recent information for shorebird feeding areas in the North Arm, the north-western part of the bay and the estuaries around Western Port. They note that the Bass River estuary is of some importance for shorebirds.

Waterbirds

Western Port is used by a suite of waterbirds (Figure 12.4) for breeding and moulting, as well as in non-breeding periods and as a drought refuge (Loyn 1978, Lowe 1982a, 1984, Dann *et al.* 1994, Loyn *et al.* 1994, Dann 2000, Heislers *et al.* 2003, Dennett and Loyn 2009). The more significant of the waterbirds in Western Port in terms of numbers are those that breed colonially (Sacred Ibis and Straw-necked Ibis — the latter does not feed in intertidal habitats — Royal Spoonbill and Australian Pelican *Pelecanus conspicillatus*) and waterfowl (Black Swan, Chestnut Teal and Musk Duck).

Hansen *et al* (in prep.) noted that declines have been particularly marked for the majority of waterbird species in Western Port during the last 12 drought years. The number of Australian Pelicans has decreased since 1974, particularly before the mid 1980s and the loss of seagrass (Dennett and Loyn 2009). Black Swans, the dominant seagrass grazers in Western Port (Dann 2000), declined significantly during the early 1980s with the sudden loss of seagrass. Since then, where seagrass has returned in areas such as Swan Bay to the east of Phillip Island, swan numbers have also returned to 1970 levels. Sacred Ibis abundance has varied extensively since the species first bred in Western Port in the early 1960s. Loyn *et al.* 1994 drew attention to the conspicuous changes in abundance of inland breeding species such as Hoary-headed Grebe, Great Egret, Grey Teal and Musk Duck in Western Port, which were rare or absent in wet years and more abundant in drier years. Over the past 12 years of less than average rainfall, many of the swamps used by ibises and spoonbills for breeding have been dry, and many ibises appear to have shifted to Mud Islands in Port Phillip Bay (Dann pers. obs.). Spoonbills have been observed breeding at farm dams on Phillip Island in recent times, presumably in response to the drying of wetlands elsewhere in Western Port (Dann pers. obs.).

Notably, Loyn *et al.* (1994) observed that the species that declined during the first 10 years of the BOCA counts fed mainly in intertidal areas or were fish-eaters. The waterbird species that increased fed in saltmarsh, fresh water or pasture (Loyn et al 1994).

Distribution of waterbirds

Waterbirds occur all around the bay, and some species such as the Black Swan, White-faced Heron and Sacred Ibis can be found on most intertidal areas of Western Port. This group is less restricted in their requirements for roosting areas at high tide compared to shorebirds. Shorebird high-tide roosts are used by most waterbird species during daylight hours, but many waterbirds roost elsewhere at night. Waterbirds also use a variety of coastal features such as rocky or sandy points, mangroves, saltmarsh, jetties and rocky reefs. Some often roost on the water at high tide (Musk Duck, Chestnut Teal and Black Swan).

Waterbirds are also less restricted in their use of feeding areas in Western Port compared to either shorebirds or seabirds. Many frequent a variety of freshwater and marine habitats as well as pasture. Consequently they are more widespread and more resilient to local environmental change. Lowe (1982a) mapped important feeding areas for ibises, spoonbills and herons.

Figure 12.4 Black Swans.
(Photo: Roz Jessop, Phillip Island Nature Parks.)

Figure 12.5 Crested Terns.
(Photo: Phillip Island Nature Parks.)

Four of the more numerous waterbird species in Western Port (Black Swan, Sacred Ibis, Royal Spoonbill and Chestnut Teal) breed in large numbers in the wetlands of French and Phillip Islands and the Mornington Peninsula. Lowe (1982a) found that numbers of Sacred Ibises and Royal Spoonbills at local colonies exceeded or equalled those counted by air over the entire bay. In contrast, Dann (2000) reported that estimated number of breeding Black Swans in the bay (600) was considerably less than the 10 800 counted by boat over the entire bay. A comprehensive account of the wetlands of French Island and their waterbirds, including some species classified as seabirds here, is given in Quinn and Lacey (1999).

Seabirds

A total of 24 seabird taxa was recorded in a three-year survey of the seabirds of Western Port (Dann *et al.* 2003). The most numerous species by far was the Short-tailed Shearwater *Ardenna tenuirostris* followed by Silver Gulls *Chroicocephalus novaehollandiae*, Little Penguins *Eudyptula minor* and Crested Terns *Thalasseus bergii* (Figure 12.5). All four of these species breed in Western Port and display seasonality in their abundance with peak numbers for most occurring in late summer-early autumn which coincides with the reported influx of juvenile clupeoid fish into Western Port (Dann *et al.* 2003). Most of the biomass recorded along boat transects (average 686 ± 395 kg) was contributed by Short-tailed Shearwaters, Little Penguins and Pied Cormorants *Phalacrocorax varius*. Biomass density (8.5 kg/km^2) was similar to that reported for Port Phillip Bay (8.1 kg/km^2) but lower than off the southern coast of Phillip Island (9.9 kg/km^2).

Several seabird species breed within Western Port or on parts of French and Phillip Islands not strictly covered by this review. There are currently about 28 000 Little Penguins breeding on the Summerland Peninsula at the western end of Phillip Island (Sutherland and Dann in press) and a small colony of less than 10 breeding birds on Barrallier Island (Dann *et al.* 2001). Large numbers of Short-tailed Shearwaters breed on Phillip Island (Harris and Bode 1981) and, since 1961, on Tortoise Head in the south-western corner of French Island (Norman and Gottsch 1968, Peter 1995). The largest Crested Tern colony in Victoria is just south of the review area, at The Nobbies off western Phillip Island (Chiaradia *et al.* 2002). The vulnerable Fairy Tern *Sternula nereis* breeds regularly at Rams Island on the southern coast of French Island and occasionally has been reported breeding at Tortoise Head (French Island) and Observation Point (Phillip Island). Caspian Terns *Hydroprogne caspia* breed in single-pair territories at a number of sites, including Tortoise Head and Rams Island. Silver Gulls breed at various sites on the southern shore of Phillip Island (Harris and Bode 1981). Little Pied Cormorants *Microcarbo melanoleucos* breed in Rhyll Swamp on Phillip Island and at a variety of sites on French Island (Quinn & Lacey 1999) and the Mornington Peninsula. The Pied Cormorant, which has a limited number of breeding sites in Victoria, breeds regularly at Rhyll Swamp (Dann, pers.obs.) and, less commonly, at a few sites on French Island (Quinn & Lacey 1999).

Data available on seabird trends in Western Port come from two sources: estimates of colony sizes of penguins, shearwaters and terns breeding in the vicinity, and long-term counts of cormorants, gulls and terns (Heislers et al. 2003).

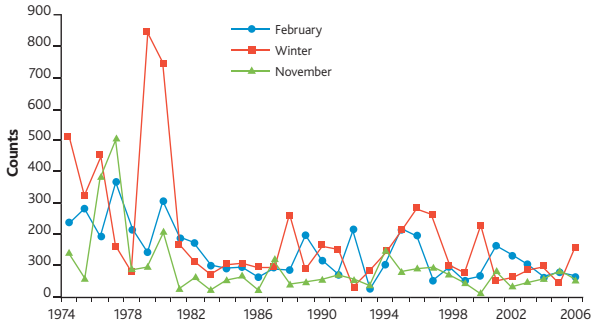
Estimates of colony size of breeding Little Penguins at Phillip Island have shown that the population has almost doubled over the past 25 years in concert with the elimination of all known terrestrial threats to the population (Sutherland and Dann in press). The small colony on Barrallier Island has remained stable for the last 10 years (Dann unpublished data). Numbers of Short-tailed Shearwaters at Tortoise Head and on Phillip Island have increased since the 1960s (Harris and Bode 1981, Peter 1995) but this trend may have changed following some recent years of poor breeding success and high adult mortality (Dann pers. obs.). Crested Tern breeding numbers have been increasing (Chiaradia *et al.* 2002) while Fairy Tern breeding numbers have been declining in recent times (Lacey pers. comm.).

Trends of seabird numbers from the BOCA counts mirror the trends shown in breeding numbers and also provide information on trends of some species for which we have no breeding numbers, such as cormorants and gulls. Silver Gull

numbers were increasing in the bay in the 1970s until active tip management to discourage them was accompanied by a decline to the present stable number (Dennett and Loyn 2009). Kelp Gulls *Larus dominicanus* breed on the southern shore of Phillip Island and particularly at Seal Rocks and have been increasing rapidly (Dann 2007). They will be recorded in increasing numbers in Western Port in the future as the number breeding locally increases.

Some piscivorous seabird species have declined over the past few decades. Little Pied Cormorant numbers crashed in the1980s and 1990s (Figure 12.6), apparently in response to widespread seagrass decline (Loyn *et al.* 1994, Dennett and Loyn 2009) as did Pied Cormorant numbers although they have recovered somewhat (Dennett and Loyn 2009). Little Black Cormorant *Phalacrocorax sulcirostris* numbers vary with patterns of rainfall elsewhere, and the species is rare or absent from Western Port in wet years (Loyn *et al.* 1994).

Figure 12.6 Little Pied Cormorant numbers decreased after seagrass losses in 1980s and have not recovered.
(Source: Dennett and Loyn 2009.)



Distribution of seabirds

The distribution of seabirds within Western Port is not uniform, and some of those that feed by pursuing their prey by diving (pursuit divers), such as cormorants and grebes, are recorded mostly in the shallower eastern and northern arms (Dann *et al.* 2003). The important feeding areas are the subtidal areas in the western, northern and eastern arms, with particular concentrations in the western arm (Dann *et al.* 2003) and the confluence of the three arms (Dann *et al.* 2001).

However, unlike other pursuit-diving species, Little Penguins are found mainly in the western and northern arms of the bay (Dann *et al.* 2001). Relatively few birds were seen in the shallower eastern arm and none over intertidal areas in their three-year study. The highest numbers of penguins per kilometre of transect were found along two transects in the centre of the bay at the confluence of the western, northern and eastern arms (Dann *et al.* 2001). More penguins were found in late autumn and winter, and fewer in mid to late summer. Peak numbers of penguins in Western Port coincide with the latter part of the seasonal occurrence of juvenile pilchards and anchovies in the bay in late summer and autumn (Hoedt *et al.* 1995). Western Port seems relatively unimportant as a feeding area for the 28 000 birds that breed on Phillip Island. In April 1994, when the number of penguins in the bay was greatest (214), it was estimated that the total number of penguins in the bay was 383. Similarly, the mean of all counts (57.4) gave an estimate of 103 birds, which was 1.5% of the estimated breeding population on the Summerland Peninsula on Phillip Island at that time (Dann *et al.* 2001). In contrast to the Phillip Island penguins, it is likely that the birds that breed at the small colony on Barrallier Island (Dann *et al.* 2001) do most of their feeding in Western Port.

Species that seize their prey on the surface of the water (such as albatrosses), surface-plunging species (Crested Terns), shallow-plunging species (Australasian Gannet *Morus serrator*) and pursuit-plunging species (Short-tailed Shearwaters) are more commonly found in the deeper western arm of Western Port (Dann *et al.* 2003).

Other species of particular significance

The saltmarshes of Western Port irregularly support small numbers of the Orange-bellied Parrot *Neophema chrysogaster*, one of the rarest and most endangered parrots in the world (IUCN 2010). Saltmarsh protection throughout the Western Port region, but particularly on French Island and along the northern and eastern sides of Western Port, is the most effective local action for the conservation of this species (see Chapter 9).

Special features

Western Port is of international significance for aquatic birds. It makes a significant contribution to Australia's obligations under a suite of international treaties and agreements including the Ramsar Convention, the Bonn Convention on Migratory Species of Wild Animals, China–Australia Migratory Bird Agreement, Japan–Australia Migratory Bird Agreement, Republic of Korea – Australia Migratory Bird Agreement and the Shorebird Reserve Network for the East Asian – Australasian Flyway.
It is also designated as part of the global network of Birdlife International's Important Bird Areas.

The importance of the bay for birds is reflected in the abundance and diversity of species, the breeding populations of some species in the bay or nearby (some being unusually large), its importance as a drought refuge for waterbirds, and its use as a non-breeding area for migrant shorebirds from the northern hemisphere and New Zealand.

Sites of National Zoological Significance

Most of the important roosting sites in Western Port for shorebirds are listed as Sites of National Zoological Significance (Andrew *et al.* 1984). The exception is Rhyll Inlet – Observation Point which, although it did not qualify for listing as a major or significant roosting site in 1984, would probably do so now, particularly because of its importance for larger shorebirds (Hansen *et al.* in prep.).

Fairy Tern.
Photo courtesy Annette Hatten.

Barrallier Island

This is small island off the north-western corner of French Island, where a small colony of Little Penguins breed (Dann *et al.* 2001). Orange-bellied Parrots have been recorded there, and some of the highest numbers of shorebirds have also been counted on the island (Western Port Shorebird Site, DNRE 2000).

Tortoise Head

Tortoise Head is a headland off the south-western corner of French Island. It is one of the most important high-tide roosts for shorebirds in Western Port, and the most important for the Mongolian Plover *Charadrius mongolus* and Eastern Curlew (Western Port Shorebird Site, DNRE 2000). It is also the site of a Short-tailed Shearwater breeding colony. Caspian Terns, Fairy Terns and Pied Oystercatchers have been reported breeding there.

Rams Island

This small island off the southern coast of French Island is an important breeding site for Fairy Terns (vulnerable in Victoria) and one of the top five more important roosting sites in Western Port.

Mouth of Yallock Creek

This site is at the northern side of the bay and is one of the major roosts for shorebirds in Western Port.

Settlement Road

This site is in the north-eastern corner of the bay and was an important shorebird roost in the 1970s but less important since. Orange-bellied Parrots have been recorded there.

Reef Island

This island and adjacent shore just north of the mouth of Bass River is an important roosting site for shorebirds, waterbirds and resident seabirds (cormorants). Orange-bellied Parrots have been recorded on the shore adjacent to island.

Importance as a monitoring and scientific asset

Western Port is of special scientific interest because it is one of very few Australian coastal sites where shorebirds and waterbirds have been counted systematically over a very long period (4–5 times each year for over 35 years) (Loyn 1975, 1978, Dann *et al.* 1994, Loyn *et al.* 1994, Heislers *et al.* 2003, Dennett and Loyn 2009, Hansen *et al.* in prep.). It is certainly the largest marine bay surveyed in this much detail for this period of time in Australia, and the vast amount of data allows analyses to be made of avian demographic trends with trends in climate (Chambers and Loyn 2006) and human interactions at local, regional and continental scales. Its role in monitoring climate change effects in coastal areas of south-eastern Australia will be invaluable.

Summary of current understanding

The presence, abundance and trends of the shorebirds, waterbirds and seabirds of Western Port are well-known (Loyn 1978, Corrick 1981, Lane 1987, Dann 1993, 1994, Dann *et al.* 1994, Loyn *et al.* 1994, Dann *et al.* 2001f, Loyn *et al.* 2001, Dann *et al.* 2003, Chambers and Loyn 2006, Dennett and Loyn 2009, Hansen *et al.* in prep.). Hansen *et al.* (in prep.) have provided the most recent overview of the significance of Western Port and its roosting and feeding sites for shorebirds and many waterbirds. The seabirds are the lesser known group among birds in Western Port, but some of the species in this group (e.g. cormorants, terns and gulls) have been counted since 1974 (Loyn *et al.* 1994), and a three-year at-sea survey revealed few other seabird species of significance within the bay (Dann *et al.* 2003).

The distribution of birds in Western Port, particularly their roosting sites is generally well known. Although the specific feeding areas of shorebirds have not been systematically monitored for almost 30 years (Andrew *et al.* 1984), some surveys of shorebird and waterbird feeding areas have been conducted in selected places in the past two years (Hansen *et al.* in prep.).

There have been a number of studies of the foraging behaviours and trophic ecology of shorebirds (Dann 1979, 1987, 1991, 1993, 1999a, b, 2000), waterbirds (Lowe 1982a, b, 1984, Howard and Lowe 1984, Dann 2000) and seabirds (Montague *et al.* 1986, Cullen *et al.* 1991, Chiaradia *et al.* 2002, 2003) in Western Port or on Phillip Island to the south. When combined these provide a good indication of the trophic relationships of the main avian groups feeding in the bay.

Many of the bird species for which Western Port is important are declining, and relatively few are apparently stable or increasing (Hansen *et al.* in prep.). These birds come from a variety of foraging guilds. Several reasons have been proposed for the declines, including mudflat reclamation on migratory feeding grounds in Asia for some shorebirds, changing rainfall patterns in Australia (including a local 10-year drought) for a variety of species, and local factors such as seagrass die-off and an increase in fisheries take for piscivorous seabirds, mainly resident species such as cormorants and terns. However there is insufficient evidence to allow the confident identification of the causes of the declines.

In summary, the biggest gaps in our understanding of the birds of Western Port, in the current context, are twofold: (1) we do not know the causes for most of the declines in numbers and whether these numbers will recover, and (2) we do not understand enough about the birds' responses to potential threats in Western Port to evaluate their impacts empirically (e.g. flow-on trophic implications of increased sedimentation, loss of mudflats, increased extraction of fish and invertebrates, behavioural disturbance from recreational activity).

Major threats

Several factors are probably driving the negative trends reported above in shorebird and waterbird numbers, and probably operate both in and outside Western Port. Locally, declining trends have been associated with seagrass die-off and climatic variables (Dann *et al.* 1994, Loyn *et al.* 1994, Chambers and Loyn 2006). Pied Oystercatchers were once hunted in Western Port, and it may be that the cessation of hunting or better protection of their breeding areas around the bay has resulted in their increase. Improved protection of breeding sites from predation and disturbance is likely also to have improved recruitment for other locally breeding species such as the Hooded Plover (Baird and Dann 2003).

Factors operating beyond Western Port include the 10-year drought in south-eastern Australia for a number of waterbird species whose numbers have declined (Hansen *et al.* in prep.), and very likely mudflat reclamation on migration routes in Asia for some others, particularly migratory shorebirds, which are declining throughout their range in Australia (Nebel *et al.* 2008).

Potential threats to various groups of birds within Western Port have been identified at various times over the last 35 years (see Loyn 1978, Lowe 1982a, Dann *et al.* 1994, Loyn *et al.* 1994, Taylor and Bester1999, Dann *et al.* 2001, Loyn *et al.* 2001, Dann *et al.* 2003, Chambers and Loyn 2006, Dennett and Loyn 2009, Hansen *et al.* in prep.). There are a number of potential threats common to these studies, as well as the community consultation process and the asset-threat matrix. These are:

- disturbance from human recreational activity
- extraction of food species
- sedimentation
- sea-level rise
- habitat loss (including loss of seagrass) and fragmentation.

Disturbance from human recreational activity

Risks

Recreational activity in Western Port can pose risks to aquatic birds, and disturbance will increase with increasing recreational use of the bay (Taylor and Bester 1999). Most of the negative consequences of recreation centre around interference with the energy budgets of the birds through either the disruption of feeding or the increased energy use resulting from flying around when disturbed at roosts. Direct disruption of feeding can be caused by recreational activities in the vicinity of feeding areas, particularly approaches by motorized vehicles such as aircraft (particularly helicopters), boats and jet skis. Dogs off leads and people walking and swimming can cause the destruction of eggs and chicks of beach-nesting birds through trampling or predation. Species in Western Port found to be extremely sensitive to disturbance during breeding include the Caspian Tern, Australian Pelican and Fairy Tern (Claridge & WBM Oceanics 1997). Numerous examples of birds being disturbed by recreational activity have been observed during the BOCA survey (Heislers *et al* 2003), but there has been little analysis of how disturbance may have an impact on the distribution and abundance of birds in Western Port.

The suite of species that forage in intertidal areas in Western Port can be classed as either obligate (obtaining more than 90% of their daily energy requirements in intertidal areas) or facultative (less than 90%) intertidal foragers (Dann 2000, Table 1). The obligate intertidal foragers all feed in intertidal areas at night, whereas most of the facultative species do not (Dann 2000, Table 1). This association is presumably because of the difficulty of obtaining sufficient energy during daylight when foraging is limited to low-tide periods.

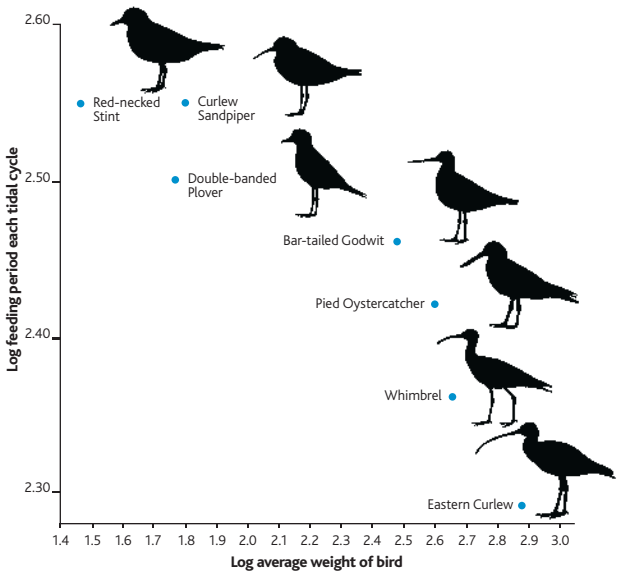
The consequences of disturbance to birds in Western Port are particularly apparent for migratory shorebirds as they are mostly specialised intertidal mudflat feeders, have apparently relatively high energetic demands and congregate at the small number of sites to roost at high-tide. The implications of disturbance to feeding shorebirds are likely to be dependent on the extent to which birds can compensate for loss of energy by adapting behaviours and extending feeding opportunities (Dann 1999, 2000, Loyn *et al.* 2001) and this also applies to the waterbirds and seabirds.

Shorebirds have some opportunities to compensate for loss of feeding time, increased foraging costs or reduced productivity in feeding areas caused by human activities. Some species do not feed for all of the time available around low tide and can make up some shortfalls in their energy budgets by extending their feeding periods. This is more so for the larger shorebirds because they feed for shorter periods than do the smaller species (see Figure 12.4).

Smaller shorebirds have less scope for extending their feeding periods around low tide because they are already feeding for much of this time. Consequently they can be found feeding at some times of the year, and under some conditions, at high tide (Dann 1999a) in what are probably suboptimal feeding areas. Feeding at night is widespread among shorebirds and may allow some compensation for energy shortfalls incurred because of some form of disturbance during daytime low tides.

The effect of disturbance varies with shorebird species, location and time. Larger species are more responsive to human activity than smaller shorebirds (Taylor and Bester 2000) but less affected overall because they require less time to feed (Figure 12.8). Small waders need to feed for long periods because they must maintain a higher metabolism than large birds to compensate for greater heat loss to the environment. Large waders are better able to exploit distant mudflats available for short periods at low tide. Specialist intertidal foragers are also more prone to be significantly affected by disturbance as they have fewer opportunities to balance energy budgets in alternative habitats.

Figure 12.8 The feeding periods each tidal cycle of different shorebird species in relation to their weights. (Source: Dann 1987.)



Areas available for feeding around high tide are quite restricted, especially in mangrove-fringed coasts, and are important for small waders that need to feed for long periods (Evans 1974, Loyn 1978, Dann 1999a). Therefore disturbance is likely to be more important when birds are feeding close to high tide, as it can reduce the time available for feeding considerably.

There are a range of factors that determine why birds roost where they do, including predation risk, proximity to productive feeding areas, and levels of disturbance. While the immediate effects of human disturbance can be dramatic, e.g. the abandonment of a roost during and for some time after the disturbance, it does not always translate into a reduction in the abundance of birds there in the future. It seems likely that levels of disturbance might have to reach some threshold level before abandonment or reduction in bird numbers at a roost occurs. That factors other than human disturbance at the roost site are involved in roost selection is demonstrated by the observation that some roosts where numbers of shorebirds have declined are among the least disturbed by humans in the bay, e.g. those on French Island (Hansen *et al.* in prep.).

Consequences

Disturbance of birds while feeding may reduce foraging success by interrupting feeding on the water or on the mudflats. This is more likely to effect smaller shorebird species (Dann 1999a,b). Furthermore:

- disturbance at high-tide roosts may reach levels that influence roost selection and increase energy demands
- disturbance at breeding sites can increase energy demands and reduce breeding productivity
- all disturbance effects are more significant for the obligate intertidal feeding species due to less capacity to compensate through changing habitats.

Because recreation in Western Port is increasing, the disturbance to roosting, feeding and beach-nesting birds is likely to increase. There are a variety of factors governing the use of high-tide roosts, feeding areas and nesting sites which we do not fully understand.

Fishing and associated activities

Risks

Fishing poses several potential and known risks to aquatic birds in Western Port beyond the disturbance of birds while feeding and roosting covered in the previous section. There is some overlap between fish species taken by humans and birds, some bird species appear prone to being entangled in discarded fishing line and the reduction in biomass and habitat disruption caused by bait extraction may be an issue for feeding shorebirds in some areas.

Many of the piscivorous birds in Western Port are declining for reasons that are unknown and, at the same time, the number of recreational fishers is increasing. Whether fishing by humans has an impact on the distribution and abundance of fishing birds is invariably difficult to determine and rarely established anywhere in the world. Evidence for any effect is generally circumstantial.

Double-banded Plover.
Photo courtesy Annette Hatten.

Entanglement too is a problem for some birds in Western Port and several species, notably Pacific and Silver Gulls, Crested Terns, Little Pied Cormorants and Pelicans, are not infrequently found in the Western Port area entangled in fishing line or with fishhooks or jigs attached and either dead or incapacitated (Phillip Island Nature Parks Wildlife Hospital, upubl. data). The shorelines around Phillip Island have a surprisingly high incidence of discarded fishing line and tackle (Dann pers. obs.). However, it is not known whether the effects of discarded fishing gear are great enough to have a significant effect on the survival of any of these species.

Bait extraction may affect the food resources available for some birds. Ghost shrimps *Trypaea australiensis* are sought by anglers (Contessa and Bird 2004) but are important prey for Eastern Curlews (Dann 2000). Potential competition may also arise between those species whose distribution or abundance has increased due to human influences (e.g. Sacred Ibis, Silver Gull) and shorebirds. Ibises have only been in Western Port in large numbers since the 1960s and probably colonised southern Victoria following the creation of huge areas of pasture where they also feed. Ibises are very numerous in intertidal areas, where they consume large amounts of some organisms that are also taken by shorebirds. Silver Gulls have also probably increased substantially in number in Western Port since European settlement and prey on species also eaten by shorebirds, as well as stealing shorebird food (Dann 1979).

Consequences

Fishing and bait collecting is likely to increase with increasing human activity in the bay, and food resources may be less available to birds as a consequence. The risk of birds being entangled in fishing line is also likely to increase as fishing activity increases. Anthropogenic changes in the abundance of some avian species (i.e. ibis) may have altered the energy available to other birds.

Sedimentation

Risks

Suspended and deposited sediments may have a number of ramifications for aquatic birds. Suspended sediments can reduce primary productivity, and consequently secondary productivity, and may also reduce secondary productivity directly by reducing the efficiency of filter-feeding mudflat biota. A number of avian species declined in Western Port at the same time that seagrass declined (Dann *et al*. 1994, Loyn *et al*. 1994), and increased rates of sedimentation, in part, were implicated in the seagrass decline (Bulthuis 1981, 1984; EPA 1995).

On the positive side, a dredge spoil island at Long Island Point has been used as a roosting site in Western Port for over 30 years, and the use of dredge spoil islands as roosts has been reported at a number of other sites (e.g. Chaney *et al*. 1978). An additional benefit of sedimentation may be the raising of some intertidal areas, which could increase the usefulness of some mudflats as feeding areas available at higher tidal levels if benthic production is not reduced.

Consequences

Increased turbidity from dredging, shipping and recreational boating may reduce primary and secondary production, with flow-on effects for all birds that feed in the bay. Increased sediment in the water column may also reduce the foraging efficiencies of sight-feeding seabirds.
The redistribution of sediments could also result in some elevation reduction, so that a greater area of the substratum may remain submerged at low tide, depleting the size of foraging areas (Lawler 1994).

The proposed Port of Hastings development is likely to involve some dredging. While unregulated dredging can pose risks, it is presumed that these risks would be managed as part of an overall EES/EMP process for this project.
The contribution of increasing use of recreational craft to increased turbidity also needs to be assessed.

Sea-level rise

Risks

Sea-level rise in Western Port is likely to cause reductions in the area of mudflats currently available to birds for feeding (Chapters 3, 7). In addition, some roosts will undoubtedly become less useful for birds as sea levels rise and storm activity alters the geomorphology of the sites, many of which are sand spits (Figure 12.9). For those seabirds nesting close to the water's edge there may also be some loss of breeding habitat if there is insufficient habitat for them above the rising sea levels. As noted above, reductions in feeding opportunities are likely to have a greater impact on shorebirds than on waterbirds, particularly the smaller species of shorebird. Seabird feeding in the bay is unlikely to be affected in the short term by sea-level rise, but some reductions in breeding area are expected for Crested Terns and Fairy Terns at The Nobbies and Rams Island respectively.

Consequences

The consequences of sea-level rise for feeding and roosting birds will depend to a large degree on the extent to which the losses are compensated by the creation of new feeding and roosting sites. If there is a net loss of feeding areas and roosting sites, some reductions in the abundance and biodiversity of birds in the bay would ultimately be expected, but it is not known at what thresholds this might occur. The importance of feeding areas at higher elevations has been mentioned previously, and changes there would be likely to be the most significant.

Habitat and productivity loss and fragmentation

Risks

There are a suite of biotic and abiotic factors that determine the use of feeding and roosting areas by birds in Western Port (Dann 2000). Loss of feeding areas or reductions in primary and secondary production through sea-level rise or sedimentation have already been discussed. However perturbations in the productivity of seagrass can also have an impact. Over the past 35 years a number of avian species was reported to have decreased in abundance around the time the seagrass die-off occurred (Dann *et al*. 1994, Loyn *et al*. 1994). The abundance of Black Swans, which are significant consumers of seagrass, declined sharply and not unexpectedly in response to the loss of seagrass habitat in the bay. Some species in other guilds, such as those that fed on invertebrates in soft sediments or on fish, declined at about the same time, and it was proposed that this was a consequence of a decline in secondary production following the seagrass die-off (Dann *et al*. 1994, Dann 2000).

Reductions in mudflat area through reclamation or infrastructural development also pose risks for feeding shorebirds, and to a lesser extent for waterbirds.

Consequences

Loss of roost sites caused by coastal development may reduce the number of species using adjoining feeding areas if suitable alternatives for roosting are not available. Loss of feeding areas per se may reduce the carrying capacity of Western Port for some species, through either reduced productivity of benthic prey or reduced availability or access for feeding birds. In particular, a loss of feeding areas at higher elevations could cause the smaller shorebirds to move elsewhere if they have no alternative mechanisms to balance their energy budgets.

Figure 12.9 Aerial photograph of Rhyll Inlet, north-eastern Phillip Island showing the main shorebird roost (sand spit indicated by red ellipse) and an adjacent roost of secondary importance (rocky island indicated by yellow square). (Photo: Phillip Island Nature Parks.)

Research that can fill key knowledge gaps

For birds, the most important risks are associated with habitat alteration including direct loss (e.g. through sea level rise), disruption or reduction in habitat quality (e.g. through changes to food levels). Mitigating these risks will require an understanding of which areas are important for feeding and roosting.

While the important roosting sites for shorebirds and waterbirds around the bay are generally well documented, it is not known what determines why the birds roost where they do. Both increasing levels of human disturbance and sea-level rise threaten the utility of many roosting sites. Most roosts are close to the water's edge and will undoubtedly be inundated as sea levels rise. This lack of understanding of the factors determining roost selection mean that it is not possible to predict if the birds will readily adapt to loss of current roost sites. It is also unclear how sea-level rise will effect the provision of roosting sites in the future. A lack of knowledge is impeding the potential mitigation of threats associated with roosting. For example, it may be possible to provide new roosts using dredge spoil or through other mechanisms that reduce the threats. In addition, the usefulness of existing roosts may be extended by raising them farther above sea level. However, it is not known how the birds would respond to the creation of new roosting sites or the modification of existing ones. Effects on other organisms, as well as on hydrology and on bathymetry would also need to be considered.

Where birds feed in Western Port is known in a broad sense for shorebirds, waterbirds and seabirds. However, more work is required on the locations of the important feeding areas for shorebirds and, to a lesser extent, waterbirds. It is over 30 years since any extensive mapping of shorebird feeding areas has occurred (Andrew *et al.* 1984) with the exception of some recent mapping at selected sites by Hansen *et al.* in draft). Given that the abundance of many shorebird species is declining and that most of these are obligate intertidal species (Table 12.1), it is of some importance that their current feeding grounds are carefully mapped and evaluated. This is particularly so because, without this knowledge, we are unable to make any predictions about the effects of loss of mudflat area or productivity on shorebirds.

Table 12.1 The species composition of the avian foraging guilds in Western Port in relation to their seasonal use of intertidal areas, dependence on intertidal areas for daily energy requirements and use of intertidal areas for feeding at night. (Sources: Loyn 1978; Lowe 1982a, b, 1984; Dann *et al.*1994; Loyn *et al.*1994; Dann 2000).

GUILD	SPECIES	Seasonal use of intertidal areas	Daily energy requirements from intertidal areas*	Nocturnal feeding in intertidal
Benthic and nekton-feeding wading birds	Royal Spoonbill	all year	obligate	yes
	White-faced Heron	summer–spring	facultative	no
	Australian White Ibis	all year	facultative	no
Piscivores	Pied Cormorant	all year	facultative	no
	Little Pied Cormorant	all year	facultative	no
	Crested Tern	all year	facultative	no
	Australian Pelican	all year	facultative	no
Polyvores	Silver Gull	all year	facultative	yes
	Pacific Gull	all year	obligate	yes
	Chestnut Teal	autumn–winter	obligate	yes
Herbivores	Black Swan	spring–autumn	obligate	yes
Benthic-feeding waders	Red-necked Stint	spring–summer	obligate	yes
	Curlew Sandpiper	spring–summer	obligate	yes
	Double-banded Plover	autumn–winter	obligate	yes
	Pied Oystercatcher	all year	obligate	yes
	Bar-tailed Godwit	spring–summer	obligate	yes
	Whimbrel	spring–summer	obligate	yes
	Eastern Curlew	spring–summer	obligate	yes

* obligate — obtains more than 90% of the daily energy requirements in intertidal areas) or facultative (less than 90%)

Unlike shorebirds, many of the waterbirds in Western Port feed in habitats other than intertidal areas, and the feeding areas of many species are widespread in the bay. The three waterbird species of particular interest are those that are obligate intertidal feeders (Royal Spoonbill, Chestnut Teal and Black Swan). The preferred feeding areas of these three species may be more restricted than those of other waterbird species and may therefore require more intensive management.

Perhaps the most pressing knowledge gap for birds in Western Port is the cause(s) of the decline in the fish-eating guild. Several other guilds are also in decline, but each of those are likely to be associated with conspicuous environmental events or processes, e.g. shorebird decline because of mudflat reclamation in the Yellow Sea, and decline in various waterbird species during the 10 years because of drought. Most of the piscivores in decline are resident in Western Port or spend much of their lives there suggesting that the causes of decline are local in provenance. Certainly production of some fish species declined after the seagrass die-off (Edgar and Shaw 1995a) and this was evident for some species at the time but others continue to decline decades later. It is not known if this decline is localised to Western Port or is occurring in other embayments on the Victorian coast. It would very instructive to have some fishery-independent monitoring of fish populations to better understand future patterns of abundance of fish-eating birds.

Bar-tailed Godwit.
Photo courtesy Annette Hatten.

Research priorities

Examine the trends of fish-eating birds in Western Port.

A high priority for avian research in Western Port is to look at the trends of fish-eating birds over the past 35 years and to identify the underlying causes for the declines. There are datasets available for fish-eating birds in other bays along the Victorian coast for comparison, which would identify how localised the phenomenon may be and direct either mitigation strategies or further research requirements.

Determine the relative significance of shorebird and waterbird intertidal feeding areas.

Systematic mapping of low-tide feeding areas of shorebirds and waterbirds is required throughout the bay, as well as an evaluation of their significance as feeding areas for aquatic birds. This would allow an evaluation of impacts resulting from changes in intertidal area or processes, and help set priorities for reservation or other protection mechanisms.

Determine the factors involved in roost selection in shorebirds including the role of human disturbance.

Determining the factors involved in roosting site selection by shorebirds and waterbirds is an important step in evaluating the options and significance of potentially threatening processes operating at roosts such as human disturbance and sea-level rise. Mitigation options resulting from this research might include the modification of existing roosts or the creation of new roosts with correspondingly less associated risk for birds.

Investigate the effects of sea-level rise on shorebirds and waterbirds.

Inevitably sea-level rise will have implications for the availability of roosting sites and intertidal feeding areas for shorebirds and waterbirds. In the first instance, modelling of shorelines under different scenarios of sea-level rise would provide some indication of where roosting sites will be lost and where new ones may be created. Modelling the profiles of intertidal mudflats for different sea level scenarios would be equally useful in predicting the extent of foraging areas and their utility in the future.

Marine mammals

Distribution

Although a variety of marine mammal species have been reported in Western Port (Menkhorst 1995, Chidgey and Crockett 2010) and the largest Australian Fur Seal *Arctocephalus pusillus doriferus* colony in the world is just outside the western entrance (Kirkwood et al 2010), it appears to have relatively little significance as marine mammal habitat. Australian Fur Seals and Bottlenose Dolphins *Tursiops truncatus* are the most frequently reported marine mammals in Western Port, but neither are abundant there (Dann *et al.* 1996). Although a number of other species of marine mammal have been reported, those that occur more frequently appear to be only passing through the southern part of the bay on their way elsewhere.

Australian Fur Seals

There are an estimated 30 000 Australian Fur Seals in the Seal Rocks colony at the western entrance to Western Port, including bulls, seals and pups (Kirkwood et al 2010). Of about 60 individuals that have been tracked by satellite over the past 10 years (Kirkwood *et al.* 2002, 2006, Arnould and Kirkwood 2008, Kirkwood and Lynch, unpublished data), none have ventured far into the bay, even though the colony is thriving and has doubled in size between the 1980s and 2007 (Kirkwood *et al.* 2010).

Seals do occur in small numbers in Western Port, and Dann *et al.* (1996) found an average of just over two per monthly trip along 81 km transects in Western Port between 1991 and 1994. Most seals were recorded in the western and northern arms of the bay, particularly at the western entrance near the breeding colony. Generally, single and mostly small individuals were seen, presumably juveniles or small females (Dann *et al.* 1996).

Figure 12.10 Australian Fur Seals on Seal Rocks.
(Photo: Roger Kirkwood, Phillip Island Nature Parks.)

Bottlenose Dolphins

Forty-six Bottlenose Dolphins were reported along 11 (32%) of 27 monthly 81 km transects between 1991 and 1994 (Dann *et al*. 1996). Usually the dolphins were seen in small pods at the two entrances, and the maximum recorded in one survey was ten (Dann *et al*. 1996). In Port Phillip Bay there is a resident pod of approximately 80–100 genetically unique 'bottlenose dolphins' as well as a small population of Short-beaked Common Dolphins *Delphinus delphis* that inhabit a small area around Mornington (Sue Mason, Dolphin Research Institute, pers. comm.). Although Western Port is utilised by many fewer dolphins than Port Phillip Bay, there are some notable resident dolphins that can be reliably found in Western Port. For example, three adult dolphins and a subadult resident can regularly be found off Somers (Sue Mason pers. comm.). It appears that dolphins are not particularly common in Western Port, however.

Other species reported

New Zealand Fur Seal *Arctocephalus forsteri* — one record of an immature male in northern Western Port (Menkhorst 1995).

Subantarctic Fur Seal *Arctocephalus tropicalis* — one record on north-western coast of Phillip Island (Menkhorst 1995).

Australian Sea Lion *Neophoca cinerea* — one record of an adult male on Ventnor northern side of Phillip Island (Kirkwood *et al*. 1999).

Leopard Seal *Hydrurga leptonyx* — various records in the two entrances and the coast of Phillip Island (Menkhorst 1995, Renwick and Kirkwood 2004).

Southern Elephant Seal *Mirounga leonina* – one record in the western arm of Western Port (Menkhorst 1995).

Common Dolphin *Delphinus delphis* — reported at the eastern entrance to Western Port (Menkhorst 1995) and occasionally inside the southern parts of the bay (Dr Roger Kirkwood, Phillip Island Nature Parks, pers. comm.).

Dusky Dolphin *Lagenorhynchus obscurus* — (Chidgey and Crockett 2010).

Killer Whale *Orcinus orca* — seen rarely in the western entrance (Menkhorst 1995) and inside the bay (Roger Kirkwood pers. comm.).

Humpback Whale (Group V) *Megaptera novaeangliae* — individuals observed inside Western Port (temporary visitors) in most winters (Menkhorst 1995, Roger Kirkwood pers. comm.)

Minke Whale *Balaenoptera acutorostrata* — one record at Stockyard Point (Menkhorst 1995).

Pygmy Right Whale *Caperea marginata* — one record at Silverleaves on northern coast of Phillip Island (Menkhorst 1995).

Brydes Whale *Balaenopointera edeni* — (Chidgey and Crockett 2010).

Blue Whale *Balaenopointera musculus* — (Chidgey and Crockett 2010).

Southern Right Whale *Eubalaena australis* — found in most years around the western entrance or southern coast of Phillip Island, also occasionally observed in the southern part of the bay. Calving, which occurs elsewhere on the Victorian coast, has not been recorded in Western Port (Menkhorst 1995, Roger Kirkwood pers. comm.)

Little Penguins.
(Photo: Phillip Island Nature Parks www.penguins.org.au.)

Major threats

Sea level rise

Sea-level rise will have most impact in the short-term on the available area of low-lying islands important to breeding Australian Fur Seals. The area available for the seals at Seal Rocks will halve with a 0.8 m sea-level rise (Roger Kirkwood pers. comm.), which is predicted to be likely by the end of this century (Ramhstorf 2010). Concomitant with sea-level rise, increased mean wave heights and storm frequencies would also reduce the habitable space for the seals. Therefore, the number of seals breeding at Seal Rocks is certain to decrease. Pup mortality rates caused by density pressure (crushing, etc.) and drowning are likely to increase, and seals are likely to opt to breed at other sites less altered by sea-level rise.

Western Port is a nursery for several fish species (Chapter 11) that are significant within the trophic structure in which seals, and probably dolphins, are high-level predators. In particular, an increase in sea temperature is likely to change the distribution and availability of prey for seals (Kirkwood *et al.* 2008) and dolphins.

Oil spills

The location of Seal Rocks, at the entrance to Western Port and adjacent to the shipping channel, makes it vulnerable to oil spills. Seals attempt to groom (chew) oil from their fur and quickly ingest toxic quantities (Roger Kirkwood pers. comm.). An oil spill in the vicinity of Seal rocks could seriously deplete the colony. Although new cleaning options for fur and rock are imminent, the capture of even modest numbers of adult seals for cleaning is unlikely and many affected individuals would be likely to perish. In contrast, in the event of oiling, many small pups could be caught and cleaned.

Research that can fill key knowledge gaps

Completion of the development of magnetic particle technology for removing oil from both fur seals and rock (Orbell *et al.* 2004, Van Dao *et al.* 2006, Orbell *et al.* 2007) may reduce the impact of an oil spill directly on younger seals if oil affected animals could be captured in significant numbers. In addition, the quick clean-up of oiled rock at Seal Rocks would minimise the number of animals affected.

It would be instructive to model the reduction of breeding habitat for fur seals in Bass Strait in relation to sea-level rise predictions, together with an evaluation of alternative (higher) sites. There appear to be a number of potential breeding sites in Bass Strait that could be used as an alternatives to existing low-lying sites as sea-level rises.

More detailed information on the occurrence of Bottlenose Dolphins in Western Port (patterns of residency, transience) is required to assess the size of the resident population. Confirmation of their taxonomic status is also required. It addition, an assessment of the origin of transients would assist in the determination of the role Western Port plays in the annual or life cycles of dolphin populations elsewhere.

13 Rocky reefs

Rachael Bathgate, Michael Keough and Gerry Quinn

Figure 13.2 Low-relief intertidal reef at Shoreham.
(Photo: M. Keough.)

Rocky reefs occupy only a very small part of Western Port, but three areas are notable — Crawfish Rock, an unusual habitat with very high biodiversity; a small reef near San Remo that is significant for its opisthobranchs; and intertidal reefs along the south-western coast, particularly Honeysuckle Reef, that have a high biodiversity. Intertidal reefs in Western Port are likely to be very vulnerable to sea-level rise. There is some evidence that there has been a loss of diversity at Crawfish Rock, most likely a result of high turbidity in the North Arm.

We identify several research gaps, including a better understanding of the biodiversity of deep channels and the impacts of recreational activities on intertidal reefs, but the two most important gaps are an assessment of risks from sea-level rise and a lack of knowledge about the sediment-based water quality threshold for algae on reefs in the North Arm. We suggest that these thresholds do not need to be resolved immediately, because seagrasses will be more susceptible to light reduction, and improvements to seagrass habitat will flow through to improvements to reef algae.

Rocky reefs around Western Port

Rocky intertidal and subtidal reefs in Western Port are an important environmental asset in the bay, but they cover considerably less area than other habitat types such as soft sediments and seagrass. A number of reefs in Western Port have a high species diversity or are unique because they support combinations of species not found elsewhere (O'Hara and Barmby 2000; Chidgey *et al.* 2009; Edmunds *et al.* 2010). Whereas impacts to seagrasses have been highly visible and seagrasses have been the subject of considerable research, reefs in Western Port have received far less attention (but see Shepherd *et al.* 2009). While there is an extensive body of scientific literature on human impacts on reef systems throughout the world, there are no targeted studies that have looked at threats to biological reef assemblages in Western Port.

The reefs discussed here are within the boundaries of Western Port as defined for this review (see Chapter 1), thereby excluding significant intertidal and subtidal reef such as Mushroom Reef and the southern coast of Phillip Island.

The most significant intertidal and subtidal reefs in Western Port are near the two bay entrances — in particular, at Griffith Point and San Remo at the eastern entrance, and on the western shoreline from Flinders to Shoreham. There are also intertidal reefs on the southern coast of French Island and the northern coast of Phillip Island, and in the northern section of the bay there are patches of low-profile reef on the eastern shore. Discrete rocks (e.g. Crawfish Rock), small rocky islands and shoals punctuate the deep channels and soft-sediment benthos throughout Western Port.

Reefs are characterised by diverse algal and animal assemblages. Subtidal reefs support a range of human activities, such as fishing, snorkelling and diving, and intertidal reefs provide opportunities for a range of passive recreational activities. In Western Port, recreational snorkelling and diving are more confined to southern sections, particularly along the western side. Reefs are linked to other ecosystem components through nutrients — their biota assimilate dissolved nutrients, and organic material may be exported when algal tissues break off or decay and are carried to other habitats. Higher in the food webs, very mobile species of predatory fish such as snapper and wrasse, and birds such as oystercatchers, can move easily between environments, creating a rapid flow of energy between habitats.

Three significant reef areas have been identified: Crawfish Rock, a small reef near San Remo, and a complex of small reefs and seagrass meadows along the western coastline of Western Port. Of these, Crawfish Rock and perhaps San Remo are the most significant on a broader scale.

Distribution

The distribution of reefs and other habitats in Western Port were summarised by Chidgey *et al.* (2009) and Edmunds *et al.* (2010), and coastal formations were described in detail by Bird (1993). The locations of intertidal and subtidal reefs are shown in Figure 13.1, but note that rocks are not distinguished from soft sediments in the Upper North Arm, and that it was difficult to detect deep-water reefs because of the high water turbidity (D. Ball, Department of Primary Industries, pers. comm.).

Western shoreline

The coastline along the Western Passage from Flinders to Sandy Point is characterised by wave-cut platforms, basalt cliffs and sandy beaches, which are subject to strong waves under certain weather conditions (WRPCC 1992, Edmunds *et al.* 2010; see also Figure 13.2). Large areas of the reefs at Merricks, Balnarring and Somers are a continuation of the basalt substratum at Flinders and Shoreham (Honeysuckle Point and Point Leo). Honeysuckle Point near Shoreham, and reefs at Flinders (outside the entrance to Western Port) have been suggested to have a particularly high biological value (Handreck and O'Hara 1994, Edmunds *et al.* 2010). These western reefs do not extend far into deeper water or north of Sandy Point. Sandstone Island, north of Cribb Point, is an outcrop of sandstone and mudstone that forms coastal bluffs and shore platforms (DPI 2011d).

North Arm, Corinella Segment and French Island

There are no intertidal reefs between the lower and upper areas of the North Arm, but there are deep-water rocky outcrops north-west of French Island, most notably Crawfish Rock, Eagle Rock and Barrallier Island. On the mainland, patchy cobbled reef reappears at Settlement Point near Corinella. In the channels between Settlement Point and French Island there are several small rocky outcrops: Pelican Island, Snapper Rock, and Elizabeth Island to the south-west. Pelican Island is an accumulation of basalt rubble on a rocky substratum, with gravel spits towards the east (Edmunds *et al.* 2010).

13 Rocky reefs

Figure 13.1 Rocky reefs of Western Port.

Elizabeth Island has a narrow, rock-strewn shore platform cut in basalt and Mesozoic sediments, with small pockets of mangroves and saltmarsh on the eastern shore (Edmunds *et al.* 2010). The small outcrops known as Rams Island and Bird Rock lie to the west of Elizabeth Island. Reef Island and Kennedy Point, north of the Bass River delta, are characterised by cobble or shingle reef. Ridges of basalt gravel extend seaward from Kennedy Point to Reef Island and link the two at low tide (DPI 2011e). There are scattered rocky shoals (presumably subtidal) offshore from the delta, including Maggie Shoal and Loelia Shoal (Figure 13.3).

Figure 13.3 Patchy rocky reefs around French Island. Symbols are as for Figure 13.1. (Source: DPI.)

On French Island there are patches of narrow cobble or shingle intertidal reefs, from Red Bluff on the south-eastern coast and becoming more defined around Tortoise Head to the west, terminating at Tankerton. The base of Tortoise Head is basalt exposed in shore platforms around the southern and western margin. The shoreline of Tortoise Head has a high-wave-energy shore platform along its southern margin. Along the western margin cobbles and sandy beaches overlie wide shore platforms (DPI 2011f).

Rhyll Segment, Eastern Entrance and Phillip Island

The intertidal and subtidal reefs at San Remo support a high diversity of one invertebrate group — opisthobranchs (sea-slugs and sea-hares) — and are listed as a threatened community under the Flora and Fauna Guarantee Act. Bluffs and low cliffs of mudstone and Older Volcanics extend along the San Remo Peninsula beside the eastern entrance to Western Port (Bird, 1993) and the coast is fronted by a wide, gently sloping shore platform covered in pebbles (DPI, 2011). This reef continues oceanwards around Davis Point, where it forms high-profile intertidal reefs and cliffs near Griffith Point. There are also subtidal reefs in the channel between San Remo and Newhaven, cobble or shingle reefs on Newhaven beach towards Cape Woolamai, and narrow stretches of intertidal and subtidal granite reefs at Cape Woolamai.

Along the northern shore of Phillip Island, intertidal reef platforms are interspersed with sandy beaches. Intertidal reefs extend from Point Grant (the eastern point of Phillip Island) almost to Cowes. There are significant subtidal reefs and reef–sediment offshore, mirroring the distribution of intertidal reefs. Notable intertidal reefs on the northern coast are McHaffie Reef at Ventnor and the reefs around Cat Bay. There are no intertidal or subtidal reefs between Cowes and Churchill Island. The weathered basalt of Churchill Island forms outcrops around the shoreline. Most of the island is bounded by coastal bluffs, but there are beach ridges of coarse gravel and cobbles in the south (Edmunds *et al.* 2010).

Special features

Marine Protected Areas

There are not many reefs within marine protected areas in Western Port. Churchill Island Marine National Park includes a small intertidal area of cobbles overlying soft sediments on the western shore of Churchill Island. There is also a small amount of hard substratum associated with Barrallier Island, a small ferruginous sandstone island in the north-west corner of French Island Marine National Park (Edmunds *et al*. 2010). Barrallier Island is a major waterbird roosting site (DSE 2003), but there is no information about any reef biota there.

Special Management Areas

A number of reefs in Western Port have a high conservation value. Honeysuckle Reef (25 ha), Crawfish Rock (45 ha) and San Remo (70 ha) Special Management Areas (SMAs) were highlighted in the earlier Marine, Coastal & Estuarine Investigation (ECC 2000), although they have no particular conservation status. A section of reef on the south-western tip of Phillip Island lies within the Summerland Peninsula SMA and Phillip Island Nature Park.

San Remo Reef is a listed community under the Flora and Fauna Guarantee Act because of its diverse and apparently unique opisthobranch and bryozoan species assemblages (DSE 2003a). The community is rare in terms of its total area and distribution (the only location in Victoria) and is therefore vulnerable. The habitat is a mix of sediments, boulders, soft-weathered basalt, algae and seagrass, extending from the intertidal to the edge of a deep channel, and is subject to rapid current flows. The opisthobranchs *Rhodope* sp. and *Platydoris galbana* are also listed as threatened under the Act.

Crawfish Rock SMA is in the main tidal channel of the North Arm and has intertidal and subtidal reef composed of ferruginous sandstone (Shapiro 1975; Edmunds *et al*. 2010). Strong and reversing currents and deep surrounding waters have prevented the accumulation of sand and gravel beaches and spits (DPI 2011h). This site is also significant because it supports a benthic fauna more often associated with deepwater communities in Bass Strait (Shapiro 1975, ECC 2000). Because the high water turbidity around Crawfish Rock reduces light penetration, many deepwater species of algae, hydroids and sponges occur at unusually shallow depths, and the high currents probably provide a good supply of food, allowing a high density of organisms. Almost 600 species have been documented at this site: 130 algae, 150 sponges, 50 hydroids, 180 bryozoans and 80 ascidians (Shapiro 1975). Note, however, that the ECC (2000) reported species numbers for hydroids and ascidians as 123 and 34 respectively. The rare FFG-listed hydroid *Ralpharia coccinea* is found at Crawfish Rock, and may be endemic to Western Port (Edmunds *et al*. 2010). This site was the subject of one of the recommendations of the Shapiro report:

'Provision be made for the declaration of 'Marine Parks' in Westernport Bay with the early assessment of Crawfish Rock, Eagle Rock (both in the North Arm south of Quail Island) and the subtidal area in the vicinity of Seal Rocks for suitability for this purpose.'

There has not been a recent update of the diversity of this site except for an algal species list compiled by Shepherd *et al*. (2009).

Honeysuckle Reef, along with Mushroom Reef, has been identified as having the most species-rich intertidal community in Victoria (Handreck and O'Hara 1994, ECC 2000). The species richness index on which this assessment was based was calculated as the number of intertidal and shallow subtidal invertebrates species found at a site out of a possible 282 species listed in the records of the Marine Research Group of Victoria (Handreck and O'Hara 1994). The authors list the caveats associated with this index, including differences in field work effort between sites, which they scored as high for both Flinders and Honeysuckle Point in comparison to other areas (e.g. medium at Newhaven, low around Cape Schanck). Recent field observations indicate that Honeysuckle Reef continues to be a site of high epifaunal diversity but that the increasing popularity of the site may have negative consequences for the natural values (Tim O'Hara, Museum Victoria, pers. comm.). The reef is predominantly flat and shallow and lies within a fairly sheltered bay area, and most of the reef is exposed at low tide. A shallow pool is used extensively by schools of young fish. Adjacent beach areas are used as a high tide roost for migratory waders (ECC 2000).

The ECC report stated that there had been a decline in intertidal invertebrates at nearby Mushroom Reef in recent years. This is probably on the basis of observations by the Marine Research Group of notable declines in some intertidal and shallow subtidal fauna in Western Port as a whole since the 1980s, thought to be the result of over-collecting (Handreck and O'Hara 1994).

Other sites

Eagle Rock is a slightly smaller outcrop north-west of Crawfish Rock. It was identified, along with Crawfish Rock, as having a significant ecological value in terms of populations of invertebrates (Shapiro 1975). Eagle Rock shares the same geomorphological characteristics as Crawfish Rock but is not exposed at low tide, approaching to within 1 m of the surface. It does not have any special conservation status, and there are no data comparable to that available for Crawfish Rock.

The intertidal rock platform at the end of Settlement Point, 1 km west of Corinella, was identified as a site of interest in the Shapiro (1975) report because of its unusual form and because of the variety of species living on and immediately around it. Settlement Point is a tidal platform of soft weathered basalt bordered on the north-west and south by soft-sediment tidal flats. The site was recognised as unique for its large population of the boring bivalve *Venerupis crenata* in the soft rock of the main portion of the platform, and for the reportedly rich subtidal algal and invertebrate assemblages along the seaward edge of the reef, which is exposed to strong currents in the adjacent deepwater channel. No other significant algal or invertebrate taxa have been recorded at this site.

The intertidal rock platform at Merricks has the only recorded population of the sea cucumber *Apsolidium handrecki* in Victoria (O'Hara and Barmby 2000).

Figure 13.4 Algae and invertebrates attached to pier pilings at Flinders. (Photo: M. Keough.)

Artificial habitats

There is little information on the epifaunal and algal species associated with artificial structures in Western Port. Previous studies have focused mainly on sessile species on marinas, wharfs and pier pilings in the north-west of the bay. A qualitative assessment of the biota on the Stony Point wharf was made by Chidgey and Crockett (2010). Submerged pier pilings are described as being dominated by encrusting invertebrates, particularly large sponges and ascidians at greater depths and some *Caulerpa* spp. and red algae towards the surface. Webb and Keough (2000) examined sessile invertebrates and recruitment on surfaces in and outside Western Port marina at Hastings, and compared them to the St Kilda marina in Port Phillip Bay. They detected differences in the recruitment and abundance of bryozoans and ascidians inside marinas compared to areas outside, and different patterns between the two bays. Webb (2000) gave details of the fouling species (e.g. ascidians, bryozoans, sponges, and serpulid worms) examined in this research. In an assessment of marine pests around Hastings, Currie and Crookes (1997) did diver surveys of wharf piles (steel industry wharf, Long Island pier and Crib Point jetty), pylons and breakwalls (Western Port marina) and collected samples for identification. Subtidal reef at Eagle Rock was also surveyed, specifically for *Sabella* and *Undaria*. They provided a complete list of introduced and native species identified in these surveys. Wright (1996) found that the assemblages on Rhyll jetty were species-poor, with the barnacle *Elminius covertus* the dominant spatial occupier. The gastropods *Nodolittorina unifasciata* and *Bembicium nanum* were the most common mobile fauna at this site.

Species of particular interest

Fish

The fish assemblages of rocky reefs in Western Port have not been well documented (Chidgey *et al.* 2009; see also Chapter 11), although it is known that species such as Snapper, King George Whiting and Australian salmon periodically visit reefs, often during a particular life-history stage (Gunthorpe and Hamer 1998). There has been much more research directed towards fish, fisheries and associated habitats in Port Phillip Bay (e.g. Morris and Ball 2006, and references cited therein). Longmore *et al.* (2002) used stable isotopes to identify habitats that are important for the food chain of commercial fish in Western Port, but this approach detects only vegetative food sources (e.g. seagrass, epiphytic algae).

Invertebrates

There is no definitive list or database of all rocky reef invertebrate species that inhabit Western Port (Dr R. Wilson, Museum Victoria, pers. comm.). However, we know that Western Port has a diverse reef invertebrate fauna (e.g. Figure 13.4), and that there is substantial variation between more oceanic and northern areas. Sites of significance for threatened marine invertebrate species in Victoria are found in reef habitats in Western Port (O'Hara and Barmby 2000). Merricks shore platform is the only recorded site for the holothurian *Apsolidium handrecki* in Victoria. San Remo reef flat is a significant site for the FFG-listed opisthobranchs *Platydoris galbanus* and *Rhodope* sp., in addition to which there are 11 other undescribed opisthobranch molluscs known only from this site. *Ralphia coccinea* (Cnidaria, Crawfish Rock) is also an FFG-listed species (Edmunds *et al.* 2010).

Summary of current understanding

Most quantitative reef surveys have been done outside the entrances to Western Port, but are mentioned here because of their proximity and likely biological links with reef communities in Western Port itself. It should be noted however, that reef communities farther into the bay are quite distinct from those at the entrances (O'Hara *et al.* 2010). The intertidal algae and macroinvertebrates of Mushroom Reef Marine Sanctuary have been monitored since 2003 as part of Parks Victoria's Intertidal Reef Monitoring Program (e.g. Edmunds *et al.* 2004, Gilmour and Edmunds 2007), with habitat mapping conducted as part of the Victorian Marine Habitat Classification System (Ball *et al.* 2006). Subtidal reefs on the southern coast of Phillip Island have been surveyed since 1999 (Edmunds *et al.* 1999, Gilmour *et al.* 2006). Fish, invertebrates and algae recorded at Phillip Island are likely to also occur at nearby sites within Western Port, and intertidal species at Mushroom Reef are likely to occur at nearby reefs that have similar physical attributes (Parks Victoria 2007c). Many species at Flinders are found on reefs to the north, such as Merricks and Honeysuckle Reef. Because wave energy dissipates as it moves into the bay these more northerly reefs are more sheltered and experience more sediment deposition, and therefore support different types of species. A significant amount of qualitative data has been collected by the Marine Research Group since 1957 through faunal surveys on intertidal reefs, and to a lesser extent subtidally. While their records provide presence–absence information, they are not a quantitative data source that would enable statistical tests of shifts in species abundance through time (Handreck and O'Hara 1994). More recently, O'Hara *et al.* (2010) undertook intertidal surveys of macroinvertebrates at 58 sites across Victoria, including sites in Western Port. Their analysis showed that intertidal reefs inside Western Port were quite distinct from ocean reefs and those at Flinders. Settlement Point was noted as having an unusual assemblage, with relatively few gastropods but many crabs, chitons and echinoderms, and small numbers of other groups such as brachiopods. Other short-term, unreplicated studies have been undertaken on reefs in the bay. For example, in 1998 the Environment Protection Authority (EPA) undertook a simple descriptive assessment of the abundance of the habitat-forming alga *Hormosira banksii* on intertidal reefs, including those on Phillip and French Islands (Ferns and Curnow 1998).

At Griffith Point, to the south of San Remo at the Eastern Entrance, a sandstone reef extends subtidally and southwards around the point, resulting in reefs exposed to varying degrees of wave force. This has been the site of several early studies of the biology and population dynamics in intertidal invertebrates, which identified the factors underlying gastropod species distribution, growth and reproduction and now provide historical data of the abundance and distribution of common species. For example, Parry (1977, 1982) investigated the ecology and reproductive biology of four species of limpets — *Cellana tramoserica, Notoacmea petterdi, Patella peroni* and *Patelloida alticostata* — between 1971 and 1975, and found that they are a food source for Sooty Oystercatchers and Purple and Blue-throat Wrasses. Quinn (1988, 1988) studied the population dynamics of the pulmonate limpet *Siphonaria diemenensis* in relation to food availability and investigated reproductive patterns and energetics in this species.

Edmunds *et al.* (2010) provided the most recent overview of reef and other habitats in Western Port as part of the Victorian National Parks Association advocacy of additional marine conservation areas for Victoria. Their study was a comprehensive literature review but did not involve the collection of any new scientific data, nor any formal scientific review. Chidgey *et al.* (2009) reviewed the current scientific understanding of ecological assets in Western Port, and noted that 'Most of the descriptions of the marine ecosystem are decades old. There is little available contemporary information on marine community characteristics, species distribution or state of ecosystem processes in Western Port.'

The only long-term scientific effort has concentrated on Crawfish Rock, largely involving the identification of the large number of algal and invertebrate species (Smith *et al.* 1975, Chidgey 2009). A recent survey of Crawfish Rock showed that there has been significant degradation of the macroalgal community (Shepherd *et al.* 2009). The observed changes in diversity and cover of algae have been attributed to lower light penetration caused by an increase in suspended solids and direct sedimentation (Shepherd *et al.* 2009). Long-term qualitative observations show no comparable decline in the invertebrate fauna at this site (Dr J. Watson, Marine Science and Ecology, pers. comm.).

In the 1970s, the kelp *Ecklonia radiata* dominated the upper canopy down to 8 m, along with *Sargassum, Scaberia* and *Caulerpa* in the more sheltered, shallow waters (Figure 13.5). A diverse red algal flora comprising *Claudia elegans, Griffithsia teges, Myriogramme gunniana* and *Rhodymenia* lived among the *Ecklonia* and below the level of the kelp to a depth of approximately 10 m (Smith *et al.* 1975, Shepherd *et al.* 2009). From the 1960s to 2002–2006 the number of algal species declined from 138 to 47. In addition to these remaining species, 20 new species were recorded, two of which are exotic: *Codium fragile* ssp. *fragile* and *Schottera nicaeensis*. In the later surveys, *Ecklonia* was significantly less abundant and had a lower depth range of 3 m, and there was a substantial decline in the number of red algal species. Crawfish Rock was described in the 1970s as having a large littoral rock platform and an abundant and diverse intertidal area (Shapiro 1975). Thirty years later, the diverse and abundant algal cover in the intertidal zone was found to have been replaced by silty surfaces and sparse algal growth (Shepherd *et al.* 2009).

The invertebrate communities at Crawfish and Eagle Rocks may have been negatively affected, albeit temporarily, by dredging of the shipping basin in the early 1970s (Watson 2009, Chidgey *et al.* 2009). In the 1970s sediment from the dredging program was found smothering the reef and sessile animals such as sponges (Watson *et al.* 2009). The sublittoral community at Eagle Rock showed distinct signs of stress from smothering by finely divided material, resulting in the death of many sponges, with partial recovery seen one year after cessation of dredging (Dr J. Watson, cited in Chidgey *et al.* 2009).

Most of the area laid bare by death of the sponges supported new colonies, but some parts over which sand had deposited did not recover. In our opinion, because of the very strong currents operating in this area that would rapidly move sediment on, it is unlikely that this dredging impact would persist or create permanent changes to fauna at these two sites.

There is a lack of information about deeper reef communities that may exist in Western Port. Chidgey *et al*. (2009) noted that 'Areas of firmer seabed may exist in the deeper channels that provide sufficient habitat for establishment of patches of sponge communities.' It should be noted here that sponges and other invertebrates, such as sea-pens and ascidians, do not necessarily need to attach to reefs. They may also attach to dead shells and small rocks embedded in soft sediments. Sponges communities were reported on the seabed of the deep channels offshore from Crib Point in the 1970s (Chapter 5). Aerial surveys conducted in the early 1970s for the Westernport Bay Environmental Study revealed features at depths of 16 m in some areas, such as the North Arm (NSR 1974). Sponge beds occupied at least 2 km of the 15 – 20 m channel off Hastings and were visible in the aerial photographs as a faint mottling (Chidgey *et al*. 2009). Because water clarity rapidly declined in the late 1970s it is now impossible to determine seabed features in water depths greater than approximately 5 m in the North Arm (Chidgey *et al*. 2009).

The existing information is largely static descriptions of fauna or flora, usually without much taxonomic resolution. A few studies have described long-term trends, but there is little information about short-term variation. The available information is confined to a few places that have been visited regularly, generally by a few individuals (e.g. Crawfish Rock and the long-term monitoring associated with shipping facilities on the western edge of the bay, both associated with studies by Dr J. Watson). No information is available on the ecological interactions in these habitats or species that are ecologically important.

More protracted ecological studies are associated only with particular locations (e.g. investigations of fouling communities at Hastings by Webb (2000), and detailed investigations of the ecology of the barnacle *Elminius covertus* on pier pilings at Rhyll (Wright 1996) and on hard surfaces provided by mangroves (Satumanatpan *et al*. 1999, Satumanatpan and Keough 2001). These studies did not provide any information applicable to the natural reefs of Western Port.

Despite the absence of detailed information, it would be assumed that habitat forming species, predominantly algae such as *Ecklonia* subtidally and *Hormosira* intertidally, would be ecologically important, as would be herbivores that consume them. This is based on the extensive studies on such species elsewhere in southern Australia and New Zealand (Sharpe and Keough 1998, Schiel and Hickford 2001, Connell and Vanderklift 2007, Smale *et al*. 2010). There is no direct information available for Western Port.

Similarly, without an understanding of temporal dynamics and important ecological processes, it is hard to predict how fauna and flora in these reef habitats would respond to additional stresses, or how they would recover if, for example, levels of suspended sediments were reduced. Although some information is available for more open coast reefs and large embayments (e.g. St Vincent Gulf, South Australia), the extent to which it can be extrapolated to enclosed embayments such as Port Phillip Bay and Western Port is unclear. This knowledge gap has been acknowledged for Port Phillip Bay, and a research program is underway to provide some of this information (DSE 2010, Hutchinson 2010).This information will be the most relevant for Western Port. Extrapolation may be simplest for the reefs in southern Western Port that are more open to the ocean, but more uncertain for areas such as Crawfish Rock that lack any Port Phillip Bay equivalent.

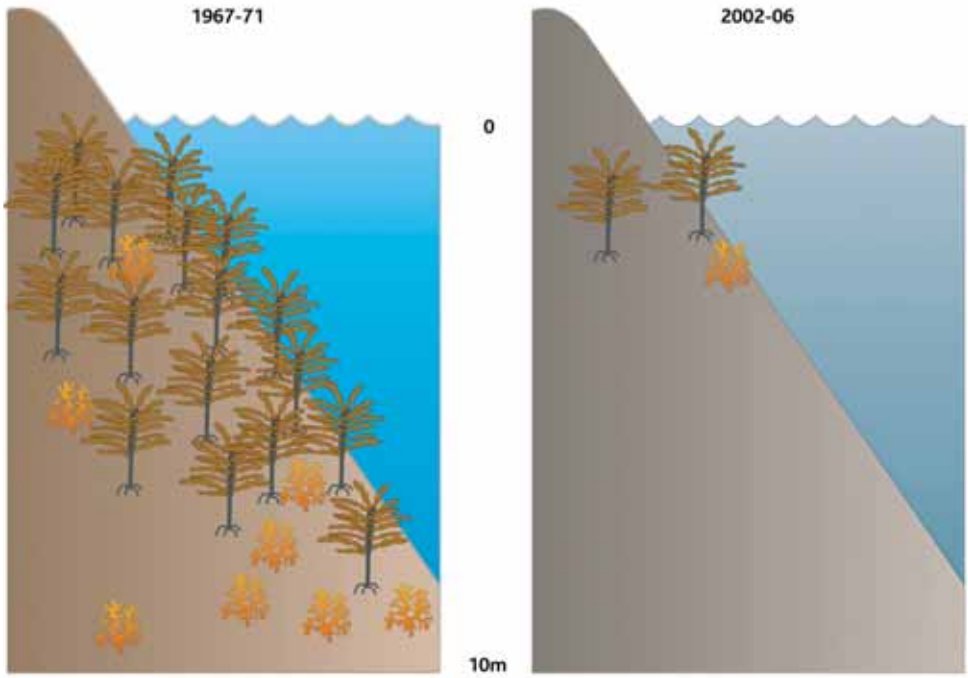


Figure 13.5 Changes to depth distribution of kelps and red algae at Crawfish Rock between the early 1970s and mid 2000s. The figure shows patterns for high current flow areas at this site, with algae now confined to very shallow depths.
Redrawn from Shepherd *et al.* (2009)

Major threats

Suspended sedimentation

Risks

Suspended sediments pose several important risks to reef biota. They can reduce light levels, inhibiting algal photosynthesis and can also affect filter-feeding animals by clogging their feeding apparatus, or by forcing animals to expend energy removing sediments from feeding structures. When material leaves the water column, deposits of material on algal surfaces can inhibit photosynthesis, and high rates of deposition can smother biota.
The accumulation of sediments can have adverse impacts on sessile temperate reef algae and invertebrates, reducing richness and abundance, whilst favouring small opportunistic taxa (Thomsen and McGlathery 2006).
In the intertidal zone, sedimentation and sand scour can significantly change the structure of algal assemblages and may inhibit movement, attachment and feeding in grazing invertebrates (Atalah and Crowe 2010).

Consequences

Deposition of sediments can be fatal if sediment depth is high and sediments remain. On reefs with little vertical relief, this can result in burial of the reef and effective loss of habitat. Where hydrodynamic processes are substantial enough and sediment resupply is low, burial and clogging can be transient. Many invertebrates and algae can tolerate brief periods of burial, reduced feeding or photosynthetic efficiency. Where resupply of sediments is high, persistent turbidity can affect algae and may reduce the growth rates of filter-feeding invertebrates.

Any activity that will result in the resuspension of bay sediments should require a site-specific assessment of the extent of any impact on local reef biota.

The hydrodynamic environment of the western shoreline and the eastern entrance prevents fine sediments from accumulating, and suspended sediments are not a serious risk in this segment.

High levels of suspended sediments in the North Arm have been implicated in the loss of some algae and the contraction of algal assemblages to shallow water at Crawfish Rock (Shepherd *et al.* 2009). While this is not based on formal monitoring or formal examination of algal thresholds, we consider it a convincing explanation. It is consistent with changes associated with seagrass declines and may be an ongoing threat that is likely to be an issue for other subtidal reef areas in the North Arm, Corinella and Rhyll segments.

Marine pests

Some non-native species are already well established in Western Port, predominantly around Hastings and other ports. Most of these are fouling species (Currie and Crookes 1997, Parry and Cohen 2001) and are often associated with artificial structures (Keough and Ross 1999). Some have spread beyond this area; for example, the bryozoan *Watersipora subtorquata*, introduced to Victoria in the 1970s, is present at Hastings, but also in the relatively open ocean environment of Flinders attached to the pier. It has also spread widely in Port Phillip Bay. Other non-native fouling species can be established on soft sediments as well. In Port Phillip Bay the ascidians *Ciona intestinalis, Styela clava* and *S. plicata* can form large aggregations away from reefs and piers. The European Shore Crab *Carcinus maenas* is already well established in Western Port and is known to exclude native crabs from its immediate habitat (Sinclair 1997).

Several other pests of concern might occur in Western Port. The brown alga *Undaria* was not recorded by Parry and Cohen (2001) in their surveys. At the time, however, it was not present in large numbers elsewhere in Victoria. It has subsequently become widespread in Port Phillip Bay and is also established along the open coast at Apollo Bay. It was not noted by Stewart *et al.* (2007) but there have not been systematic surveys since Parry and Cohen's investigation. Dr J. Watson has been following some sites on the western side of Western Port for nearly 40 years, but she has not observed these species (Watson 2009). *Sabella spallanzanii* has been found on aquaculture mussel ropes, but not near farms (Cohen *et al.* 2000). Improved treatment of mussel ropes is expected to prevent further translocation of this species to Western Port.

The green alga *Codium fragile* subsp. *tomentosoides* has been present on reefs in Western Port since 1998 and is considered a threat to shellfish and aquaculture because it attaches to oysters and mussels and removes them from natural substrata (Campbell 1999, Ferns and Curnow 1998). The EPA marine science unit studied populations at the San Remo marine community and nearby Newhaven (Campbell 1999). At Newhaven, *Codium* plants were found growing on *Hormosira banksii* in the low intertidal zone with anecdotal evidence of negative effects on the density of grazing invertebrates. In New Zealand *Codium* has been found to exclude *H. banskii*, (Dromgoole 1975) and, when abundant, may compete for nutrients and therefore alter the nutrient uptake and recycling rates of other macroalgae. Studies to determine such species interactions have not been conducted for *C. fragile* in Western Port. Eradication attempts in Western Port and at other locations have been unsuccessful (Campbell 1999), and one year after detection the species had spread several kilometres.

Risks

The brown alga *Undaria* is considered a high risk species. Specimens at Flinders were removed at an early stage of infestation, thus preventing further spread (Parry and Cohen 2001). Transport from Port Phillip Bay to Western Port may occur via recreational vessels or on eastward currents from the entrance to Port Phillip Bay. The recreational risk is related to increasing population, and is most effectively managed by educating recreational boat users. There is no obvious risk management tool for natural dispersal from Port Phillip Bay.

The polychaete *Sabella spallanzanii* is in the same category as *Undaria*, with potential spread from Port Phillip Bay by natural means. In Port Phillip Bay, *Sabella* are widespread on hard surfaces, but most individuals occur on soft sediments. It does affect recruitment of other animals on hard surfaces (Holloway and Keough 2002), but risks of natural dispersal can not be reduced by any obvious management action. Its presence on some hard surfaces would not trigger major concern about ecological impacts.

In addition to the San Remo marine community, introduced *Codium* has spread to areas within Churchill Island Marine National Park and is able to anchor on shells and small rocks in fine sediment environments (Campbell 1999, Curnow 1998).

Several species of national concern could colonise rocky reefs (Hayes *et al.* 2005). The management of risks should focus on preventative actions, and should be addressed as part of studies associated with any port development.

In Port Phillip Bay, rocky reefs in the more oceanic southern part of the bay are not invaded by the more serious pests, and many invasive species are associated with artificial structures (particularly marinas and piers in low energy environments) and more sheltered rocky reefs (Keough and Ross 1999, Valentine and Johnson 2003). This suggests that the risk to the more extensive reef areas of Western Port, which are in the southern sections, is not great.

Consequences

Hormosira banskii and other native algae are at risk from direct attachment to fronds by *Codium*, with possible interspecific competition between the two species for attachment space and other resources. Curnow (1998) stated that there is a high potential for *C. fragile* ssp. *tomentosoides* to threaten the ecological integrity and biodiversity of important and unique local habitats through competition with other algae and exclusion or avoidance of grazing invertebrates.

Pathogens

Recent marine pathogen outbreaks in southern Australia include two mass mortality events of *Sardinops* in 1995 and 1998–89 caused by a herpes virus originating from tuna farms in South Australia (Gaughan *et al.* 2000). Abalone in southern Australian waters are affected by a fatal herpes-like virus known as ganglioneuritus, which also has its origins in aquaculture operations in south-western and central Victoria (Gavine *et al.* 2009). The number of recorded diseases in the marine environment has been increasing over the last decade, with predictions that most host–parasite systems will experience more frequent or severe disease impacts with warming, with increased transmission rates and host susceptibility (Harvell *et al.* 2002; Lafferty *et al.* 2004). While many disease-related mass mortalities in the ocean are associated with warming waters, there remains a lack of baseline data required to distinguish climate effects and other anthropogenic disturbances (Harvell *et al.* 2002).

Risks

Pathogen risks are poorly known but are expected to rise with warmer sea temperatures (Harvell *et al.* 2002, Lafferty *et al.* 2004). Pathogen risks can also be increased through intensive aquaculture. Aquaculture is not specifically reef related, but a risk to be watched. Management tools include license conditions for aquaculture leases and inspection requirements associated with export permits.

Consequences

Conseqeunces include population decline in affected species, fisheries collapse and associated economic loss. There are also potential (trophic) consequences for predators (e.g. sardines for penguins) and alterations to reef biological assemblages (e.g. abalone and algal growth).

Pathogens are poorly known, but there is no evidence that the risk for Western Port is higher than for other areas of southern Australia.

Nutrients

Increased availability of nitrogen and phosphorus from anthropogenic sources can have both stimulating and adverse impacts on algal growth. While nutrient enhancement may stimulate the growth of macroalgae, it may also favour the growth of epiphytic algae which can smother and shade underlying macrophytes and sessile fauna. Increased nutrients from sources such as sewage effluent can also cause the loss of important habitat-forming algae, as is the case with *Phyllospora comosa* in New South Wales (Coleman *et al.* 2008). On intertidal reefs, nutrient enrichment can cause a decline in the abundance of perennial macroalgae, and an excessive growth of filamentous and mat-forming algae such as *Enteromorpha* and *Cladophora*, leading to changes in the structure and functioning of marine communities (Worm *et al.* 1999; Arundel *et al.* 2009). Anthropogenic eutrophication reduced brown algal diversity on intertidal reefs in New South Wales (Russell and Connell 2005), (Coleman *et al.* 2008) and caused the loss of *H. banksii* and Bull Kelp *Durvillaea potatorum* around the Boags Rocks sewage outfall (Brown *et al.* 1990).

Photo courtesy Michael Keough.

Risks

Elevated nutrient levels in the Corinella–Rhyll segment may mean this area is vulnerable (to algal blooms, but see Chapter 14) but there is little significant reef in this area, and flushing in the remainder of the bay plus low nutrient inputs suggest other areas are unlikely to be affected. Localised high nutrient inputs from some creeks, e.g. Merrick's Creek (Counihan *et al.* 2003) could affect nearby intertidal and shallow subtidal reefs such as Merricks Reef and Honeysuckle Reef. Nutrient pollution from sewage occurs in a number of areas (EPA 1999). As of 2003 the town of Somers was unsewered, and run-off from the area was identified as a potential threat to nearby Honeysuckle Reef (DSE 2003). This area is now sewered, with houses being connected progressively.

In general, nutrients are considered to be low in Western Port, although elsewhere in this report, we identify a need to better characterise nutrient dynamics, particularly in northern Western Port. This is a priority more associated with soft sediments and seagrasses, but at this stage nutrients are not considered a major risk to reefs.

Consequences

The loss of habitat-forming macroalgae has major effects on the diversity and abundance of associated fauna. Trophic effects at several levels may occur as a result of changes in the amount and source of detrital inputs into marine food webs, and there can be impacts on the diversity and abundance of fauna that directly or indirectly use the structure provided by macroalgae as habitat (Bishop *et al.* 2010). Algal growth in the intertidal zone is often regulated by grazing gastropods, but the unregulated removal of gastropods combined with high algal growth caused by eutrophication can exclude adult animals and recruits and create permanent shifts in intertidal community structure. Also, the decomposition of large amounts of plant biomass can result widespread hypoxia or anoxia (Arundel *et al.* 2009).

These effects typically occur as a result of very high levels of nutrients, as a result of substantial inputs and/or long retention times. Neither of these situations is applicable to the main reef areas of Western Port.

Increased temperature

The likely temperature-driven (climate change) impacts on temperate reefs are range-shifts of both macroalgae and invertebrates, local extinction of species that have northern range limits along the southern coastline (i.e. no poleward range shift possible), and changes in species' life cycle events that are influenced by seasonal and interannual variations in climate (Wernberg *et al.* 2009). Another potential effect for reef species with dispersing larvae is an accelerated (i.e. shortened) larval development time, thereby reducing the mean distance of larval dispersal, with potential consequences for connectivity of populations on individual reefs.

Wernberg *et al.* (2009) suggest that the combined effects of increased temperature and non-climate stresses (pollution, reduced water quality) will make Australian temperate reef communities more vulnerable to perturbations (e.g. storms, diseases, invasive species) many of which are projected to increase in frequency and/or severity in response to climate change. Reduced resilience is predicted to lead to the loss or alteration of algal habitats and associated ecological function – changes which will happen progressively from 2030 to 2100. A decrease in resilience of temperate species is a likely consequence of physiological adjustment to elevated temperatures (Wernberg *et al.* 2010). For example, cold-water species of algae such as kelps are likely to become less abundant as ocean temperatures increase towards their tolerance limit (Wernberg *et al.* 2010). In the intertidal zone, increased temperatures may also lead to desiccation of algae and seagrasses and mortality of animals through heat stress.

Although temperature effects are important they are beyond the scope of this report, which is focused on short to medium-term management, and we do not consider them further. Western Port does have a wide temperature range, typical of shallow bays, so it is possible that some resident species may not be threatened in the short-medium term by changes in mean temperature. The likelihood of increased temperatures is very high, but the exact ecological changes are unknown and the consequences are hard to assess at this time.

Sea-level rise

Rising sea levels are an important issue for intertidal reefs but less so for subtidal reefs. Intertidal zones will move shoreward, but the extent to which this is possible depends on geomorphology, where the question is about the existence of suitable habitat at higher elevations. If there is capacity for habitat migration, there may be issues associated with tenure of land above the current high tide levels. Artificial structures (groynes, sea walls, etc.) may provide reef-like structures and additional habitat for species that are displaced.

The consequences of sea-level rise may range from simple habitat migrations to complete loss of particular local assets, with no opportunity for relocation.

Alteration of physical coastal processes

Potential changes to physical processes are described in Chapter 4. For reefs, some of the effects of concern are increased storm intensity and frequency associated with climate change. This might cause the dislodgement and death of canopy-forming algae and animals that live on and around reefs. Wind speeds may also increase in intertidal areas, leading to increased turbidity near subtidal reefs and an increase in the magnitude of freshwater pulses (and sediments) to the bay.

The construction of permanent structures such as groynes and seawalls to mitigate the effects of climate change can alter coastal processes, but for reef biota these structures can also provide additional habitat. This additional habitat may be beneficial in the case of species threatened by habitat loss, or deleterious if it facilitates the spread of harmful species.

Risks

Changes to oceanographic processes may alter transport of propagules (larvae, spores, etc.) and change physical conditions, altering patterns of sediment deposition.

Assessing the risks from altered coastal processes requires an understanding of larval dispersal patterns and the role of currents. This information is not known for many species, even in well-studied areas. This is a substantial knowledge gap, which has not been narrowed even by extensive studies. In the case of Western Port, large changes are not expected, and this risk is considered low for reefs. A more formal assessment could follow any finding of dramatic alteration to coastal processes.

Acidification

One of the important consequences of climate change is expected to be an increase in ocean acidity (see Chapters 3 and 4). Calcareous algae and other calcifying organisms inhabiting reefs, such as gastropods, bivalves, crabs and urchins, will be adversely affected if oceans continue to increase in acidity (Wernberg *et al.* 2009). The direct impacts of increased seawater acidity on reef organisms include impairment of the calcification process (e.g. shell formation in gastropods, and the growth of coralline algae) hypercapnia (acidification of body fluids), decreased fertilisation and abnormal larval development. While the impacts to Victorian reef species are potentially severe they have yet to be determined, and studies under true field conditions have yet to be undertaken. Laboratory studies on the response of temperate Australian species to increased seawater CO_2 have shown negative or no effects (Havenhand *et al.* 2008; Byrne *et al.* 2010).

This is a potentially important issue, beyond the scope of any local management action. At present, predicting the responses of single species to increased acidity is difficult, in part because of the time scales involved. Rocky reefs include ecologically important species across many phyla that might be expected to be vulnerable to acidification, so the risks are generally high. Ecosystem consequences are even more difficult to predict, and range from mild to catastrophic.

We cannot identify a direct management action to mitigate this threat, but we will need to predict the kinds of ecological change, in case the impacts of acidification require adjustments to other management of Western Port. This will be required for the Victorian marine environment, not just for Western Port.

UVB

Ultraviolet (UV) radiation at the Earth's surface comprises UVA (320 to 400 nm wavelength) and UVB (280 to 320 nm). UVB is particularly harmful and can cause negative effects on reproduction, development, and behavior in many marine organisms. Solar UV radiation may reduce the photosynthetic uptake of atmospheric carbon dioxide (Häder *et al.* 2011). Many aquatic organisms can tolerate some level of UV stress, and may show adaptive responses such as avoidance strategies, repair mechanisms and the synthesis of UV-absorbing substances for protection. Global environmental change is predicted to increase the exposure of shallow water organisms, such as those in Western Port, to damaging effects of UV (for explanation see Häder *et al.* (2011). In addition, there are likely to be interactive effects of UV with other stressors such as desiccation, increased temperature, and pollutants (Przeslawski *et al.* 2004, Häder *et al.* 2011).

Risks

Climate change, acid deposition, and changes in other anthropogenic stressors such as pollutants alter UV exposure levels in coastal marine waters (Häder *et al.* 2011).

Consequences

The main consequence is a decreased health and resilience of organisms, lethal and sublethal mutation, reproductive failure and population effects. There is no obvious management response, and these risks will not be considered further here.

Extraction

Intertidal reef assemblages, particularly those near population centres, can be impacted by collection of invertebrates, and various species of molluscs are particularly targeted. Human traffic on intertidal algae can crush and dislodge important habitat-forming algae (e.g. *Hormosira banksii*). The collection of many invertebrates in the intertidal zone in Western Port is prohibited under the Victorian *Fisheries Act 1995*, but in Port Phillip Bay, legislation alone has not prevented collection (Keough & Quinn 2000). There are no qualitative data on extraction of intertidal reef biota from Western Port, but the impacts of extraction, such as reduced

reproduction in broadcast spawning gastropods and changes to the species abundance and diversity, are well documented in the literature from Victoria and elsewhere.

Signage at access points for visitor education is either not highly visible, or is non-existent. There are no qualitative data on extraction of intertidal reef biota from Western Port, but the impacts of extraction, such as reduced reproduction in broadcast spawning gastropods and changes to the species abundance and diversity, are well documented in the literature from Victoria and elsewhere.

The Marine Research Group (Handreck and O'Hara 1994) noted that around 17% of 204 intertidal or shallow subtidal invertebrates from the Flinders, Shoreham and Point Leo areas have not been recorded since 1970, with 60% recorded prior to, and since 1970. They caution that there is increasing evidence of reduced abundance of many species since the 1980s, particularly cowries, some crabs and large molluscs (e.g. *Cabestina spangleri*). They also note a decline in muricid (predatory) snails from areas such as Reef Island as populations of edible mussels, among which they seek smaller food species, have been progressively reduced by recreational harvesting (Handreck and O'Hara 1994).

Subtidal reefs

There is significant recreational fishing pressure in Western Port (Chapter 11). With the advent and accessibility of technology, such as GPS and 'fish finders', this pressure is significantly compounded. With these contemporary fishing aids, reefs and other features where fish accumulate, can be easily identified and located. Internet listing of the coordinates of many of these sites means that previously 'secret' fishing spots are now widely advertised. The direct impact of recreational fishing pressure on subtidal reefs is currently unknown (see Chapter 11). Important recreational fish species such as snapper and King George whiting utilise reefs among a mosaic of other habitats – including unvegetated sediments, seagrass meadows and invertebrate aggregates.

Risks

Effects of extraction and human trampling on intertidal reefs are likely to increase with urbanisation of surrounding coastal areas unless accompanied by a significant increase in compliance and enforcement. Extraction of reef-associated fish is also likely to increase with urban expansion.

Consequences

Depletion of intertidal reefs has been recorded for reefs close to Melbourne (Keough and Quinn 2000), but has not been detected readily along the open coast, between Western Port and Port Phillip Bay (Bathgate 2011) or to the east of Western Port (King 1992). The levels of visitation of Western Port reefs are more likely to resemble those of the outer coast than metropolitan Melbourne, so harvesting is not regarded as a serious risk.

No management action would be associated with intertidal exploitation. Port Phillip Bay and Western Port are already under broad legislative protection (*Fisheries Act 1995* (Vic)), and the effectiveness of this legislation depends on education, community support, and enforcement.

The small number of reefs within Western Port would make it difficult to assess any effects of intertidal extraction, which might be concentrated in the southwest of the bay, and it would be hard to find enough sites that might serve as reference. Therefore, while lack of knowledge of intertidal exploitation is a gap, the geographic limitations of Western Port mean that we do not advocate any research, except via student projects.

Toxicants

A range of toxicants enters Western Port, from diverse sources. Many of these sources, particularly the northern catchments, are not close to reef areas, and when sources are close, the overall contaminant levels are not high.

Some heavy metals appear to be elevated in Western Port, particularly around industrial sites and areas where boating activity is concentrated, such has jetties and marinas. Webb and Keough (2002) found that zinc and lead levels in the water column at Hastings jetty were unchanged since the 1970s, but that levels of copper and cadmium were significantly higher. The source of copper is most likely the antifouling paints applied to boats and related structures. Copper is toxic to aquatic life and can have sublethal effects, e.g. on the growth and reproduction in reef organisms. Larval stages can be particularly sensitive to the effects of heavy metals.

Rees *et al.* (1998), studied sediments in the bay for attainment and trends in toxicant concentration since the major studies undertaken in the 1970s. They found that most metals, organics and organo-metallics were below ANZECC (2000) guidelines for sediment quality. They found elevated levels of arsenic in clay-silt fractions and in streams draining the Koo Wee Rup swamp in the north, most likely of geological origin (EPA 2008).

Tributyl tin is associated with shipping facilities, particularly commercial operations. In Western Port, this means the area around Hastings. There is no evidence of substantial organotin contamination, but there have not been extensive investigations. Any such effect would be expected to be localised, and changes to the risk level would be associated with the mobilisation of contaminated sediments or increased shipping. The areas where this might occur are not close to reefs, and any effects would be more likely to be associated with organisms living on artificial structures. While the bay-wide risk is not considered high, organotin contamination would require some consideration in any port or marina expansion.

Risks

Overall, contaminant sources are well separated from most reef areas, and it is expected that contaminant effects would be seen in other habitats before any effect reached reefs. Risks overall are considered low for reef habitat.

Consequences

Detected levels of metals, etc. are below trigger levels for individual toxicants, and overall inputs into Western Port are less than is characteristic of overseas locations in which contaminants affect biota. In general, contaminant effects are more associated with soft-sediment environments, perhaps because rocky reef areas tend to be better flushed.

13 Rocky reefs

Photo courtesy Michael Keough.

Research that can fill key knowledge gaps

The major threat associated with reefs is from sediments, through the effects of suspended sediments on light penetration and the deposition of sediments. These effects are expected to occur in the northern parts of Western Port. Reducing suspended levels would be beneficial for those reefs. Invasive species are a concern, and removal of animals from reefs is a risk, although it is not well quantified.

There are serious long-term risks associated with climate change, particularly loss of intertidal reef habitat through rising sea levels, changes to fauna from increased temperature and potential range extensions of species from lower latitudes, and ocean acidification, which may have profound consequences.

The management of reef habitats is based on understanding the nature of these assets. Reefs in Western Port support recreation and are valued for their biodiversity. There is some evidence to identify important assets, such as Crawfish Rock or Honeysuckle Reef, but these data are limited. There have been no recent systematic surveys, and some potentially important habitats such as the walls of deeper channels have not been surveyed at all.

There is a suggestion of changes to reefs in northern Western Port that is related to sedimentation, but little ecological knowledge to know how much reduction in suspended sediments would be necessary to reverse these changes, and of the time scale on which any recovery might occur. This lack of ecological information extends to questions about how resilient reefs are, and the common knowledge gap of connectivity. In the case of Western Port this translates to a lack of knowledge about whether northern reefs are largely isolated, or whether they form a network that is well connected by currents.

There is also no recent information on marine pests on reefs. This prevents any assessment of the condition of reef assets and extent to which they may have been degraded by invasion.

Research that can fill key gaps

We have identified a range of research needs that will develop better understanding of rocky reefs in Western Port. This research would provide a better understanding of the biodiversity value of different rocky reefs and the resilience of particular reef areas, but it would not directly alter management actions in the short term.

The major threats identified to reefs are related to sediment levels in Western Port, and the need to reduce resuspension rates and the introduction of new fine sediments, and the major issues are covered elsewhere.

Research needs:

1. Better characterise the biodiversity of rocky reefs and related habitats. Although there have been past surveys of individual areas such as Crawfish Rock, most of the reef areas have not been surveyed extensively, and the fauna of channel walls is poorly known. We do not necessarily advocate formal protection for any important assets, because the protection afforded by MPA status would not mitigate the major threats to locations such as Crawfish Rock. Indeed, Dr J. Watson (pers. comm.) has suggested that the turbid conditions and strong currents prevent much recreational use of this area, and we argue that the major threats are linked to water quality, particularly turbidity.
2. Thresholds for successful growth of brown algae. The loss of canopy-forming brown algae in northern parts of Western Port has been a concern. While suspended sediments are the most common cause for this decline, there is a possible role for nutrients and contaminants derived from the north-eastern section of Western Port. These other influences may combine with suspended sediments in limiting algae. A better understanding of these limits to growth would allow refinement of water quality targets for the northern sections of Western Port:

Photo courtesy Michael Keough.

a. Is growth (and presence) of habitat-forming brown algae limited by suspended sediments alone in northern parts of Western Port, or do nutrients and catchment-derived contaminants play a role?

b. If so, do sediments, nutrients and contaminants act simply (additively) or do they have synergistic effects.

c. What are the relationship between levels of these stressors and growth of healthy seaweeds?

This information is desirable for Western Port, but it is not regarded as a high priority for two reasons. First, seagrasses typically have a higher light requirement, so it is likely that water criteria developed for seagrass will encompass light requirements for brown algae. Second, if this information is needed, there is current work from South Australia, and a recently commissioned project for Port Phillip Bay, which will provide similar information. An independent algal investigation is not advocated for Western Port.

3. How vulnerable are intertidal rocky reefs to sea-level rise, and what scope is there for the migration of intertidal fauna and flora? Many of the intertidal reefs have a low relief, often without substantial areas above current high-water levels. The assessment will need to include:

a. The likelihood of complete submergence of these reefs, along with the availability of suitable habitat above the current high water mark.

b. The potential for other habitat areas to become available as a result of other development in Western Port or through coastal hardening in response to sea level rise. This assessment should include the suitability of constructed habitats for intertidal reef biota and also the potential for additional habitat to facilitate the spread of invasive species.

4. Develop an understanding of whether recreational harvesting of fauna is a threat to intertidal reefs. Increasing population in Melbourne's south-eastern suburbs might be expected to increase recreational pressure on reef areas. This pressure might be expected to be greatest on the reefs of south-western Western Port, which would be easily accessible from population growth corridors, and less on reefs of northern Phillip Island, access to which will still be somewhat restricted. Evidence from elsewhere along the Victorian coast suggests that such effects are weak in areas outside metropolitan Melbourne. This item is considered a low priority for substantial investment, but might be appropriate as a postgraduate study.

5. Reassess risk from invasive species. The most recent assessment for Western Port is nine years old. The most likely vectors for translocations (new introductions or range extensions from Port Phillip Bay) are commercial and recreational vessels, which are likely to be focused in the north-western section of Western Port, particularly around Hastings.

6. As a long-term research need, develop a better understanding of the likely effects of increased ocean acidity.

14 Ecosystem processes

Jeff Ross

Photo courtesy Parks Victoria.

Natural ecological processes underpin the important habitats and the diverse range of animals they support in Western Port and provide key ecosystem services. This chapter focuses on the various processes involved in the cycling of nutrients and primary production.

Understanding if and where nutrients accumulate in marine systems is an important element of any environmental management strategy, particularly where nutrients are considered a major threat. In Port Phillip Bay it is well established that the process of denitrification in subtidal sediments permanently removes much of the excess nitrogen that enters in the bay. Westernport Bay, however, is very different to Port Phillip Bay and our understanding of nutrient cycling there is inadequate. Over a third of Westernport is intertidal seagrass and bare mudflat yet we know little about nutrient transformation in these environments (elsewhere in the world, similar mudflats have been shown to be either sources or suppliers of nutrients to the marine ecosystems).

The decline and limited recovery of seagrass in the eastern section of the bay is symptomatic of nutrient and sediment loads exceeding the system's capacity to process and assimilate them. However, our understanding of the ecological thresholds of the major habitat forming primary producers such as seagrass and the consequences of a habitat shift for nutrient and sediment dynamics is limited. In the absence of this knowledge our ability to prioritize nutrient and sediment reduction strategies is constrained

A multi-stage research program is proposed that would develop a nutrient and sediment budget for Westernport, identifying key areas and habitats for the transformation and removal of nutrients and the settlement and resuspension of sediments. The recommended stages will allow a rapid assessment, which will determine the need for detailed formal measures of nutrient cycling, and the need for a formal process based model for Westernport. Such a model, coupled with improved sediment and hydrodynamic models, will allow detailed exploration of the benefits that would be expected from alterations to catchment inputs of nutrients and sediments, but also the capability to predict the response of the Westernport ecosystem to future climate scenarios.

Ecosystem processes in Western Port

The important habitats and the diverse range of organisms they support in Western Port (see Chapter 2) depend on natural ecological processes such as nutrient cycling to maintain ecosystem health. In this chapter we review the current scientific understanding of key ecosystem processes and the major threats to these processes, and identify major gaps in our knowledge. In aquatic ecosystems the various processes involved in the cycling of nutrients are critical to ecosystem structure and function, because the major primary producers — phytoplankton, microphytobenthos, macroalgae and macrophytes — are often limited in their rates of production and growth by the availability of nutrients, in particular nitrogen and phosphorus. (See Box 1 for a general description of key processes involved in the cycling of nitrogen and phosphorus in coastal bays and estuaries.) The dynamics of nutrients and primary production affects a range of other ecosystem processes and services. For example, seagrass meadows help stabilise sediments, reducing turbidity, provide important habitat for many fish, invertebrate and plant species, and they also produce oxygen, store carbon and provide an important food source for many organisms.

This review focuses on our understanding of the cycling of nutrients and primary production in Western Port, given the dependence of a wide range of ecosystem goods and services. This is particularly important because two of the major threats to Western Port — nutrient and sediment loads from the catchment — could affect nutrient cycling processes and primary production (Chapter 3). This chapter will cover what is known about the major sources of nutrients and sediments in Western Port, with a review of internal nutrient cycling processes (including the influence of sediments) and the ultimate fate of nutrients (e.g. export to Bass Strait, burial or denitrification).

Distribution

The bay's hydrodynamics largely determine how nutrients and sediments are transported throughout the system, including their potential fate. For example, the residence time of nutrients entering the bay, and thus the time available for biological uptake, ranges from days in the Western Entrance to months in the Eastern Arm (Hinwood 1979).

The hydrodynamics of Western Port, including the division of the bay into five segments based on biophysical characteristics (Figure 1.3), is detailed in Chapter 4. To provide a context for this review, the important characteristics of each segment of the bay that play a key role in driving and shaping nutrient cycling processes and primary production are summarised here, based largely on the review by Counihan *et al.* (2003).

Box 1
Nutrient cycling

For the purposes of this report, a general description of the key processes involved in the cycling of nitrogen and phosphorus in coastal bays and estuaries is provided. For a detailed review of the biogeochemical cycling in estuaries, see Bianchi (2007).

Nutrients are conveyed to coastal waterways from catchment, atmospheric and oceanic sources in either dissolved or particulate forms. Nitrogen fixation is also a source of nitrogen in coastal waterways and is often associated with seagrasses and benthic microalgae.

Nitrogen cycling

In the water column, dissolved nitrogen (ammonia and nitrates) is taken up by phytoplankton, which can then be consumed by zooplankton and then other organisms at higher trophic levels. Ammonia is produced either by excretion or through the microbial breakdown of detrital organic material. The detrital organic material may originate from external sources or from organisms in the bay. The detrital material that reaches the sediment is also subject to microbial breakdown, which releases the ammonia into pore water within the sediment. Nitrifying and denitrifying bacteria can convert ammonia to nitrates, and nitrates to N2 gas, which is then lost to the system. Nitrate and ammonia in pore water can also diffuse back to the water column. In Port Phillip Bay, nitrogen does not accumulate in the system despite large loads, because of the efficient nitrification and denitrification processes.

Benthic (microalgae, macroalgae, seagrass) producers1 play a central role in regulating these fluxes and processes, particularly in shallow systems like Western Port. Benthic microalgae live within surficial sediments, taking up dissolved nutrients from pore water. Macroalgae typically take up most of their nutrients from the water column, whereas seagrasses also take up nutrients from the pore water. The organic material formed during primary production can then enter the food web before it is recycled back into the system during metabolism and microbial breakdown of detrital matter. Importantly, the producers interact with the nutrient cycling in different ways e.g. they take up nutrients from different places, at different rates and in different ratios, and provide a variety of different feedbacks, both positive and negative. Seagrass beds, for example, can enhance the trapping of suspended organic material, and benthic microalgae oxygenate the sediments, promoting nitrification. Thus, the composition of primary producer habitats has major implications for the transformation and ultimate fate of nutrients.

Phosphorus cycle

The phosphorus cycle is very similar to the nitrogen cycle, but there are a few key differences associated with availability. Inorganic phosphorus generally occurs in two forms: dissolved, and available or absorbed to particles. The absorption and desorption of inorganic phosphorus is complex, with different fractions likely to vary significantly in exchange times. Fractions with long to very long exchange times play an important role in the storage of phophorus in sediments and its release into the water column. For example, bottom water anoxia, higher temperatures and reduced salinity are known to enhance phosphorus release from sediments to the water column, where it is available to algae, plankton and seagrasses. Sediments exposed to air have a greater capacity for phosphorus adsorption compared to subtidal sediments.

1 This also includes mangroves and saltmarsh plants, which are covered in Chapters 8 an 9.

Western Entrance

At present the catchment inputs of sediments and nutrients to the Western Entrance segment are relatively small because of the small number of freshwater sources. Because there is a large exchange of water with Bass Strait as a result of the width of the entrance and predominant swell from Bass Strait towards the Western Entrance, these inputs are not likely to lead to water quality issues and concomitant shifts in the balance of the key nutrient cycling processes or the composition and biomass of primary producers. The high water velocities combined with relatively coarse sediments in the Western Entrance are likely to lead to moderate rates of resuspension and deposition, but the sediment is likely to be suspended only for short periods. The intertidal area in the Western Entrance is fairly small, so mangroves, saltmarshes and intertidal seagrass meadows (including macroalgae and microalgae) are likely to play a minimal role in nutrient cycling (Figure 14.1).

Lower North Arm

Catchment inputs of sediments and nutrients to the Lower North Arm are also relatively small due to the relatively small number of freshwater inputs. However, high concentrations of sediments and nutrients have been recorded in waters entering from Watsons Creek. There is a net flow of water into the Lower North Arm from the Western Entrance because of the net clockwise flow (Figure 14.2) of water around French Island caused by the shape of Western Port and tidal influences. As the other key source of nutrients and sediments will be from the Western Entrance where water column concentrations of nutrients are relatively low, the Lower North Arm is not likely to experience water quality issues, although the discharge from Watsons Creek represents a risk to localised ecosystem function in the Yaringa Marine National Park. Unlike the Western Entrance, the Lower North Arm has extensive tidal flats with a large cover of seagrass and extensive areas of saltmarsh and mangroves on the fringes. The subtidal area covered by seagrass is small compared to the intertidal area.

Upper North Arm

The Upper North Arm has relatively high inputs of sediments and nutrients from the catchment because of the large inputs of freshwater from Lang Lang River, Bunyip Drain, Yallock Creek and Cardinia Creek. In the Upper North Arm there are also significant sediment inputs due to coastal cliff and clay bank erosion, particularly on the northern shore. The tidal currents and range in the Upper North Arm are smaller and as such water residence times are longer (weeks to months). The combination of moderate currents and fine sediments (muds) within the Upper north Arm results in high rates of resuspension and deposition. Thus, the Upper North Arm is likely to have higher water column nutrient and sediment concentrations. Due to the net clockwise flow the Upper North Arm is likely to contribute nutrients and sediments to the adjoining Corinella basin (Figure 14.3). Given the large biomass of seagrass and fringing saltmarshes and mangroves on the large tidal flats in the Upper North Arm their extensive biomass is likely to play a significant role in nutrient and sediment dynamics.

Corinella segment

Catchment inputs of sediments and nutrients to this segment are very small because there are no significant terrestrial sources. The key source of nutrients and sediments is from the Upper North Arm. These inputs, combined with weaker tidal currents and ranges, lead to longer residence times (weeks to months) and higher water column concentrations of nutrients and sediments. Of the three long-term fixed sites in Western Port, the Corinella site clearly has the highest concentrations of nutrients and suspended solids, often exceeding the ANZECC guidelines and SEPP objectives (EPA 2008, 2011 – see Chapter 4). Chlorophyll concentrations at the Corinella site are also clearly higher than at the two other fixed stations, regularly exceeding the SEPP objective. The concomitant effects of high suspended solid concentrations on water clarity are evident in the shallow Secchi depths recorded at the Corinella site compared to the other two fixed sites. Despite the presence of extensive tidal flats, the area covered by seagrass is relatively small. Seagrass was extensive in the segment in the early 1970s, but there has been very little recovery since the large seagrass decline in Western Port between the early 1970s and 1980s, in contrast to most other areas in Western Port where there has been significant recovery.

Rhyll segment

Catchment inputs of nutrients and sediments to the Rhyll segment are moderate, primarily via the Bass River. A key source of nutrients and sediments to Rhyll segment is likely to be the Corinella segment, because of the net clockwise movement of water in Western Port. Although water is exchanged with Bass Strait through the Eastern Entrance, the exchange (and potential dilution) of water column nutrients and sediments is likely to be minimal because of the narrow entrance. This segment also exchanges with the Western Entrance segment, which is likely to lead to a net export of nutrients and sediments given the generally higher water column concentrations in the Rhyll segment. Tidal movement of water in the Rhyll segment is moderate to large and hence water residence times are reduced. The moderate coverage of seagrass and fringing saltmarshes and mangroves in the segment is likely to play a significant role in nutrient cycling and sediment dynamics.

14 Ecosystem processes

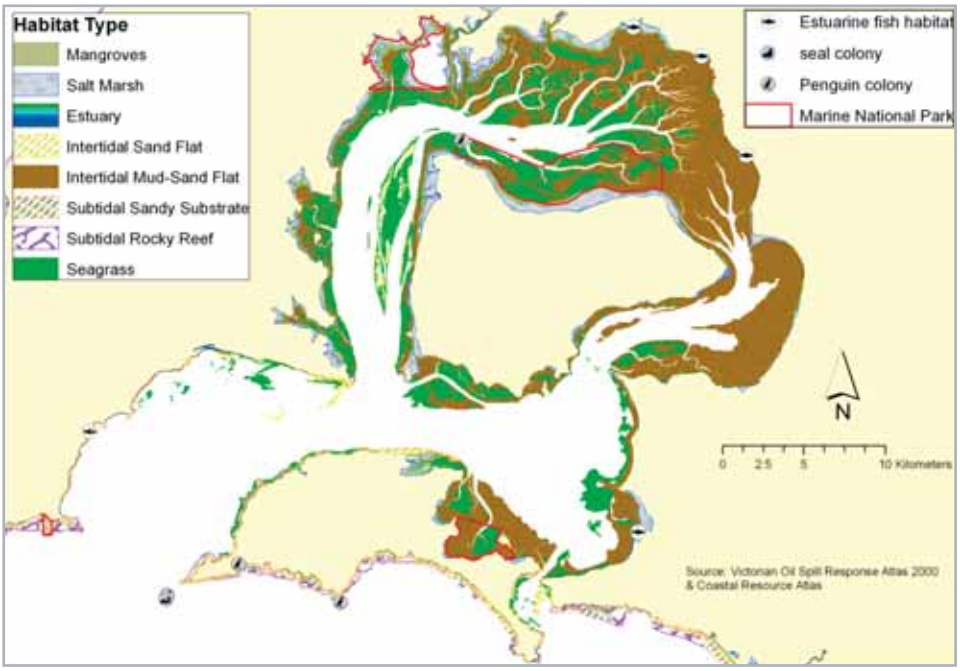


Figure 14.1 Marine habitat map of Western Port, showing the extensive intertidal flats that dominate the ecosystem.
(Source: EPA 2008.)

Figure 14.2 Water circulation in Western Port.
(Source: Hancock *et al.* 2001.)

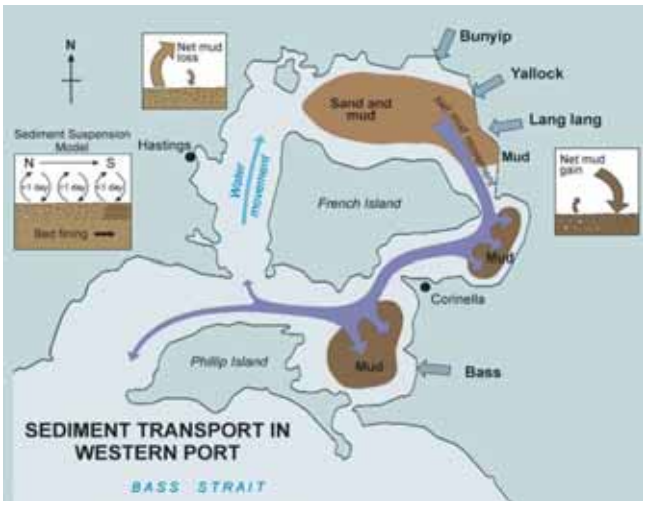


Figure 14.3 Suggested clockwise sediment redistribution in Western Port. (Source: Hancock *et al.* 2001.)

Special features

Tidal flats

A major feature of Western Port is the large expanses of intertidal mudflats — about 40% of the total area (Figure 14.1) — that support a mosaic of seagrass, macroalgal and unvegetated mudflat habitats (Figure 14.4). Although we still have much to learn about the ecological role of these habitats, particularly with respect to nutrient cycling (see below), there is little doubt that, because of their abundance, primary producers (including their functional and structural aspects) and the variety of organisms they support are central to the structure and function of the Western Port ecosystem. For this reason the extensive loss of seagrass between 1973 and 1984 was widely viewed as a serious concern (see EPA 1996). Although there has been significant recovery since, there has been little recovery on the eastern side of Western Port, particularly in the Corinella segment and the eastern section of the Upper North Arm segment. Understanding the consequences of shifts in the composition of primary producer habitats on the intertidal mudflats is critical because of the differences in the ecological processes and services they provide. Equally important for prioritising management actions is the need to understand the processes that are limiting the recovery of seagrasses.

Figure 14.4. Aerial oblique infra-red photograph of the embayment head (looking west) from the area of the tidal divide. Darker red-brown areas have significant Caulerpa, and paler areas are dominated by *Zostera*. (Source: Harris *et al.* 1979.)

Identifying and quantifying the major sources of nutrients (both dissolved and particulate) and the dynamics of their delivery in space (e.g. which segment) and time (e.g. pulse flood inputs versus constant base flows) is fundamental to understanding the response of the key processes controlling their fate and impacts on the Western Port ecosystem. This would also underpin the reliability and robustness of models developed to understand the current state and predict the effects of future changes on the bay's ecosystem. The inputs of sediments and the resuspension of sediments from the seafloor is also discussed because of the potential direct and indirect effects on nutrient cycling processes and primary production, particularly as sediment is known to carry nitrogen and phosphorus.

From a management viewpoint, identifying the key sources of nutrients is vital for prioritising mitigation actions because many of the environmental problems identified in Western Port, such as the high nutrient and suspended solid concentrations and the decline and limited recovery of seagrass in the eastern section of the bay (EPA 2008) are symptomatic of nutrient and sediment loads exceeding the system's capacity to process and assimilate them.

Sources

Diffuse and point sources

Because nutrient and sediment loads are recognised as the significant threats to the health of the Western Port ecosystem, there has been considerable investment in understanding and quantifying the loads in Port Phillip Bay and Western Port; most recently through the Water Quality Loads Monitoring Program (Melbourne Water 2009). To help identify pollutant sources and the priorities for pollution management, and to test alternative land management scenarios and actions, the PortsE2 model was produced and calibrated against water quality monitoring data (Melbourne Water 2009). The original E2 catchment model used for the PortsE2 model has now been redesigned (Stewart 2011; see Chapter 4). The new model, called WaterCAST, provides a closer match to measured flows for the Western Port region and can calculate uncertainties associated with total suspended solids and nutrient loads. It will be available later in 2011.

In the Western Port catchment, diffuse sources contribute a far greater fraction of the sediment and nutrients loads entering the bay than point sources (Melbourne Water 2009). The major inputs of total nitrogen, total phosphorus and sediments to Western Port enter the Upper North Arm via the Lang Lang, Bunyip and Cardinia catchments. Erosion of the shoreline in the Upper North Arm also contributes a significant sediment load to the bay, mostly as fine sediments (Counihan *et al.* 2003). Because of the net clockwise direction of water flow within the bay, much of the sediment delivered into the north-east of the bay is transported into the Corinella and Rhyll segments, where much of it is deposited (Hancock *et al.* 2001). There is also a major influx of nutrients from the Upper North Arm to the Corinella segments. Total suspended solids, nutrient and chlorophyll concentrations in the water column of the East Arm (Corinella segment) regularly exceed SEPP and ANZECC guidelines for Western Port (EPA 2008, 2011). This highlights the vulnerability of this region due to the large sediment and nutrient inputs, long residence times and high rates of sediment resuspension.

Atmospheric inputs

Airborne nutrients can be added to Western Port via rainfall, fog, and the fallout of gases and particulates. While atmospheric phosphorus loads are considered to be minimal (Harris *et al.* 1996), atmospheric sources of nitrogen are a significant input into Port Phillip Bay, estimated to be 1000 tonnes per year (Harris *et al.* 1996). While it is expected that loads to Western Port would be significantly less because of the much lower contaminant loads from vehicles and industries, further research is needed to determine how important atmospheric sources are to the loads in the bay. The recent combining of CSIRO atmospheric models (Hurley *et al.* 2005) and comprehensive EPA air pollution inventories will provide rates of atmospheric deposition in the region at fine spatial and temporal scales (see Chapter 4). EPA has since refined the estimate of atmospheric nitrogen loads to Port Phillip Bay that is less than the earlier 1000 tonnes.

Groundwater inputs

Counihan *et al.* 2003 noted that, before the Koo Wee Rup Swamp was drained, few streams discharged directly into the Upper North Arm of Western Port. Instead, the water was filtered through the wetland systems and entered the underlying groundwater. For Port Phillip Bay, analyses by Otto (1992) and HydroTechnology (1993) found that, while the loads of nitrogen and phosphorus from groundwater were not large, the concentration of nitrates in some aquifers was particularly high. It was predicted that there would be a minor rise in nutrient loads via groundwater as historical loads moved slowly into Port Phillip Bay (Harris *et al.* 1996). Predicted annual inputs of phosphorus and nitrogen were in the order of 8–25 tonnes and 34–82 tonnes respectively (Otto 1992, HydroTechnology 1993). There is no information available on the amount of groundwater and its associated nutrient concentrations entering Western Port, and further research is necessary to better constrain subsequent nutrient budget and modelling.

Nitrogen fixation

While rates of nitrogen fixation have generally been considered to be low relative to denitrification in estuarine and coastal systems (Harris 1999), there is growing evidence that a significant fraction of the nitrogen demand of benthic microalgae can be met by nitrogen fixation. This may be particularly important in Western Port, because the extensive mudflats undoubtedly support a large biomass of benthic microalgae. On the intertidal mudflats of the Huon estuary in Tasmania, Cook *et al.* (2004) demonstrated that nitrogen fixation, most probably by cyanobacteria, was likely to be the major source of nitrogen for benthic microalgae during summer at one of their study sites. In Gippsland Lakes, nitrogen fixation in the water column has also been observed; severely nitrogen-limited conditions over the summer months have been highly conducive to blooms of Nodularia, which can fix atmospheric nitrogen (Webster *et al.* 2001). High rates of nitrogen fixation are also often observed in sediments that support seagrass (Welsh 2000). Understanding the importance of nitrogen fixation in the major habitats in Western Port is thus a necessary component of future process studies.

Ocean inputs

Water exchange with Bass Strait via the western entrance is high because of the width of the entrance, whereas exchange via the eastern entrance is considerably less. Although oceanic inputs of nutrients are likely to be minimal because of the low nutrient concentrations in Bass Strait, the inflow of Bass Strait water containing effluent from the Eastern Treatment Plant, which discharges treated sewage at Boags Rocks to the west, is a potential source of nutrient influx into Western Port (EPA 2002). A pathway that could entrain the coastal discharge from Boags Rocks eastward into the Western Arm of Western Port was demonstrated by Black and Hatton (1994), and more recently by Harrison *et al.* 2011a). However, the long-term Western Port monitoring program undertaken by EPA Victoria has been confined to bay and catchment sources, so that quantifying nutrient exchanges with Bass Strait waters (including inputs of sewage effluent) is not possible at present.

Aquaculture

Aquaculture can also provide a source and a sink of nutrients and particulate matter. Although mussel aquaculture has a complex interaction with nitrogen cycling processes in coastal waters, there is a net removal of nutrients from the ecosystem through the harvesting of the mussels. A large aquaculture fisheries reserve off Flinders is used for low-level growing out of Blue Mussels, *Mytilus galloprovincialis*. At present levels, mussel aquaculture is likely to have a very minor role in nutrient cycling in Western Port.

Transformation and fate of nutrients

In shallow coastal systems like Western Port, benthic nutrient cycling processes and sediment – water column exchanges play a key role in shaping the structure and function of the ecosystem because the major primary producers are often limited by the availability of nutrients. One of the best-known Australian examples is Port Phillip Bay (see Longmore and Nicholson 2010, and references therein). Despite high nitrogen loads and a very long residence time, nitrogen does not currently accumulate in the system because of efficient nitrification and denitrification processes (Harris *et al.* 1996). Ammonia is produced following the initial breakdown of organic material. If oxygen is still available, the ammonia is oxidised to ammonium nitrate by nitrifying bacteria (nitrification). The nitrate is then available for denitrification, an anaerobic bacterial process that occurs within anoxic pockets in the sediments, in which nitrate is reduced to nitrogen gas. The nitrogen gas escapes to the atmosphere and is lost from the system.

In systems with short residence times like Western Port, where the volume of water entering the Western Arm each day from Bass Strait is almost equivalent to the total volume of the bay (0.8 km3), microbial processes in the sediments are generally considered to be less important in alleviating elevated nutrient inputs, because excess nitrogen in the water column is rapidly diluted through exchange with nutrient-poor coastal waters (Nixon *et al.* 1996). However, water residence times vary markedly between basins in Western Port, ranging from days in the south-west to months in the

north and east (Counihan *et al.* 2003). Because the basins in the north and east are also subject to the greatest sediment and nutrient loads, nutrient processing in the sediments is likely to be particularly important in shaping the ecosystem response. For example, the chance of an algal bloom in the water column will be reduced if excess nitrogen is removed by efficient denitrification in the sediments. There is also growing evidence that tidal flats can be very important in regulating nutrient availability in shallow systems, despite short residence times, because of the greater contact between nitrogen in the water column and the sediments (e.g. Eyre *et al.* 2011). Our lack of understanding of the role that the vast tidal flats, both vegetated and unvegetated, play in the bay's nutrient dynamics is a major knowledge gap that needs to be addressed if we are to understand the consequences of shifts in the balance of primary producer habitats.

While reasonably extensive data exist on water column nutrient concentrations in Western Port, there has been only one study of nutrient cycling in sediments (Longmore *et al.* 1998). Longmore reported that approximately 33–55% of the remineralised nitrogen in the sediments at the three sites studied in the summer of 1998 was lost as gas. Although this is within the range of denitrification efficiencies reported for other Australian bays and estuaries. However, these authors noted that there was uncertainty in this figure because of the uncertainty in the composition of the organic matter being broken down, and they highlighted the need for a better understanding of the spatial and seasonal effects of benthic nutrients fluxes.

Historically, much of the understanding of the cycling of nitrogen and phosphorus comes from relatively deep temperate coastal systems such as Port Phillip Bay (Longmore and Nicholson 2008) and Chesapeake Bay in eastern USA (Kemp *et al.* 2005). Less is known about how these nutrients are cycled in shallower systems where extensive tidal flats support a mosaic of seagrass, macroalgal and unvegetated mudflats. This is particularly important because tidal flats are often the major zones of organic matter accumulation, on account of their sheltered nature.

Seagrass habitats

Measured rates of denitrification in temperate seagrass habitats are typically very low, because coupled nitrification–denitrification in the rhizome of temperate seagrasses is typically suppressed by competition for nitrogen resources between nitrifying/denitrifying bacteria and seagrass and benthic microalgae (e.g. Risgaard-Petersen *et al.* 1998, Welsh 2000). Because of the vast area covered by seagrass in Western Port and the limited recovery of seagrass in the Corinella segment and sections of the Upper North Arm, the lack of understanding the functional contribution of seagrass to nitrogen cycling is a significant knowledge gap that requires attention.

Unvegetated benthic habitats

Mudflats may act as both sources and sinks of nitrogen, with benthic microalgae exerting an important influence on nitrogen processes in these environments (Sunback *et al.* 1991, Underwood and Kromkamp 1999, Cook *et al.* 2009, Joye *et al.* 2009). There is no information available on the biomass or productivity of benthic microalgae in Western Port. Bulthuis *et al.* (1984) compared concentrations of nutrients in waters ebbing from unvegetated mudflats with that from areas covered by seagrass in Western Port, and reported higher suspended solids, phosphate, and silicate fluxes coming off unvegetated mudflats but no difference in ammonia or oxidised nitrogen. A more comprehensive assessment, comparing both flood (inputs) and ebb (outputs) is required to assess net fluxes of nutrients and sediments over vegetated and unvegetated mudflats. This should be accompanied by detailed process measurements in order to understand the mechanisms underpinning nutrient and sediment transformation across the tidal flats. Work elsewhere has demonstrated that key transformation processes such as denitrification are influenced by the presence of benthic algae, benthic fauna and the elevation of the tidal flat (Joye *et al.* 2009). Because these factors may be linked to manageable factors such as the removal of important benthic fauna for bait (see below), understanding the mechanisms can greatly assist management.

The study by Cook *et al.* (2004) on nitrogen cycling on intertidal mudflats in the Huon estuary, Tasmania, found very low rates of denitrification, most likely because of the assimilation of nitrogen by benthic microalgae in competition with nitrifying/denitrifying bacteria for ammonia and nitrate. Their estimates of nitrogen assimilation by benthic microalgae indicated that nitrogen fixation was likely to be a major source of nitrogen during summer. The tight recycling and storage of nitrogen in benthic microalgae may support a significant source of secondary production. This would help explain why Edgar *et al.* (1994) reported similar rates of secondary production by macrofauna living in the sediments of unvegetated mudflats and seagrass beds in Western Port. Assessing the biomass and productivity of benthic microalgae, the influence on nitrogen cycling and the implications for secondary production should be a high priority because of the vast area of unvegetated mudflats in Western Port.

Role of macrofauna

Macrofauna modify sediment biogeochemistry via their feeding, excretion, bioturbation, burrow construction, and burrow irrigation activities (Pelegrí *et al.* 1994, Pelegrí and Blackburn 1995, Webb and Eyre 2004, Dunn *et al.* 2009). Rates of denitrification are generally stimulated by the presence of benthic macrofauna (Kristensen and Blackburn 1987, Pelegrí *et al.* 1994, Pelegrí and Blackburn 1995, Tuominen *et al.* 1999, Bartoli *et al.* 2000, De Roach *et al.* 2002, Webb and Eyre 2004, Nizzoli *et al.* 2007).

Callianassids (ghost shrimps) frequently dominate the sediments of coastal environments, where their role in regulating benthic nutrient fluxes is well known (e.g. Forster and Graf 1995). Their burrowing and irrigating activities stimulate fluxes of solutes and gas through the sediments, enhancing bacterial growth and increased rates of nitrification and denitrification in the sediments (e.g. Huttel 1990). Three species of callianassid shrimp are widely distributed in Western Port: *Biffarius arenosus* and *Trypea australiensis* mainly inhabit intertidal and shallow subtidal

sediments (< 10 m), whereas *Neocallichirus limosus* tends to inhabit deeper (> 10 m) sediments (Boon *et al.* 1997). In Port Phillip Bay, Bird *et al.* (1999) demonstrated the influence of callianassids on the exchange of solutes across the sediment–water interface. In northern New South Wales, Webb and Eyre (2004) demonstrated that *Trypea australiensis* enhanced pore water exchange and rates of denitrification, but concentrations of benthic microalgae were 50% lower compared to sediments unoccupied by shrimp. Although similar experiments have not been conducted in Western Port, it is reasonable to expect similar effects where callianassids reach similar densities. Importantly, Contessa and Bird (2004) reported a significant effect of bait pumping on the abundance and burrow structure of callianassids on a tidal flat in Western Port. Although the effects are likely to be very localised, efforts to minimise the disturbance of mudflats may be warranted.

The Soldier Crab *Mictyris platycheles* is present in Western Port in very large numbers on intertidal mudflats at low tide. Webb and Eyre (2004) found that its congener, *Mictyris longicarpus*, significantly reduced benthic microalgae production and sediment oxygen demand.

Because a number of other macrofauna could significantly influence benthic fluxes of nutrients in Western Port, future studies involving process measurements should also measure the composition and biomass of macrofauna and identify marine pests.

Primary production

Detailed descriptions of the major primary producer habitats are provided in Chapters 5 and 7–10. The focus here is to summarise our current understanding of the process of primary production . With the exception of seagrass, there have been very few studies that have quantified rates of primary production in Western Port. While it is likely that seagrass contributes the most to the primary production that underpins the Western Port ecosystem, the role of other producers, particularly macroalgae and benthic microalgae on the unvegetated tidal flats, is likely to be significant. In Port Phillip Bay, which is much deeper, the contribution of benthic microalgae to production is thought to be equivalent to half that of phytoplankton in the water column (Murray and Parslow 1999).

Seagrass

There has been extensive work on the growth rates of seagrasses in Western Port, including the importance of environmental factors (e.g. Clough and Attiwill 1980; Bulthuis 1981, 1983, Campbell and Miller 2002; Campbell *et al.* 2003; Miller *et al.* 2005: see Chapter 10). This work has included a variety of techniques for measuring productivity (from traditional growth measurements to pulse amplitude modulation fluorometry) and the influence of environmental factors (manipulative experiments and gradient studies).

EPA undertook repeated measures of *Zostera tasmanica* condition and water quality at three sites of varying water quality in Western Port from 1998-1999 (Campbell and Miller 2002; Campbell *et al.* 2003, Miller *et al.* 2005). Changes in shoot characteristics shoot density, above-ground biomass and photosynthetic characteristics were associated with higher concentrations of suspended matter and lower light fluxes at the more northern sites (Charing Cross and Crib Point), suggesting a strong influence of light availability on *Z. tasmanica* growth. This is consistent with the findings of earlier studies in Western Port (Clough and Attiwill 1980; Bulthuis 1981, 1983).

Campbell and Miller (2001) reported significantly higher leaf tissue nitrogen content and nitrogen : phosphorus ratios at the northern sites compared with Newhaven in the south. Epiphyte nitrogen content was similar at all sites, but nitrogen : phosphorus ratios were higher at Charing Cross compared with Newhaven. The nutrient content of seagrass and algal tissue is recognised as both an integrator of nutritional history and a bioindicator of water quality (Duarte 1990, Schaffelke and Klumpp 1998, Udy *et al.* 1999). The relatively low tissue nitrogen content at Newhaven (1.17 ± 0.14%) is consistent with the value reported previously in Western Port (0.7–1.1%) by Bulthuis and Woelkerling (1981). According to Duarte (1990) the growth of seagrass leaves is likely to be limited by nitrogen when tissue nitrogen concentration is less than 1.8%. Nitrogen limitation is therefore likely at Newhaven, a finding consistent with Bulthuis and Woelkerling (1981) who showed that nitrogen limited the growth of *Z. tasmanica* during its active spring and summer growth phase because of nitrogen depletion in the sediments surrounding the roots. In contrast, the tissue nitrogen content of *Z. tasmanica* at Charing Cross (2.06%) and Crib Point (2.21%) in 2001 was higher than previously reported in Western Port (Bulthuis and Woelkerling 1981), suggesting that nitrogen availability at these sites in Western Port had increased over the previous 20 years and seagrasses found here are unlikely to be nitrogen-limited.

It is generally agreed that increasing nutrient loads to seagrass habitats can increase the growth of more opportunistic phytoplankton, macroalgal or epiphytic species, causing shading and a decline in seagrass health. Morris *et al.* (2007) carried out nutrient addition experiments at three sites in Western Port, Victoria using a commercial NPK fertiliser, and reported an increase in ash-free dry weight of seagrass leaves and loose algae at two of the three sites studied. They concluded that Western Port seagrass habitat is sensitive to increased loads of nutrients within the water column, particularly in the northern section.

Benthic microalgae

There have been no published estimates of the biomass or productivity of benthic microalgae. To put this in context, Murray and Parslow (1999) reported that benthic microalgae in Port Phillip Bay make a substantial contribution, amounting to about half the phytoplankton production. As this is in a comparatively deep bay, the role of benthic microalgae is likely to be significant in Western Port because of the large area represented by shallow unvegetated sediments.

Macroalgae

Marine macroalgae (mainly *Asperococcus, Caulerpa, Enteromorpha, Gracilaria* and *Polysiphonia* species) are common on sediment habitats in Western Port. Bulthuis (1981) reported that they may account for 27% of the total macrobenthic standing crop, yet very little is known about the role of macroalgae in the Western Port ecosystem. For example, does the turnover of macroalgae represent a major source of nutrients and/or detritus? Has macroalgae filled in gaps created by seagrass loss? Does it smother and displace seagrass? Understanding the nutrient and light requirements of macroalgae and the other major primary producers would help address such questions.

Phytoplankton

The only studies that have assessed phytoplankton productivity were carried out in the 1970s (Bulthuis 1976; 1977; Robinson and Harris 1974). Bulthuis (1977) compared productivity at a variety of depths in the Western Entrance, North Arm and East Arm, with the highest rates recorded in the Western Entrance, possibly associated with good water clarity and ocean flushing. Bulthuis (1976, 1977) considered that phytoplankton productivity was limited by both nitrogen and phosphorus, but this conflicted with the results of Robinson and Harris (1974). A comparison of historic ambient nutrient concentrations to those required for optimal growth for diatoms indicated that Western Port waters were probably nitrogen-limited and possibly also phosphorus-limited (Longmore 1997).

Secondary production and higher trophic levels

The importance of intertidal seagrass as a site of elevated invertebrate species richness secondary production (particularly by epifauna), compared with intertidal unvegetated habitats in Western Port, was demonstrated by Edgar *et al*. (1994). Flow-on effects to higher trophic levels, including callianassid and caridean shrimps, fish and birds have been extensively reported in Western Port (e.g. Howard 1984; Howard and Lowe 1984; Watson *et al*. 1984; Boon *et al*. 1997; Edgar *et al*. 1995a,b; Longmore *et al*. 2002: see Chapters 11 and 12 for more detail). Although Edgar *et al*. (1994) reported lower rates of secondary production for epifauna in unvegetated mudflats compared to seagrass habitats, rates of secondary production of infauna were similar in the two habitats. Primary production of microalgae living on the surface of unvegetated mudflats is likely to be a key food source fuelling this secondary production. (Secondary production in the water column is covered in Chapter 5.)

Major threats

Sediment and water quality

Nutrient inputs

Human activities such as land clearing, the production and application of fertilisers, discharge of human wastes, animal production, and combustion of fossil fuels have mobilised nitrogen and phosphorus. Nutrient inputs to coastal waters often stimulate primary production because the major primary producers — phytoplankton, microphytobenthos, macroalgae and macrophytes — are often limited in their rates of production and growth by the availability of nutrients, in particular nitrogen and phosphorus. However, excessive nutrient availability and plant growth is now a serious environmental problem in many coastal ecosystems because it disrupts the balance between the production and metabolism of organic matter in the coastal zone. Most notably it favours faster-growing plants, particularly phytoplankton and epiphytic macroalgae. This can result in a number of negative environmental consequences. The decomposition of the phytoplankton derived organic matter can increase the depletion of oxygen from bottom waters, with flow-on effects for benthic macrofauna and oxygen-dependent nutrient processing (e.g. coupled nitrification–denitrification). Increased phytoplankton biomass and the growth of epiphytic algae decreases light availability for benthic producers such as seagrass. See Cloern (2001) for a comprehensive review of the coastal eutrophication problem.

Risks

The major system-specific attribute that modulates the response to nutrient inputs is the set of hydrodynamic properties that determine the residence time of the nutrients and plankton in the receiving waters. In Western Port, the Corinella segment is particularly vulnerable to algal blooms because of the reduced tidal currents and flushing (i.e. long residence time) and the net clockwise direction of water flow within the bay, which delivers nutrients from the Upper North Arm where catchment inputs are greatest. This is evident from ongoing monitoring, with total suspended solids, nutrient and chlorophyll concentrations in the water column of the east arm (Corinella segment) regularly exceed SEPP levels and ANZECC guidelines for Western Port (EPA 2008).

Greater tidal flushing and lower nutrient concentrations in the remainder of the bay suggest that these areas are less likely to be affected. However, the inputs of nutrients in regions with large tidal flats that allow greater contact between sediment and water may lead to more subtle changes in the composition of primary producers. High nutrient inputs from some creeks could have major impacts on local primary production and nutrient cycling. A notable example is Watsons Creek, which discharges very high nutrient concentrations into Yaringa Marine National Park. There also remains some uncertainty surrounding the quantity of nutrients that enter the Western Arm from the Boags Rocks outfall on the outer coast. Although these inputs are likely to be diluted by strong tidal currents, a greater understanding of the magnitude of these inputs is needed.

In general, nutrient concentrations are considered to be low in Western Port, but this is not uniform because of the combination of clockwise circulation, location of major sources and longer residence times rendering the Upper North Arm and Corinella segments susceptible to adverse impacts. For the same reasons the effects of sediment inputs, and the synergistic effects between sediment and nutrients, are most likely in this region. As discussed elsewhere in this review (Chapters 9, 10, 15), there is a need to better characterise nutrient dynamics and the consequences for the major primary producers, particularly in the Upper North Arm and Corinella segments.

Consequences

The loss of habitat-forming species such as seagrass has major effects on the diversity and abundance of associated fauna. The work of Edgar *et al.* (1994) in Western Port highlighted the importance of intertidal seagrass as a habitat with elevated species richness of invertebrate fauna and secondary production (by epifauna) compared with intertidal unvegetated habitats. Flow-on effects to higher trophic levels, including fish and birds have also been reported in Western Port (e.g. Edgar *et al.* 1995a,b; Longmore *et al.* 2002). The decline and limited recovery of seagrass in the eastern section of the bay (EPA 2008) may be symptomatic of nutrient and sediment loads exceeding the system's capacity to process and assimilate them. Further work is required to separate natural fluctuations from the role of nutrient and sediment inputs in the decline (including thresholds) and potential recovery of seagrass. This is essential for the efficient prioritisation of catchment mitigation strategies.

In other areas of Western Port, nutrient inputs from creeks and drains can also have adverse affects on local biodiversity and food webs. The proliferation of ephemeral algae can lead to the smothering of seagrass and microphytobenthos, particularly on intertidal mudflats. The concomitant shift in habitat structure and organic matter produced is likely to lead to flow-on effects for associated biodiversity and food web structure.

Sediment inputs

Risks

Catchment sediment loads to Western Port have been extremely high because of the development of the surrounding area, including the draining of Koo Wee Rup swamp. Coast sediment loads have increased over the last 40 years (Hancock 2001). In addition to catchment sources, shoreline erosion makes a significant contribution to Western Port's sediment loads. The delivery of sediment loads is accompanied by the resuspension of sediments, predominately from sediments on the bay floor (Wallbrink *et al.* 2003); see separate threat description below.

Sediments entering the bay directly affect the transparency of the water column, and thus the light available for photosynthesis, while they remain in suspension. In areas of reduced water movement, suspended sediments can settle out and completely smother benthic producers. Sediment inputs may also alter the grain size composition of sediments, a key determinant of faunal composition (Gray and Elliot 2009). Sediment inputs also transport nutrients (particularly phosphorus), metals and other contaminants.

Because of the net clockwise direction of water flow within the bay, much of the sediment delivered into the north-eastern part of the bay is transported into the Corinella and Rhyll segments, where much of it is deposited (Hancock *et al.* 2001). Total suspended solid concentrations in the water column of the East Arm (Corinella segment) regularly exceed SEPP levels and ANZECC guidelines.

Other areas of high suspended sediment concentrations and deposition include the mouths of the rivers, creeks and drains from which the sediment is discharged. This is largely a result of the rapid change in water velocity, although the change in salinity also leads to elevated flocculation (or 'turbidity maximum') and deposition of particulate organic matter.

Consequences

High suspended sediment concentrations can lead to a decrease in the rate of primary production in the water column and on the bottom. This is also likely to have consequences for the composition of producers, as they differ in their light requirements. A sustained deposition of sediments can lead to the death of benthic producers by smothering, and a shift in sediment composition can result in a shift in the composition of fauna. Because of the importance of primary production as a source of organic matter and oxygen for the bacterial processes associated with nutrient transformation, shifts in the rates and role of processes such as nitrification and denitrification are also likely to change.

Tidal mudflats flat are naturally sites of elevated sediment deposition, however the effects of elevated loads from anthropogenic sources remain unclear. The decline and limited recovery of seagrass in the eastern section of the bay, where sediment loads are known to accumulate, is consistent with the effects of elevated loads from anthropogenic sources exceeding the system's capacity to process and assimilate them.

The hydrodynamic environment of the Western Shoreline and the Eastern Entrance prevents fine sediments from accumulating, and suspended sediments are not considered a serious risk in these segments.

Sediment resuspension

Turbulent tidal currents and wind-induced wave action play a key role in sediment resuspension. On tidal mudflats, fine sediments settle out during calm weather, but the water column (over both the flats and associated channels) becomes highly turbid when wind-induced wave action disturbs bottom sediments. Benthic microalgae are also resuspended with the fine sediment (mud), often constituting a major fraction of water column phytoplankton biomass (e.g. de Jonge and van Beusekom 1995). In systems in which phytoplankton production in the water column is limited, resuspended microalgae may provide an important food source for zooplankton grazers and suspension feeders on tidal flats.

Where inputs of fine sediments are elevated over natural levels as a result of human activity, the resuspension of this sediment can lead to extended periods of reduced light available for benthic production. This has major implications for rooted macrophytes such as seagrass. The resuspension of bottom sediments may also have a significant effect on the flux of nutrients via the advective transport of sediment pore water (rich in nitrogen, phosphorus and carbon) into the water column. This includes catchment-derived nutrients delivered to the sediments as particulate organic matter or adsorbed to inorganic particles.

Risks

Figure 14.6 shows where mud is likely to settle out as a result of reduced tidal current velocity, but because of the shallow nature of these areas and the long fetch, sediments are also likely to be suspended by wind-induced wave action. This is evident at the monitoring site in the Eastern Arm, where total suspended solid concentrations in the water column regularly exceed SEPP levels and ANZECC guidelines. Changes in wind and wave patterns caused by climate change may also produce a significant change the resuspension dynamics in Western Port (see Chapter 4).

Consequences

As discussed above, resuspended sediment leads to a decrease in light available for benthic primary producers, and can also lead to smothering during settlement. The resuspension of benthic microalgae may also lead to elevated chlorophyll concentrations in the water column. Change in the cover of important sediment stabilisers such as seagrass will also lead to significant feedbacks; for example, increased resuspension following the loss of seagrass may limit the recovery potential because of reduced light levels and smothering. Resuspension of bottom sediments may also lead to elevated fluxes of nutrients and elevated rates of phytoplankton production in the water column.

Pests

Although biological invasions in the marine environment are a major global environmental problem, only a small percentage of invaders are likely to cause ecological change. In Port Phillip Bay, Ross and Keough (2006) reported significant effects of introduced macrofauna on nutrient fluxes in Port Phillip Bay, most notably a drop in denitrification in the presence of the European fanworm *Sabella spallanzanii*. Drops in denitrification efficiency will ultimately feed back to changes in water column productivity because of a greater internal recycling of nutrients between the sediment and water column.

Non-native species have been reported in Western Port, predominantly around Hastings and associated ports where pest surveys have been conducted. Of these species, most that have been recorded are 'fouling' species (Currie and Crookes 1997, Parry and Cohen 2001) and are often associated with built structures (Keough and Ross 1999). A number of these fouling species also establish on soft sediments, but there have not been any comprehensive surveys of native or pest flora and fauna of soft sediments in Western Port for over 30 years. In the absence of useful data, our discussion here focuses on pest species we believe are more likely to affect ecosystem processes such as nutrient cycling and primary production. This includes pest species that have already been identified in Western Port, as well as several other pests of concern that might occur in Western Port. Although not strictly an introduced species, we also include *Noctiluca scintillans*, a large toxic dinoflagellate that has extended it range south with the greater influence of the East Australian Current and is now blooming more regularly along the coast of south-eastern Australia, including major influxes into Port Phillip Bay since 1993 (Hallegraeff *et al.* 2009).

Risks and consequences

Sabella spallanzanii is one of the higher-risk pest species for Western Port, with the potential to arrive from Port Phillip Bay via shipping or on currents. However, it is unclear whether it would attain the high densities on subtidal sediments in Western Port that are required to exert significant change on ecological processes, because it tends to occur where bottom stress from currents is generally low, which is not the case for much of Western Port.

The brown alga *Undaria* also has the potential to spread from Port Phillip Bay via shipping or on currents. An infestation at Flinders was removed at an early stage of infestation, thus preventing further spread (Parry and Cohen 2001). However, like *Sabella* it is not likely to proliferate on the vast tidal flats of Western Port. Thus it seems unlikely that either of these pests will lead to significant changes in nutrient cycling or primary production, aside from possible local effects on the composition of flora and fauna.

The European shore crab *Carcinus maenas* is already well established in Western Port, but there is no information on its impacts beyond its effect on native crabs on rocky reefs (Sinclair 1997). In Tasmania *Carcinus maenas* is a major predator of bivalve and gastropod molluscs in intertidal and subtidal soft sediment habitats (Walton *et al.* 2002, Ross *et al.* 2004). The indirect effects of predation on nutrient cycling and primary production have not been examined.

Another pest species that poses a risk to ecosystem processes is the aquarium strain of *Caulerpa taxifola* that has been present in New South Wales since 2000. The ability of *Caulerpa* to grow rapidly and potentially out-compete native seagrass is a major concern. Research results to date from NSW indicate that dense native seagrass beds are relatively resistant to invasion from *Caulerpa*, however, sparse seagrass beds (*Zostera* in particular) may be at risk from *Caulerpa*. Differences in the composition of fish and invertebrates have also been reported between native seagrass and *Caulerpa* beds.

In the water column, the range expansion and increasing dominance of the warm-water dinoflagellate *Noctiluca scintillans* in southern Australia presents a significant risk to food web dynamics and nutrient cycling in the water column of Western Port. *Noctiluca* is a major grazer of phytoplankton and contains high concentrations of ammonium, so that Noctiluca blooms (red tides) produce potentially toxic ammonium concentrations in surface waters from excretions (Montani *et al.* 1998). When present in such blooms, *Noctiluca* may act as one of the most important organisms contributing to nitrogen cycling in Western Port. On the eastern coast it has been observed to outcompete local zooplankton when in bloom proportions (Dela-Cruz *et al.* 2002, 2003).

Climate change

Climate change poses a significant long-term threat to ecosystem processes. Sea-level rise will result in a change in bathymetry. Intertidal zones will move shoreward, which may lead to a decline in the area of tidal flats. As the average depth of water over tidal flats increases, light available for photosynthesis will decline. Differences in the light requirements of primary producers will lead to shifts in composition and changes in other ecosystem services. However, the consequences of sea-level rise are likely to be far more complex. For example, as light available for photosynthesis decreases, heat and desiccation stress may increase. Changes in the water depth will depend on a complex interplay between the sedimentary processes of accretion and erosion. The presence of existing barriers such as roads and potential adaptation measures such as the construction of seawalls will also influence these sediment processes and the ability of tidal flats and associated habitats to migrate inland. The need for hydrodynamic and sediment models that can resolve these processes has been highlighted in Chapter 4; this will underpin our ability to predict the likely consequences of sea level rise for the composition and biomass of primary producer habitats and rates of production.

Sea surface temperature (SST) off the coast of Victoria under medium emissions is predicted to increase by 0.3 to 1°C by 2030, rising to 0.6 to 2°C by 2070 (CSIRO and BoM 2007). Most of the ecosystem processes discussed are strongly influenced by temperature. Therefore, increased temperature will lead to changes in the rates of processes such as respiration, production, and denitrification, but the implications for the ecosystem are difficult to predict. Further, the effects of changes in temperature will not act in isolation from other climate driven changes.

Another possible consequence is a shift in the dominance of existing species or the establishment of species not found in Western Port due to increasing temperatures and the strengthening of the East Australian current (CSIRO and BoM 2007). As discussed above, species such as *Noctiluca* have the potential to have major consequences for food web dynamics and nutrient cycling water column; increases in their frequency and abundance have already been observed on the (lower) east coast of Australia (Dela-Cruz *et al.*, 2002, 2003).

Palacios and Zimmerman (2007) reported increased productivity by *Zostera marina* under elevated concentrations of dissolved CO_2 in short-term and long-term mesocosm experiments, and contended that the area of *Z. marina* coverage would increase by 35% if atmospheric CO_2 concentrations doubled, based on current water quality and bathymetry. For a review of the potential effects of increased CO_2 on microalgal growth, see Beardall and Raven (2004).

It is clear that climate change will have significant consequences for ecosystem processes, but the overall consequences will probably reflect very complex interactions between the various elements of climate change and other environmental variables such as nutrient and light availability.

Research that can fill key knowledge gaps

The major threats to ecosystem processes in Western Port, particularly nutrient cycling and primary production, are changes in sediment and nutrient inputs resulting from changes in freshwater flows, nutrient availability and the light climate. Therefore, identifying the key sources of nutrients and sediments and their pathways is vital for prioritising mitigation actions. The collection of callianassid shrimps from tidal flats for bait also presents a threat to nutrient transformation processes. Although the effects are likely to be local, the spatial extent should be assessed. Climate change also poses a significant long-term threat to ecosystem processes.

The efficient allocation of resources for management depends on a sound understanding of the linkages between mitigation actions and ecosystem health. In Western Port the decline and limited recovery of seagrass in the eastern section of the bay (EPA 2008) is symptomatic of nutrient and sediment loads exceeding the system's capacity to process and assimilate them. Our understanding of the ecological thresholds of the major habitat-forming primary producers such as seagrasses is limited, yet this information is essential when setting nutrient and sediment reduction targets and allocating resources to on ground works. Furthermore, the consequences for nutrient and sediment dynamics if there is a shift to unvegetated mudflats is poorly understood. Benthic microalgae, which colonise the sediment surface, are major primary producers in many coastal estuaries and bays, providing an important food source, assimilating nutrients and stabilising sediments, yet the biomass of benthic microalgae and functional role of unvegetated tidal flats has not been assessed in Western Port.

More broadly, our understanding of the functional roles (primary production, respiration, nitrogen fixation, denitrification, and secondary production) of the major habitats (e.g. saltmarsh, mangroves, seagrass meadows, unvegetated sediments and open water) and the thresholds of change is a significant knowledge gap. This includes the interdependency and connectivity of these habitats for the flow of energy, nutrients and organisms.

A systems-level understanding of Western Port will not only assist with immediate management decisions, including efforts to mitigate the effects of changes in nutrient and sediment inputs, but also greatly improve the capacity to predict and manage for future changes to the Western Port ecosystem.

Research that can fill key gaps

A multistage research program is proposed that would develop a nutrient and sediment budget for Western Port, identifying key areas and habitats for the transformation and removal of nutrients and the settlement and resuspension of sediments. The recommended stages would include a rapid assessment that would determine the need for detailed formal assessments of nutrient cycling and the need for a formal process-based model for Western Port. Such a model, coupled with improved sediment and hydrodynamic models, would allow a detailed exploration of the benefits that would be expected from alterations to catchment inputs of nutrients and sediments, but also the capability to predict the response of the Western Port ecosystem to future climate scenarios.

Budget model

A budget for nitrogen and phosphorus at the scale of the five segments of Western Port would identify the key sources and sinks at scales relevant to management. This would complement with the current SEPP Schedule (F8) which recognises only two water quality segments in Western Port. The major building blocks of this exercise already exist — the PortsE2/WaterCAST model to parameterise catchment sources, and the 3DD hydrodynamic model to estimate exchanges between segments. Importantly, the budget should include the breakdown of total nitrogen and phosphorus catchment inputs into bioavailable versus refractory (unavailable) fractions. There is a marked variability in the bioavailability of different forms of nitrogen and phosphorus, and thus in the ecological response to changes. Most load monitoring sites are a reasonable distance upstream of the entry point to the bay, so that transformation in the intermediate lower river – upper estuarine reaches should also be taken into account.

Process measurements

Nutrient cycling

The budget would help identify the important areas of nutrient transformation. Detailed process measurements would identify and quantify the ecosystem processes (e.g. nitrification, denitrification, nitrogen fixation) responsible and the role of the different benthic habitats. This includes the measurement of production, respiration and net ecosystem metabolism, and thus the importance of each habitat as a source or sink of organic matter for higher trophic levels. Assessing differences in the key nutrient processes and feedbacks across the major habitats in Western Port would help identify the consequences of habitat changes (as previously observed). The knowledge would provide the mechanistic understanding that would help constrain the budget and process model.

Drivers of primary production

The composition and biomass of the main primary producers in Western Port have major implications for associated food webs. Understanding the degree of nutrient and light limitations would significantly enhance our ability to predict the consequences of changes in nutrient and sediment inputs, given their direct and indirect effects on nutrient availability and light levels. The current understanding of phytoplankton composition and behaviour in Western Port is poor. Because of the range expansions already observed for a variety of planktonic organisms, it is important to define existing plankton assemblages and subsequent shifts, in concert with the other process measurements and modelling proposed.

Water quality targets for sediments and nutrients

Interactive effects of nutrients and sedimentation on the major primary producers, including feedbacks via sediment stabilisation and nutrient transformation, are likely to be common in Western Port. Seagrasses and mangroves are the highest priority for research because of their importance in sediment stabilisation and nutrient cycling. Understanding the interactive effects and feedbacks would help to prioritise management actions to refine water quality targets and reduce loads.

Habitat connectivity: the role of seagrass detritus

Seagrass beds produce a significant biomass of detritus (both seagrass and associated epiphytes) that can be deposited locally in the bed or transported to other habitats. The consequences for detritus-based food webs and nutrient processing will depend on transport processes and how labile the detritus is. This could have major implications for lateral energy flow between habitats, energy transfer to higher trophic levels, and nutrient availability.

Causes of elevated chlorophyll in the Corinella segment

It is important to determine the causes underpinning the elevated levels of chlorophyll and other water quality parameters in the Corinella segment; for example, are they related to excess nutrients and uptake in the water column, or do they reflect resuspension of benthic microalgae? Assessing the species composition of phytoplankton, including temporal and spatial patterns, will help determine whether algae in the water column are planktonic or benthic, or both. If they are benthic it will also be important to determine whether the biomass of benthic microalgae is elevated due to nutrient inputs.

Process-based biogeochemical model

There are two major building blocks — the PortsE2/WaterCAST catchment model and the 3DD receiving water model, including a primary production module. The expansion to include other key biogeochemical processes (e.g. denitrification, fixation) and improved hydrodynamic and sediment modules (as outlined as priorities in Chapter 4) will provide further insights into the current system state and function and help to predict the response of Western Port to management actions and future climate scenarios.

15 Consolidated research needs and prioritisation

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The research needs identified in each chapter, which related to specific assets of Western Port, have been combined and then screened against three criteria: management benefit, immediacy, and likelihood of a successful outcome. This resulted in an integrated list of research needs, which were then assigned to three priority categories. The highest priority research needs are those that are achievable and would make an immediate difference to management of the Western Port ecosystem. Some other research is essential but is not immediately needed (e.g. quantifying responses to some aspects of climate change), some that might not feed directly to management actions but would lead to more informed decision-making (e.g. determining how much Western Port has changed since its previous assessment 36 years ago, to help set more realistic goals for environmental condition), and research needs where several additional steps would be needed to realise management benefits. We have classified the most urgent research needs into five categories, as follows.

Physical processes

Sediments, nutrients and other contaminants are important threats to the bay and its assets, and understanding how this material enters Western Port and how it moves about the bay requires a sophisticated suite of models that can describe the complex patterns of water movement around Western Port. We have a partial understanding of these processes, but we need to:

1. *Obtain detailed and up-to-date bathymetry for Western Port.*
2. *Calibrate hydrodynamic models to ensure accurate representation of water movement.*

Nutrients and sediments

The continued health of Western Port depends on important groups of plants – habitat forming species such as seagrasses and mangroves and algae that are responsible for important ecosystem services, particularly nutrient cycling, and we need to know relationships between these species and water quality, particularly sediments and nutrients There is uncertainty about the extent to which nutrients may be an issue in northern parts of Western Port, and uncertainty about the relative importance of different sources of suspended sediments. We need to:

3. *Determine a preliminary nitrogen and phosphorus budget.*
4. *Measure nutrient cycling in major habitats (unvegetated soft sediments and seagrass habitat).*
5. *Assess the degree of nutrient and light limitation of the major primary producers, seagrass and possibly microphytobenthos.*
6. *Determine water quality targets for sediments and nutrients that support seagrasses (and possibly microphytobenthos).*

Seagrasses

Seagrasses are among the most important habitat-forming species in Western Port, but to understand the loss and recovery of seagrass habitat we need to:

7. *Determine which species of Zostera are present in Western Port.*
8. *Determine the capacity for Zostera to recover and colonise new areas.*

Toxicants

Our review found an important knowledge gap about the extent to which toxicants entering Western Port pose a threat to marine environments. We should:

9. *Make an initial estimate of the risk from toxicants beyond discharge points.*

Iconic species

The remaining research gaps relate to iconic species, specifically the fish that are responsible for much of the recreational value of Western Port and the shorebirds and waterbirds that use Western Port so extensively. We need to:

10. *Determine linkages between fish and habitats, to better understand the significance of habitat changes from seagrass to algae.*
11. *Determine the effects of recreational fishing on fish stocks.*
12. *Examine the trends in abundance of fish-eating birds in Western Port.*

The approximate cost and delivery time for these highest-priority research needs are provided. For the remaining research priorities we identify the approximate cost and delivery time, and the kind of management benefit for each of these recommendations. For every research need we also provided our assessment of whether the capacity currently exists in Victoria or southern Australia to deliver this research, and the potential funding sources and delivery modes, e.g. the likelihood of securing funding from sources other than state agencies, and the potential for research to be delivered through competitive grants, contracts or student research projects.

15 Consolidated research needs and prioritisation

The preceding nine chapters identified a range of research needs that are specific to particular assets or to our physical understanding of Western Port. The needs identified in each chapter may apply very specifically to individual assets (e.g. water quality thresholds for seagrasses), but the same research need might appear repeatedly and be amenable to consolidation into a larger research program. Similarly, a research need that applies to one asset may not be an urgent priority, but its importance increases if it appears multiple times.

A systems view of Western Port also means that there may be research needs that transcend individual assets, and relate to some of the interdependencies and feedbacks described in Chapters 4–6.

In this section we collate the individual research suggestions and prioritise them against several criteria:

- an explicit statement of how agencies responsible for managing Western Port will benefit from the research
- professional judgement of the likelihood of research generating clear recommendations or solutions within the next few years
- an assessment of whether the research would feed immediately into management, or if it is one step towards a clear management outcome; and if it is not a direct benefit, how many other steps are needed.

We also provide an informal assessment of whether the technical capacity exists in Victoria or southern Australia to conduct the research.

The results of this screening and prioritisation are presented in tabular form below, and consist of 43 individual recommendations, which can be grouped around several themes.

Recommendations were assigned one of three priorities, from highest to lowest 1, 2, and 3. The highest-priority items are knowledge gaps that are major impediments to scientific understanding and management, and for which gaining the information will enhance management immediately. High-priority items have been scoped roughly. Low and medium-priority items are those for which the management benefit is less immediate, or which have a lower chance of a successful outcome. In some cases a knowledge gap is more strategic, building on our understanding of the Western Port ecosystem and its threats. Some of these ideas would be ideal projects for research students.

Priority 1

Priority 1 recommendations are divided into three categories, as follows:

1A: Accurate hydrodynamic modelling for Western Port.

An accurate model would enable us to predict the movements of water, sediments, and nutrients. These recommendations require brief data collection, and would yield immediate results:

Obtain detailed and up-to-date bathymetry for Western Port.

This work is already underway and will immediately improve the accuracy of the hydrodynamic models.

Calibrate hydrodynamic models to provide an accurate representation of water movements.

This recommendation has already been commissioned. It is an essential step in the verification of the circulation models.

1B: The relationship between habitat forming plants and water quality.

We have identified several important vegetation types: seagrass, mangrove and saltmarsh. The most important relationship is the one between seagrasses and water quality, particularly because of the declines in seagrass in Western Port since 1975.

At present the major management actions for Western Port are focused around sediments, which have been recognised as an important issue for at least 30 years. During this review it became apparent that there are signs of concern about nutrients, particularly the elevated chlorophyll-*a* levels in the Corinella segment and some concerns about the amount of nutrients coming from Watsons Creek. Sediments have traditionally been considered a higher priority, so it will be important to clarify the situation regarding nutrients. We have identified a multistage investigation into nutrients, and we suggest that a relatively cheap initial investigation may be sufficient to determine whether nutrients are a problem. If they are a problem, a larger-scale investigation would be warranted:

Determine preliminary nitrogen and phosphorus budgets.

Measure nutrient cycling in unvegetated soft sediments and seagrass habitats.

Sediments are a major issue in Western Port, particularly in relation to seagrasses (Chapters 7, 10). Our review has identified several major research needs relating to sediments, which have important implications for how we manage the bay.

Estimate the contribution of coastal erosion to nutrient and sediment budgets.

A major gap in our knowledge is the relative size of catchment input, coastal erosion, and resuspension as sources of sediments; an important review 10 years ago summarised our knowledge to date, but acknowledged great uncertainty about the contribution of shoreline erosion. One of our important recommendations is to clarify these different sources. If, for example, catchment inputs are the major source of suspended material, then a management focus on the source is clearly appropriate. And if shoreline erosion is the major contributor, then rehabilitation should be emphasised. We also note that when planning for the future of Western Port, we need to consider how these sources might change in the future, particularly as a result of the various effects of climate change and urban development.

Sediments and nutrients are important aspects of water quality in Western Port, because they are such important influences on the ecosystem engineers. The most important of these engineers are the seagrasses, which have undergone considerable changes. We do not have a good idea of the water quality requirements of the common seagrasses in Victoria's bays (*Zostera* species), so we recommend a series of research projects to develop better water quality targets for them in Western Port. As a first step, we need to know whether these targets should be based on sediment levels, or on a combination of sediment and nutrient levels.

Assess the degree of nutrient and light limitation for the major primary producers, seagrasses and possibly microphytobenthos.

Determine water quality targets for sediments (and if necessary) nutrients that support seagrasses, and possibly microphytobenthos.

Before we can take these steps, we need to know which seagrasses are present in Western Port, because the taxonomy of *Zostera* has changed recently.

Determine which species of Zostera are present in Western Port.

Developing water quality thresholds for seagrasses is not a simple task, but there is important work underway that will inform a Western Port research project. The most important is a substantial Port Phillip Bay project, which is part of DSE's Reefs and Seagrass program, and there are similar initiatives underway in other eastern states.

The preceding work would define water quality conditions suitable for seagrasses, but this is no guarantee that seagrasses will be present even if those conditions are met. The other part of this equation is the processes of colonisation and growth of seagrasses.

Determine the capacity of Zostera to recover and colonise new areas.

The final aspect of water quality is the toxicants arriving through catchment discharges. They are diverse and vary between the major sources around Western Port, and their effects are unknown. It is possible that they do not disperse far from the estuaries, but they may be spread more widely. We recommend an initial investigation into their spread.

Make an initial estimate of risk from toxicants beyond discharge points.

1C: Other ecological knowledge gaps.

We have identified some important gaps in our understanding of the ecology of Western Port, and we recommend research to understand the relationship between habitat-forming marine plants and biodiversity, with fish as the highest priority.

Determine linkages between fish and habitats.

This research would tell us whether parts of Western Port's biodiversity rely on specific associations with, for example, seagrass, or whether different plants are ecologically interchangeable so that, for example, the replacement of seagrass by particular green algae might not result in a cascading ecological change.

The other ecological recommendation is for a detailed examination of the apparent decline in numbers of fish-eating birds in Western Port.

Examine the trend in the diversity and abundance of fish-eating birds in Western Port.

We suggest that such a decline would be cause for some concern.

Interpreting change and managing recreational fisheries sustainably require knowledge of the state of fish populations. We do not have this knowledge for Western Port, and we recommend that the gap be filled.

Investigate the state of fish populations in Western Port and the effects of recreational fishing on fish stocks.

Research themes for the full set of recommendations

The research priorities are presented around an ecosystem view of Western Port. We begin by highlighting the need to better understand how sediments, toxicants and nutrients are moved around the bay from the points at which they enter. We then consider the important ecosystem processes that move nutrients in and out of Western Port, whether those nutrients are a major threat, and the relationships of the major primary producers with nutrients, sediments and toxicants. Filling these research gaps provides us with a springboard for thinking about the condition of the major Western Port assets, and how bay and catchment management might improve it. Doing this requires an understanding of what Western Port is like today, rather than 36 years ago when the last major investigation (the Westernport Bay Environmental Study) was undertaken. There are considerable gaps in understanding how the ecosystem will respond to human changes, and filling these gaps depends on understanding some of the key ecological links, and also filling important knowledge gaps for several major threats. Beyond nutrients and sediments, we highlight predicted responses to climate change (particularly sea-level change) and the importance of toxicants.

Hydrodynamics and sediment dynamics

The research priorities identified here would provide a model that would better predict the water circulation in Western Port and its links to catchment flows and oceanic exchange. A well-developed and calibrated circulation model is central to our ability to understand the Western Port ecosystem. It also provides an essential tool for predicting the consequences of management actions that might alter catchment inputs or change the structures or bathymetry of Western Port, and in assessing the consequences of climate change.

We identify several steps to improving the Western Port circulation model, including some that are crucial and can be resolved quickly (improved bathymetry and current meter deployments to calibrate the model).

The movements of sediments are also tied to hydrodynamics (as are nutrients, toxicants and the spread of pest species), and the need for a fully developed sediment transport model for Western Port was highlighted in the initial Western Port study and reaffirmed in 2003. Despite the progress made in the previous decade, serious knowledge gaps remain. These gaps are more pressing today, as a fully developed sediment model will need to include the input of new sediments as a result of shoreline erosion (which was less severe in the 1970s). This sediment model will also be an important part of understanding the environmental conditions necessary for seagrasses to flourish, and in predicting the likely effects of catchment remediation on seagrass recovery. It will also be needed to predict the future water quality conditions in Western Port under changing climates.

Ecosystem processes

In coastal ecosystems, the flow of nutrients is almost always a major issue in urbanised areas. There are no major treated sewage discharges directly into Western Port, unlike Port Phillip Bay, but nevertheless there are several point and diffuse catchment sources, and EPA Victoria has recorded elevated chlorophyll-a levels in the Corinella segment of the bay. As discussed in Chapter 14, it is not clear what these chlorophyll levels represent, and, more importantly, whether nutrients are a serious issue for water quality and for aquatic flowering plants and algae. This knowledge gap is not new; the lack of a nitrogen budget has been highlighted in the past, and it remains a serious gap. The likely role of sediments, both directly (via the nutrients absorbed to sediment particles) and indirectly (via changes in the underwater light regime) in modifying the ecosystem response to nutrient inputs has also been identified as a key knowledge gap that warrants further research, but with little progress to date. We propose a multi-stage research program that would develop a nutrient budget for Western Port, identifying key areas and habitats for the transformation and removal of nutrients. The recommended stages will allow a rapid assessment, which will determine the need for detailed formal measures of nutrient cycling, and the need for a formal process-based model for Western Port. Such a model, coupled with improved sediment and hydrodynamic models, would allow detailed exploration of the benefits that would be expected from alterations to catchment inputs of nutrients and sediments.

Two other important ecosystem functions for Western Port are provided by the major primary producers: habitat structure and trophic transfer of carbon and energy. Habitat structure is provided by larger aquatic plants, but we know very little about the role in the food web of benthic microalgae, which are the dominant primary producers on mudflats where seagrass is absent. The consequences of shifts between seagrass and microalgae dominated habitats for energy transfer to higher trophic levels remain unclear. These habitats also influence nutrient cycling processes in very different ways.

An important research need, therefore, is to understand the light requirements of the important habitat-forming plants and benthic microalgae. When combined with an improved understanding of the relative contributions of sediments and nutrients, this would allow water quality targets to be refined, and would enable us to calculate the extent to which changes in catchment inputs would change the amount of seagrass habitat.

Sparse young *Avicennia marina* near Grantville.
Photo courtesy Adam Pope

How different is the Western Port ecosystem from when it was described in detail in 1975?

The Westernport Environmental Study provided a detailed snapshot of Western Port in the early 1970s. Since that time there have been dramatic changes in the bay as seagrass has fluctuated and turbidity has changed. The land use in the surrounding areas has also changed dramatically, and the expansion of Melbourne south-eastward has increased the recreational use of the bay. In reviewing our current understanding of Western Port, it became apparent for many of the bay's most valuable assets that we do not know whether Western Port of today is similar to conditions reported in the 1970s, or whether it has changed dramatically. This knowledge gap is not uniform; we have good records of numbers (and trends) of shorebirds and waterbirds over the entire period since 1975, and we have assessments of seagrass cover from a decade ago. In other areas there has been no examination of biodiversity or ecosystem structure and function for nearly 40 years.

This lack of information is important in several ways. It impedes a robust assets approach if the distribution of assets is poorly known. It limits our ability to set targets for a 'healthy' ecosystem when we do not know if there has already been irreversible change, although long-term bird monitoring does provide one such data set. In one of the striking cases, changes in taxonomic knowledge means that we do not know which species of *Zostera* seagrass are present in Western Port. With such a knowledge gap, we do not know if the ecological studies undertaken by Bulthuis many years ago are relevant to today's seagrasses. We also do not know if the information that will be gained from a large research program in Port Phillip Bay over the next few years can be extrapolated to Western Port, or if Western Port needs a detailed data collection program of its own.

We propose a series of studies aimed at updating our understanding of the 'state' of Western Port. Individual studies range from urgent and achievable to more strategic needs.

Deepening our understanding of ecological links

We have emphasised the critical links between sediments and nutrients and habitat forming plants, because our professional judgement is that the presence of these species affects a large number of ecosystem components. In some cases this is based on work on other shallow coastal embayments, but we are confident that the results can be applied to Western Port. In other cases differences between Western Port and other areas make us cautious, or we have no information at all.

This relationship is most important when thinking about resilience — the ability of a natural ecosystem to resist external environmental forces, and its ability to either recover or accommodate some change without losing essential ecosystem functions. Resilience is becoming the focus for ecologists and natural-resource managers in relation to climate change, with the suggestion that our goal is to produce resilient natural systems that are then likely to be more robust to changes in climate. While it is important, ecological resilience is poorly known for most individual species and habitats. We make several research recommendations about understanding the resilience of the Western Port ecosystem, or elements of it. These recommendations range from important ones, such as understanding the basis of seagrass resilience, to more strategic suggestions about resilience research that would be suitable for post-graduate students. In the case of seagrass, understanding the ways by which *Zostera* seagrasses recover from loss and colonise new areas is an important link between water quality targets, which determine suitable habitat for seagrasses, and the actual colonisation of these areas by seagrass. The view of *Zostera* described in Chapter 10 is of a 'weedy' species that at one time occupies only a portion of suitable habitat, with the exact areas occupied changing through time. Similarly, the resilience of mangroves and coastal saltmarsh to climate change depends strongly on whether they can maintain rates of sedimentation sufficient to keep up with the increase of sea levels: if they can, these vascular plant communities can probably survive in the face of higher sea levels; if they cannot, they will be drowned and lost.

Other recommendations cover research gaps about species of concern, such as identifying suitable habitat for listed species in Western Port.

Understanding particular threats

We have highlighted the need for a greater understanding of sediments and nutrients as important present threats to Western Port assets, and elsewhere we highlight other threats that need to be better characterised (e.g. invasive organisms). We have developed recommendations associated with several other threats of concern, for which we see important knowledge gaps.

Climate change

We have identified a need to refine our ability to predict changes to sea levels (average and storm surges) in Western Port and to incorporate catchment inputs and a range of other influences. We identify additional needs to understand changes to, particularly, nearshore habitats, and the potential for migration and mitigation, plus an understanding of the consequences of higher temperatures to fish and mangrove–saltmarsh interactions.

Catchment-derived toxicants

Toxicants remain a major knowledge gap. Western Port has a range of toxicant sources, and these sources differ dramatically in the mix of materials entering the bay. Despite the range of toxicants, there has been little recent assessment of their effects, and we are constantly becoming aware of chemicals capable of having strong effects at low concentrations. We identify this as a strategic research need, and identify a staged approach, in which the first step is to understand the extent to which toxicants spread beyond the catchment and drains into the wider Western Port.

Effects of harvesting

With the cessation of commercial fishing, recreational fishing is the main means of harvesting. It occurs over a wide area of the bay, and includes the shoreline collection of animals from the intertidal reefs and mudflats. Although the harvesting of mangroves ceased many years ago, it has left a legacy of denuded shorelines and increased erosion, particularly along the eastern parts of Western Port. Mangroves may be removed in the future if some proposed coastal developments go ahead. In concert with an increase in recreational fishing, fish-eating birds seem to have declined in Western Port over recent decades. Ghost shrimp are an important source of food for some bird and fish species, and there are high levels of ghost shrimp extraction for bait in some parts of Western Port.

The research needs are summarised in the following tables, grouped under these themes.

Hydrodynamics and sediment dynamics						
Theme	**No.**	**Priority**	**Brief Description**	**Details**	**Justification/Benefit**	**Chapters**
Improving hydrodynamic models of Western Port	**1**	1	Improve bathymetry of Western Port	Update bathymetry by incorporating the Future Coasts LIDAR surveys, DEM products and additional multi-beam campaigns.	Information on finer-scale hydrodynamics needed to accurately represent the effects of tidal flats and channels in circulation of nutrients and sediments.	4, 14
	2	1	Calibrate hydrodynamic models to ensure accurate representation of water movements	Additional data needed to calibrate, refine and validate the existing hydrodynamic, dispersion and catchment loading models. Existing and proposed deployment of ADCPs (current meters) would provide consistent dataset to validate hydrodynamic models for key forcing dynamics (tidal, weatherband).	Essential step to demonstrate confidence in the models used for informing decisions.	4, 14
	3	2	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. Needed for accurate predictions of climate effects.	4
Develop a complete sediment transport model	**4**	2	Measure residence time of sediments entering the bay	Determine temporal changes in sediment and sediment associated nutrient inputs to the bay over time. Determine the residence time of fine material in the tributary channels.	Data would enable assessment of the response to catchment rehabilitation works. This would also assist in determining the timeframe where benefits from remedial catchment action could be seen.	4, 14
	5	2	Refine understanding of effects of seagrass on sediment transport	Although sediment is identified as being a contributing factor in seagrass decline, there is little evidence that quantifies the dynamics (at appropriate scales) in areas of seagrass that would relate to sediment accumulation thresholds.	Necessary for sediment transport model, but information may be extrapolated from other areas, where similar seagrasses occur.	4, 10
	6	1	Estimate contribution of coastal erosion to nutrient and sediment budgets	This additional loading of nutrients associated with coastal erosion in the North Arm is currently not well quantified and requires time series wave modelling to integrate with the hydrodynamics and sediment processes. A pilot program is underway. This component should also include consideration of the relationship between mangroves and shoreline erosion.	Needed to quantify a source of sediments and nutrients that has been suggested by several reviews to be important. Needed to inform a full sediment budget that would place catchment inputs into context.	4, 14
Other physical environmental understanding	**7**	3	Incorporate contributions from groundwater and in-stream processes to provide more robust modelling.		Identifying the origin of nutrients from the catchment, atmosphere and within bay processes is important to prioritise management of water quality in Western Port.	4, 14
	8	2	Atmospheric inputs into Western Port	Quantify atmospheric inputs from prevailing emissions and episodic dust storms, ash and smoke at adequate observational scales. Likely to be done as part of current EPA/CSIRO project.	Comprehensive atmospheric modelling can provide a consistent and dynamic means of representing this elusive loading term.	4, 14
Climate effects	**9**	2	Identify contribution of waves to sea-level changes in Western Port	Future work should aim to quantify the contribution of waves to sea level extremes along the Victorian coastline, specifically in Western Port and in areas of the coast where there are links between open coast events and Western Port. High-resolution bathymetric LIDAR data sets that are being developed as part of the Future Coasts Program would allow these finer-scale studies.	Need estimation of contribution of waves to extreme sea levels as part of planning for climate change.	4
	10	2	Determine contribution of rainfall to coastal inundation (when accompanied by storm tide)	Determine contribution of rainfall to coastal inundation, including when accompanied by storm tide. Should be done in conjunction with R9.	In addition to coastal inundation due to extreme sea levels, a storm tide may also be accompanied by inundation due to rainfall. This additional contribution to inundation, which has been taken into account in previous studies, would potentially increase the area affected by inundation.	4
	11	2	Incorporate shoreline erosion into climate change predictions	The consideration of environmental processes that change the shoreline and adaptive responses should be priority areas of future work.	Previous studies haves regarded the topography of the coastline as being constant throughout the 21st century. However, during this time period, environmental processes, such as the erosion of beaches and soft cliffs, are likely to have changed the morphology of the shoreline.	4

15 Consolidated research needs and prioritisation

Ecosystem Processes						
Theme	**No.**	**Priority**	**Brief Description**	**Details**	**Justification/Benefit**	**Chapters**
A nutrient budget for Western Port	**12**	1	Determine preliminary N & P budget	Budget for N and P at the scale of the five basins/segments most commonly identified for WP, including contrast with budget for two recognised water quality segments in the current SEPP Schedule (F8) for Western Port. This should include an improved understanding of the speciation (bioavailable vs total nutrients) of nitrogen and phosphorus catchment inputs, including the effect of in-stream transformation between load monitoring sites and loads at the catchment discharges (see R7, R8).	Needed to understand the key sources and sinks of N and P in Western Port at scales relevant to management. Provides valuable (first cut) insight into the cycling and impact of nutrient loads. Different forms of N and P vary markedly in their bioavailability and thus, ecological response.	14
	13	1	Measure nutrient transformation in major habitats	Large-scale exchange measurements to compare material exchange (e.g. N and P) between the intertidal areas and the bay and whether this varies between vegetated unvegetated habitats. Process measurements to quantify key nutrient recycling pathways (e.g. denitrification, N-fixation) in the different benthic habitats.	Understanding differences in the key nutrient processes and pathways (e.g. storage, transformation and export) across the major habitats in Western Port will help identify the consequences of habitat changes (as previously observed) for nutrient management. The knowledge will also provide the mechanistic understanding that will help constrain the nutrient budget (R12) and process model (R14).	14
	14	2	Build a process-based biogeochemical model		Major building blocks exist (i.e. PortsE2 catchment model and the 3DD receiving water model including a primary production module). The expansion to include other key biogeochemical processes (e.g. denitrification, fixation) and improved hydrodynamic (R2) and sediment modules (R4) would provide further insight into the current system state and function, and enable us to predict the response of Western Port to management actions and future climate scenarios.	4, 14
Sediment and nutrient thresholds for important plants	**15**	1-3	Assess the degree of nutrient and light limitation of major primary producers	Assessment of nutrient (N vs P) and light limitation in the major primary producers (benthic microalgae, seagrass, macroalgae, phytoplankton).	The composition and biomass of the major primary producers in Western Port has major implications for associated food webs. Understanding the degree of nutrient and light limitation would significantly enhance our ability to predict the consequences of changes in nutrient and sediment inputs given their direct and indirect effects on nutrient availability and light levels.	14,10
	16	1-3	Determine water quality targets for sediments and nutrients that support seagrasses, microphytobenthos, reef algae, saltmarshes, and mangroves	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, and reef algae the lowest.	Anthropogenic pressures rarely act in isolation. In Western Port, interactive effects of sediment and nutrient loads are highly likely. Understanding the interactive effects and feedbacks would assist the prioritisation of management actions to reduce loads	14,10,13,8
	17	2	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. associated with climate change) stressors. Species contributing to chlorophyll-a measures would determine whether elevated levels in Corinella segment are related to excess nutrients and uptake in the water column or reflect resuspension of benthic microalgae.	5,14
	18	2	Determine the role played by dead plant material from the dominant vascular plants in the availability, transport and transformations of nutrients, including for higher trophic levels	Need better understanding of the effects of vascular plant detritus (seagrasses, mangroves and coastal saltamarsh) on nutrient budgets, productivity and mangrove survival, and how the detrital path compares to microalgal foodwebs.	Vascular plants produce a significant biomass of detritus (from the plants themselves and from associated epiphytes) that can be deposited locally in the seabed or transported to other habitats. The consequences for detrital based food webs and nutrient production would depend on transport processes and how labile the detritus is. This may have major implications for lateral energy flow between habitats, energy transfer to higher trophic levels and nutrient availability. The comparison with microalgal-based food webs will provide additional information about the ecological consequences of seagrass loss, which leads to a decline in seagrass detritus.	8

How different is the Western Port ecosystem from when it was described in detail in 1975?						
Theme	**No.**	**Priority**	**Brief Description**	**Details**	**Justification/Benefit**	**Chapters**
Resolve seagrass identities	**19**	1	Determine which *Zostera* seagrasses are present in Western Port	Taxonomic identification and mapping of seagrass. Genetic identification of *Zostera tasmanica* and *Z. nigricaulis* including historical specimens, as planned for Port Phillip Bay.	Necessary to understand which species of *Zostera* and determine amount of knowledge transfer possible from other seagrass studies.	10
Characterise present biodiversity	**20**	2	Determine whether deep channels harbour reef fauna	Examination of walls and floor of deep channels, to determine if they act as *de facto* reefs, and if this and if this information alters our picture of overall Western Port 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 modifications to channels.	13
	21	2	Identify differences between current state of Western Port soft sediment faunal assemblages and earlier descriptions	This should be done as a priority for unvegetated soft sediments. It should be done as a lower priority for other habitats, to assess degree of change.	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 an improved basis for assessing threats to biodiversity assets within Western Port.	6-13
	22	2	Estimate extent of invasion of key habitats	Introduced species - extent of invasions, spp present in various habitats. Would be achieved as part of R22.	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.	7,13
	23	2	Determine affinities of Western Port biota	The geological history of Western Port suggests a stronger link with the East coast of Australia than with Port Phillip Bay. There is the possibility of some immigration from Port Phillip Bay (native and invasive species) that may have resulted in some 'homogenisation' of the fauna of the two bays.	Used for refining the identification of individual marine assets.	7
	24	3	Characterise importance of saltmarshes and mangroves for biodiversity	Determine whether Western Port saltmarsh and mangroves harbour a fauna that differs from that occurring elsewhere in southeastern Australia. Some assessment already made for mangrove fish, but not done for birds or invertebrates. Clarify the taxonomic and structural diversity of coastal saltmarsh, particularly with reference to the purported lack of species diversity.	Used in refining the identification of individual marine assets.	8
Trends through time	**25**	2	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.	Determine the loss or gain of mangroves and saltmarshes in Western Port, including changes to locations of saltmarsh–mangrove boundaries. It is still unclear whether mangroves in Western Port are advancing seaward or landward. Assessment to include data from the 1840s from surveyors' maps, aerial photography (1940s and later), and other remotely sensed images.	Two earlier reports used historical surveys to describe losses of Western Port peripheral vegetation. These analyses could be repeated with the specific intention of quantifying losses, gains and floristic changes around different parts of the Western Port coast. Also use historical aerial photographs, as in studies of Gippsland wetlands.	8, 9

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Deepening our understanding of ecological links						
Theme	**No.**	**Priority**	**Brief Description**	**Details**	**Justification/Benefit**	**Chapters**
Resilience of habitat forming species	**26**	1	Determine capacity for *Zostera* to recover and colonise new areas	Studies of *Zostera* spp. biology, reproductive strategies, and environmental tolerances (light, temperature. salinity, and nutrients). Build on earlier studies. R15 and 16 would help address environmental tolerances.	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.	10
	27	2	Identify determinants of saltmarsh and mangrove recovery and seedling establishment	Morphodynamic patterns: more detailed understanding on the effects of sedimentation/ erosion on saltmarsh and mangrove recovery as well as seedling survival are needed, as well as how mangroves are affecting the sediment budget on different scales.	Provide better understanding of scope for revegetating areas of shoreline erosion.	8
Functional links between organisms and habitat	**28**	1	Determine linkages between fish and habitats	Identification of fish assemblages associated with *Amphibolis* seagrass beds, subtidal reefs, Caulerpa cactoides algal beds, and benthic sessile invertebrate habitats. This assessment could also include effects of non-native Codium. This work has already been done for mangroves.	Critical habitats for sustaining fish at various life stages currently unknown for many species and habitats. Need to know the value of habitats in this respect to justify protection where necessary. Benefit is to sustain populations of important fish species.	11
	29	2	Relationships between sea levels, sedimentation/erosion rates and vascular plant communities	Factors controlling the distribution of different plant taxa in coastal saltmarsh, including their relationship with elevation, sedimentation/ erosion, and tidal inundation:	The long-term survival of mangroves and coastal saltmarsh in the face of increasing sea levels depends on their ability to maintain sediment levels higher than mean sea level. There are some long-term data (sediment elevation tables) from monitoring programs, which need to be maintained or expanded.	9
	30	3	Updated and finely scaled spatial description of subtidal soft sediment areas	Surveys to identify areas of functional importance, e.g. ecosystem engineering and biogenic structures. Determine the features that support high benthic biodiversity in Western Port.	Biomass estimates of macrofauna, particularly those that are important for nutrient cycling. This information feeds into geochemical models.	7
	31	2	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. Need 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 environmental changes would affect the functioning of ecosystems or parts thereof, even if they are not in the direct path of any disturbance event. Would 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.	8,9
Species of particular interest	**32**	3	Investigate the marine and estuarine requirements of the listed Australian grayling	Work on the estuarine requirements of the listed Australian grayling in Westernport is currently being undertaken by the Arthur Rylah Institute and Melbourne Water. The biology of the marine larval/early juvenile phase is currently unknown with respect to factors such as distance dispersed.	Without information on the biology of the marine phase of Australian Grayling, threats cannot be determined and management actions cannot be implemented.	11
	33	2	Determining the species, locations and timing of fish spawning	Fish eggs and larvae have been sampled in only a limited way in the southern part of Western Port. Bay-wide sampling at monthly intervals over 1–3 years is needed to identify species, location and time of fish spawning (including Snapper and Australian Grayling). This work would also contribute to the biodiversity assessment of Western Port.	There is little information on the importance of Western Port 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.	11
	34	2	Determine relative significance of shorebird and waterbird intertidal feeding areas	Systematic mapping of low-tide feeding areas of shorebirds and waterbirds throughout the bay and an evaluation of significance.	This has never been carried out systematically and is a necessary next step to determining effects on shorebirds and waterbirds of sea-level rise, land claim and recreational activities.	12
	35	1	Examine the trends of fish-eating birds in Western Port and Corner Inlet	Comparison of the long-term trends in fish-eating birds in Western Port (35 years of data) and Corner Inlet (30 years of data). The data have been collected, and only collation, analysis and interpretation are required.	This comparison would allow identification of those species that are in decline especially in Western Port and hence be more associated with processes in the bay.	12

Understanding particular threats

In addition to the individual items presented at the end of individual chapters, we included the following:

Assessment of toxicant concentrations away from discharge points in northern Western Port

In several places in this report the issue of toxicants has been raised, but there is considerable uncertainty about the actual risks posed by these chemicals. This uncertainty relates to toxicants around major discharge points, and increases with distance from sources.

Inputs from major catchments have been characterised to some degree, with sampling of a wide range of toxicants upstream in catchments and in estuarine sediments (CAPIM), along with measurement of nutrient inputs from major discharges in north and eastern parts of Western Port. This sampling has revealed a wide range of toxicants in estuarine sediments.

There are three fundamental knowledge gaps in relation to toxicants in Western Port:

- Do toxicants extend beyond major discharge points at levels that are a concern, based on current sediment guidelines?
- If so, what is the bioavailability of these toxicants?
- Are water column concentrations of any toxicants a concern to Western Port assets?

The major discharge points differ considerably in the kinds of toxicants they emit, and there is also variation in the sensitivity of components of the Western Port ecosystem to particular toxicants. For example, hydrocarbons are a potentially serious threat to mangroves because of the effects of oils on pneumatophores. Seagrasses may be sensitive to herbicides entering the water, and recent attention has been drawn to the risks posed by endocrine disrupting compounds to a range of vertebrates, particularly fish.

This research need can be addressed in two stages:

1. Sample sediments away from the major entry points (Watsons Creek, Lang Lang River, Bunyip River, Bass River) and screen for the major toxicants. This would be most useful if combined with the sampling done through the Victorian Centre for Aquatic Pollution Impacts and Monitoring (CAPIM), and should screen the same toxicants.
2. If sediment level concentrations of particular toxicants are high enough to be a concern, initiate the second phase of assessing ecological risk:
   - Sample marine fauna (and/or flora) in areas of high and low concentrations.
   - Where possible, use passive samplers to determine the biologically available segment of particular toxicants. Passive samplers provide more efficient estimates than conventional water quality samples, but they are not available for all toxicants of concern. CAPIM is developing new passive samplers that might be deployed for this purpose.

Theme	No.	Priority	Brief Description	Details	Justification/Benefit	Chapters
Toxicants	**36**	1	Initial estimate of risk from toxicants	Measure levels of toxicants away from major discharge points.	First cut at indicating whether toxicants are likely to be a threat across wide areas of Western Port.	7-14
	37	2–3	Impacts of toxicants on vegetation	Understand effects of toxicants on habitat-forming species, including seagrasses (Priority 2) mangroves (Priority 2), saltmarshes (Priority 2) and algae (Priority 3).	Input to refinement of water quality targets. Conditional on outcome for R36.	8, 9, 10
	38	2	Investigate the climate change and toxicant effects on fish	Investigate the tolerances of fish species, particularly more vulnerable eggs and larvae, to reduced water quality in north of bay, and increased temperature associated with climate change. Concurrent with laboratory studies to determine vulnerability of eggs and larvae to varying levels of water quality parameters including suspended sediments, toxicants, temperature, salinity and UVB.	The early life stages are the key to sustaining healthy fish populations but are also the most vulnerable to changes in water quality though climate change or toxicant input. Understanding the tolerances of eggs and larvae of key species to water quality changes would help in setting water quality targets.	11
Harvesting	**39**	1	Effects of recreational fishing on fish stocks	Continue existing monitoring of recreational fishing, and extend to include fisheries-independent surveys. For example, the annual survey of juvenile King George Whiting in Port Phillip Bay could be extended to include Western Port.	The closure of commercial netting in Western Port means that commercial catch rates can no longer be used to measure trends in fish populations and fishery independent monitoring is needed to track changes in abundances. This could inform future regulation and enforcement considerations by government.	11
	40	3	Effect of shoreline harvesting on invertebrates	Determine degree and impact of recreational harvesting on intertidal reefs and mudflats (i.e. bait pumping) around Western Port.	This information might link to changes in enforcement in future, an action that would be the responsibility of Fisheries Victoria. This item is considered a low priority for substantial investment, but might be appropriate as a student project through one of the tertiary institutions.	13
Climate Change	**41**	2	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.		13
	42	2	Effects of sea level rise on shore birds	Model likely changes to the locations of high-tide bird roosting sites and areas of intertidal mudflat feeding areas at various degrees of sea-level rise.	Sea-level rise may reduce feeding areas which is particularly significant for small migratory shorebirds. Modelling likely changes to elevation and extent of mudflats would allow prediction of effects. Sea-level rise will change the distribution of roosting sites around the bay, and future management of birds will require the prediction of roosting site losses and gains.	12
Changes to Habitat Quality	**43**	2	Factors determining roost selection in shorebirds including the role of human disturbance	Evaluate the primary factors determining roost site selection by shorebirds, including levels of human disturbance, other disturbances, predation risk, microclimate, energetic considerations and geomorphology.	Human activity is increasing in the bay, and disturbance at high-tide roosts is an important issue for many stakeholders. An assessment of what is important in roost quality would allow appropriate management responses to be made, as well as guide any plans for artificial roosts.	12

Dependencies

The research needs have been grouped according to their role in understanding the Western Port ecosystem, with physical processes listed first. In developing a Western Port research program, there are some important dependencies, two of the most important of which are (a) the sequence in which research should be done, and (b) cross-links between research items.

Who's on first?

Some of the recommended research needs could be done immediately because they satisfy immediate research or management goals, but some form a logical sequence so that one is necessary before the next can be done; for example, where the results of one research program are needed before we can determine exactly how another program is carried out.

The case where this is most obvious is the relationships between water quality, seagrasses and a range of Western Port assets. To protect these assets, extensive and healthy seagrasses are required, and we need to understand how water quality limits seagrass condition. This information would allow water quality targets to be set for Western Port waters and provide tools for prioritising catchment management options. We have recommended research programs to develop these thresholds. We have also identified uncertainty about the relative roles of nutrients and suspended sediments in limiting seagrass, and a lack of any data to assess the risk from toxicants. A research program to identify suspended sediment thresholds for seagrass would be different from one targeting nutrients, and much less complex than a program to simultaneously examine nutrients and sediments.

For these reasons, research on a nutrient budget (R12 to R14) should precede the determination of water-quality thresholds (R16). Similarly, the first step in understanding nutrients in Western Port (R12) will quickly provide information that will allow tasks R13 and R14 to be designed more completely.

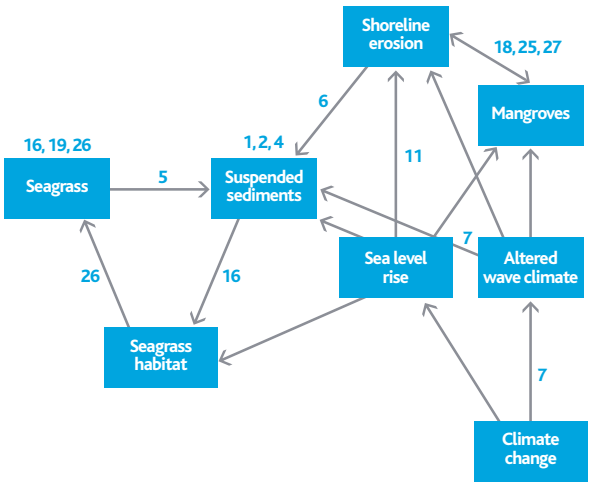
As a second example, we recommend further investigations into the conditions that allow mangrove seedlings to establish. This is an important research gap given the shoreline erosion around some sections of Western Port (e.g. Grantville). This research is important as a way of stemming habitat loss, but its priority would be altered by the contribution of shoreline erosion to the sediment budget of Western Port. This is a long-standing knowledge gap that has prevented us from understanding the sources of suspended sediments in Western Port. A finding that shoreline erosion is a substantial source of sediments would mean that actions to reduce this erosion would improve conditions for seagrasses (and possibly microphytobenthos), and would make mangrove revegetation an even higher priority.

The other important aspect of timing is that some of our recommendations relate to climate change. As discussed earlier, steps to limit the rise in atmospheric (and oceanic) CO_2 are urgent, but mitigating the impacts will need to be done increasingly more in coming decades. For example, we see it as important to quantify the extent to which the few intertidal reefs in Western Port will be lost, but the sea level rise that will threaten these reefs will not be seen until much later this century. Other recommendations are much more immediate, to inform actions on much shorter scales. For example, the calibration of the Western Port hydrodynamic model is fundamental to a range of tasks related to predicting the effects of catchment changes, major capital projects, urbanisation, and other potential disturbances.

Linkages

We have seen that the research needs are not independent: information from one will feed into others. For example, there is an important and dynamic relationship between suspended sediments and seagrass. It is possible to describe some of these linkages graphically (Figure 15.1). Knowledge gaps associated with these relationships are reflected in a substantial number of our recommendations.

Figure 15.1 Example of relationships between sediments and seagrass cover in Western Port, showing important functional relationships. The numbers in blue indicate numbered recommendations from the tables earlier in this chapter.



But these linkages are complex. Although we can identify (in most cases) the direction of the changes, predicting the net effect is difficult because it depends on the magnitude of some of the effects. These complexities are well known to managers, and the tools required to deal with them are diverse.

Where the target of management is simple (e.g. the survival of the San Remo reef community; see Chapter 13), management options can be clear. In Western Port, for example, the small reef area at San Remo is listed under the Flora and Fauna Guarantee as an ecological community notable for its high diversity of nudibranch molluscs. Because of its location in the well-flushed south-eastern section of Western Port, suspended sediments, nutrients, and toxicants, are not expected to be a major threat. Instead, the plausible threats to this asset are effects of visitation and recreation, including collecting, as the reef is a matrix of reefs and seagrass, and in the long term, climate change. Devising management options for the intensity and type of visitation is straightforward, and it would be simple, were it deemed necessary, to quantify the threat posed by visitation and collection. Similarly, rising sea levels associated with climate change may result in previously intertidal areas becoming submerged, and there is limited scope for shoreward migration of the intertidal zone. The number of management options and research needs is small and relate to mitigation through coastal hardening and an assessment of the likelihood of this community surviving if it were completely subtidal.

But some research requires simultaneous examination of a range of influences on water quality and a number of important feedback loops. A substantial number of management actions are possible, but the challenge is to identify those actions likely to result in the most beneficial outcomes. Exploring these options is likely to require formal quantitative relationships to be determined between various threatening processes and Western Port assets. As discussed in Chapter 6, such models can be extremely complex, but the minimum model required for Western Port is a coupled biogeochemical model (R14), which also includes changes to major habitat-forming plants, particularly seagrasses and mangroves.

The need for a complex model (which is partly developed already) also has implications for the management of future research. It will be important to coordinate research to ensure that research proceeds in such a way that dependent research projects have the necessary information available to them when they are designed in detail. With an ecosystem model as the endpoint, it will also be important to coordinate research so that the data collected from individual research projects is suitable for the needs of that model.

16 References

Acevedo S., Jenkins G. & Kent J. (2010) *The horizontal and vertical distribution of fish eggs and larvae in relation to the proposed desalination plant for Victoria, September 2007 to August 2008*. Department of Primary Industries, Victoria.

Adam P. (1990) *Saltmarsh ecology*. Cambridge University Press, Cambridge.

Adam P. (1994). Saltmarsh and mangrove. In: Adam P. (ed.) *Australian vegetation*. 2nd Edition. Cambridge University Press, Cambridge, pp. 395-435.

Adam P. (2008) Likely impacts of climate change on coastal marshes and floodplains. *Climate change, sea level rise and wetland adaptations*. Conference proceedings, Sydney.

AGC Woodward-Clyde Pty Limited (1993) *Coastal Zone Inquiry (Australia). Victorian case study - coastal management and decision making processes in the Westernport region*. Victorian Department of Premier and Cabinet & Resource Assessment Commission Coastal Zone Inquiry.

Ahsanullah M. (1976) Acute toxicity of cadmium and zinc to seven invertebrate species from Western Port, Victoria. *Australian Journal of Marine and Freshwater Research* 27:187-196.

Ahsanullah M., Negilski D.S. & Tawfik F. (1980) Heavy-metal content of *Callianassa* spp in Western Port. *Australian Journal of Marine and Freshwater Research* 31:847-850.

Andrew D., Lumsden L. & Dixon J. (1984) Sites of zoological significance in the Westernport region. E*nvironmental Study Series* No. 327. Ministry for Conservation, Victoria.

Anon. (2004) *Flinders comprehensive sanitary survey 2003*. Water Ecoscience, Report No. 2741-010104.

Anon. (2005) *Flinders Annual Update Report 2003-2004*. Water Ecoscience, Report No. 3162-270405.

Anon. (2007) *Flinders Harvesting Area Triennial Update Report 2003 -2007*. Ecowise, Environmental Report 3321-130707.

Anon. (2008) *Flinders Annual Update Report 2007- 2008*. Ecowise, Environmental Report VE326800-010608.

Anon. (2010) *Flinders Aquaculture Fisheries Reserve Annual Reporting of Shellfish Harvesting Area*. Musselco Australia Pty Ltd & Fisheries Victoria (DPI).

ANZECC (2000) *Core environmental indicators for reporting on the state of the environment*. Australian and New Zealand Environment Conservation Council State of the Environment Reporting Task Force.

ANZECC & ARMCANZ (2000) *Australian and New Zealand guidelines for fresh and marine water quality*. Australian and New Zealand Environment and Conservation Council, Agricultural and Resource Management Council of Australia and New Zealand.

Argent R.M. (2006) *PortsE2: A decision support system for water quality improvement in Port Phillip and Western Port*. CEH, Report No. 01/06.

Argent R.M., Perraud J.M., Rahman J.M., Grayson R.B. & Podger G.M. (2009) A new approach to water quality modelling and environmental decision support systems. *Environmental Modelling and Software* 24(7): 809-818.

Arnott G.H. (1974) An ecological study of the zooplankton of Westernport and adjacent Bass Strait waters. *Environmental Study Series* No. 95. Ministry for Conservation, Victoria.

Arnould J.P.Y. & Kirkwood R. (2008) Habitat selection in a benthic diver: the foraging areas of female Australian fur seals (*Arctocephalus pusillus doriferus*). *Aquatic Conservation: Marine and Freshwater Systems* 17:S53-S67.

Arundel H., Pope A. & Quinn G. (2009) *Victorian Index of Estuary Condition. Recommended themes and measures. Draft Report*. Deakin University, School of Life and Environmental Sciences.

Atalah J. & Crowe T.P. (2010) Combined effects of nutrient enrichment, sedimentation and grazer loss on rock pool assemblages. *Journal of Experimental Marine Biology and Ecology* 388:51-57.

Attiwill P.M. & Clough B.F. (1978) Productivity and nutrient cycling in the mangrove and seagrass communities of Westernport Bay. *Environmental Studies Series* No. 253. Ministry for Conservation, Victoria.

Baggalley S. (2009) *Population biology and trophic interactions of two newly discovered mangrove crab species in the Upper Spencer Gulf, South Australia*. Honours thesis, Flinders University.

Baird B. & Dann P. (2003) The breeding biology of the Hooded Plover, *Thinornis rubricollis*, on Phillip Island, Victoria. *Emu* 103:323-328.

Bamford M., Watkins D., Bancroft W., Tischler G. & Wahl J. (2008) *Migratory Shorebirds of the East Asian - Australasian Flyway; Population Estimates and Internationally Important Sites*. Wetlands International - Oceania.

Barnard J.L. & Drummond M.M. (1978) Gammaridean Amphipoda of Australia, Part III. The Phoxocephalidae. *Smithsonian Contributions to Zoology* 245:1-551.

Barnett E.J., Harvey N., Belperio A.P. & Bourman R.P. (1997) Sea-level indicators from a holocene tide-dominated coastal succession, Port Pirie, South Australia. *Transactions of the Royal Society of South Australia* 121:125-135.

Barson M. & Calder D. (1981) The vegetation of the Victorian coast. *Proceedings of the Royal Society of Victoria* 92:55-65.

Bathgate R.S. (2010) Ecological processes and connectivity of gastropod populations in a system of Marine Protected Areas. PhD thesis, University of Melbourne.

Bax N., Williamson A., Aguero M., Gonzales E. & Geeves W. (2003) Marine invasive alien species: a threat to global biodiversity. *Marine Policy* 27:313-323.

Beardall J. & Raven J.A. (2004). The potential effects of global climate change on microalgal photosynthesis, growth and ecology. *Phycologia* 43: 26-40.

Benson D., Howell J. & McDougall L. (1996) *Mountain devil to mangrove. A guide to natural vegetation in the Hawkesbury-Nepean catchment*. Royal Botanic Gardens, Sydney.

Bird E.C.F. (1971) Mangroves as land builders. *The Victorian Naturalist* 88:189-197.

Bird E.C.F. & Barson M.M. (1975) Shoreline changes in Westernport Bay. *Proceedings of the Royal Society of Victoria* 87:69-78.

Bird E.C.F. (1980a) Historical changes on sandy shorelines in Victoria. *Proceedings of the Royal Society of Victoria* 91:17-32.

Bird E.C.F. (1980b) Mangroves and coastal morphology. *The Victorian Naturalist* 97 48-58.

Bird E.C.F. (1986) Mangroves and intertidal morphology in Westernport Bay, Victoria, Australia. *Marine Geology* 69:251-271.

Bird E.C.F. (1993) *The coast of Victoria. The shaping of scenery*. Melbourne University Press, Parkville, Melbourne.

Bird F.L. & Poore G.C.B. (1999) Functional burrow morphology of *Biffarius arenosus* (Decapoda : Callianassidae) from southern Australia. *Marine Biology* 134:77-87.

Bird F.L., Boon P.I. & Nicholas P. (2000) Physicochemical and microbial properties of burrows of the deposit-feeding Thalassinidean shrimp *Biffarius arenosus* (Decapoda: Callianassidae). *Estuarine, Coastal and Shelf Science* 51:279-291.

Bishop M.J., Coleman M.A. & Kelaher B.P. (2010) Cross-habitat impacts of species decline: response of estuarine sediment communities to changing detrital resources. *Oecologia* 163:517-525.

Black K.P. & Hatton D.N. (1994) *South-eastern ocean outfall study. Volume 2: 3-dimensional hydrodynamic and dispersal (plume) modelling*. VIMS report to Melbourne Water.

Black K.P. (2001) *The 3DD suite of numerical process models*. ASR Ltd, New Zealand.

Black K.P., Harrison S., Bosserelle C. (2011) *Port Phillip and Western Port receiving water quality modelling: nutrient dynamics and primary production*. ASR technical report 2007-EPA3. Published as EPA publication No. 1379.

Blake S. & Ball D. (2001) *Victorian Marine Habitat Database: Seagrass Mapping of Western Port*. Report No. 29. Marine and Freshwater Resources Institute.

Ball D., Blake S. & Plummer A. (2006) Review of Marine Habitat Classification Systems. *Parks Victoria Technical Series* No. 26. Parks Victoria, Melbourne.

Bartoli M., Nizzoli D., Welsh D.T. & Viaroli P. (2000) Short-term influence of recolonisation by the polycheate worm Nereis succinea on oxygen and nitrogen fluxes and denitrification rates: a microcosm simulation. *Hydrobiologia* 431:165-174.

BMT WBM (2007) *Port Phillip Bay and Western Port 'PortsE2' model application and assessment project - scenario assessments*. WBM Oceanics, Brisbane.

Bonan G.B. (2002) *Ecological climatology. Concepts and applications*. Cambridge University Press, Cambridge.

Boon P.I. & Cain S. (1988) Nitrogen cycling in saltmarsh and mangrove sediments at Western Port, Victoria. *Australian Journal of Marine and Freshwater Research* 39:607-623.

Boon P.I. & Brock M.G. (1994) Plants and processes in wetlands: an introduction. *Australian Journal of Marine and Freshwater Research* 45:1369-1374.

Boon P.I., Bird F.L. & Bunn S.E. (1997) Diet of the intertidal callianassid shrimps *Biffarius arenosus* and *Trypea australiensis* (Decapoda: Thalassinidea) in Western Port (southern Australia), determined with multiple stable-isotope analyses. *Marine and Freshwater Research* 48:503-511.

Boon P.I., Raulings E., Roache M. & Morris K. (2008) Vegetation changes over a four-decade period in Dowd Morass, a brackish-water wetland of the Gippsland Lakes, south-eastern Australia. *Proceedings of the Royal Society of Victoria* 120:403-418.

Boon P.I., White M. & Sinclair S. (2010) Climate change impacts on Victoria's coastal vegetation (mangroves and saltmarsh): Western Port case study. *Institution of Engineers Australia: Practical responses to climate change 2010 conference*. Melbourne, 2010. Paper/Presentation No. 107.

Boston K.G. (1981) The introduction of *Spartina townsendii* (s.l.) to Australia. *Occassional paper* No.6. Melbourne State College, Melbourne.

Braccini J.M., Walker T.I. & Conron S.D. (2008) *Evaluation of effects of targeting breeding elephant fish by recreational fishers in Western Port*. Draft final report to Fisheries Revenue Allocation Committee. Fisheries Research Branch, DPI, Queenscliff.

Bridgewater P.B. (1975) Peripheral vegetation of Western Port Bay. P*roceedings of the Royal Society of Victoria* 87:69-78.

Bridgewater P.B. & Kaeshagen D. (1979) Changes induced by adventive species in Australian plant communities. In: Bridgewater P.B. & Kaeshagen D. (eds.) *Werden und Vergehen von Plflanzengesellschaften*. Cramer, Lehre, pp. 561-579.

Bridgewater P.B., Rosser C. & de Corona A. (1981) *The saltmarsh plants of southern Australia*. Botany Department, Monash University, Clayton.

Bridgewater P.B. (1982) Phytosociology of coastal saltmarshes in the Mediterranean climatic region of Australia. *Phytocoenologia* 10:257-296.

Bridgewater P.B. (1999). Mangroves. In: Bridgewater P.B. (ed.) *Flora of Australia. 2nd Edition*. Department of the Environment and Heritage, Canberra & CSIRO Publishing, Melbourne, pp. 416-420.

Brooke C. & Kinrade P. (2006) *Climate Change Impacts in Western Port: Report prepared for the Western Port Greenhouse Alliance*. CSIRO Marine and Atmospheric Research - Climate Impacts and Risk Group, Marsden Jacob Associates and Department of Sustainability and Environment.

Brown V., Davies S. & Synnot R. (1990) Long-term monitoring of the effects of treated sewage effluent on the intertidal macroalgal community near Cape Schanck, Victoria, Australia. *Botanica Marina* 33:85-98.

Bucher D. & Saenger P. (1991) An inventory of Australian estuaries and enclosed marine waters: an overview of results. *Australian Geographical Studies* 29:370-381.

Bulthuis D.A. (1976) Bioassay of phytoplankton nutrient limitation. *Environmental Studies Series* No. 121. Ministry for Conservation.

Bulthuis D.A. (1977) Phytoplankton biomass and productivity in Westernport. Preliminary data. *Environmental Study Series* No. 179. Ministry for Conservation.

Bulthuis D.A. (1981) Distribution and summer standing crop of seagrasses and macroalgae in Western Port, Victoria. *Proceddings of the Royal Society of Victoria* 92:107-112.

Bulthuis D.A. & Woelkerling W.J. (1981) Effects of in situ nitrogen and phosphorus enrichment of the sediments on the seagrass *Heterozostera tasmanica* (Martens ex Aschers) den Hartog in Western Port, Victoria, Australia. *Journal of Experimental Marine Biology and Ecology* 53:193-207.

Bulthuis D.A. (1983) Effects of in situ light reduction on density and growth of the seagrass *Heterozostera tasmanica* (Martens ex Aschers) den Hartog in Western Port, Victoria, Australia. *Journal of Experimental Marine Biology and Ecology* 67:91-103.

Bulthuis D.A. & Woelkerling W.J. (1983a) Seasonal variation in standing crop, density and leaf growth rate of the seagrass, *Heterozostera tasmanica*, in Western Port and Port Phillip Bay, Victoria, Australia. *Aquatic Botany* 16:111-136.

Bulthuis D.A. & Woelkerling W.J. (1983b) Biomass accumulation and shading effects of epiphytes on leaves of the seagrass (*Heterozostera tasmanica*) in Victoria, Australia. *Aquatic Botany* 16:137-148.

Bulthuis D.A. (1984) Loss of seagrass in Western Port, a proposal for further research, 1984/1985. *Internal Report Series* No. 72. Ministry for Conservation, Victoria.

Bulthuis D.A., Brand G.W. & Mobley M.C. (1984) Suspended sediments and nutrients in water ebbing from seagrass-covered and denuded tidal mudflats in a southern Australian embayment. *Aquatic Botany* 20:257-266.

Bunt J.S., Williams W.T. & Bunt E.D. (1985) Mangrove species distribution in relation to tide at the seafront and up river. *Australian Journal of Marine and Freshwater Research* 36:481-492.

Bunt J.S. (1995) Continental scale patterns in mangrove litter fall. *Hydrobiologia* 295:135-140.

Burns K.A. & Smith J.L. (1977) Distribution of petroleum hydrocarbons in Westernport Bay (Australia): Results of chronic low level inputs. In: Burns K.A. & Smith J.L. (eds.) *Fate and effects of petroleum hydrocarbons in marine organisms and ecosystems*. Pergamon Press, New York, p. 442.

Burns K.A. & Smith J.L. (1982) Hydrocarbons in Victorian coastal ecosystems (Australia) - chronic petroleum inputs to Western Port and Port Phillip Bays. *Archives of Environmental Contamination and Toxicology* 11:129-140.

Butcher A.D. (1979) The Westernport Bay Environmental Study. *Marine Geology* 30:1-10.

Butler S. & Bird F.L. (2007) Estimating density of intertidal ghost shrimps using counts of burrow openings. Is the method reliable? *Hydrobiologia* 589:303-314.

Butler S.N. & Bird F.L. (2010) Monitoring the macroinvertebrates and soft sediments in the Marine National Parks in Western Port. *Parks Victoria Technical Series* No. 60. Parks Victoria, Melbourne.

Byrne M., Soars N., Ho M.A., Wong E., McElroy D., Selvakumaraswamy, P., Dworjanyn, S. A. & Davis A.D. (2010) Fertilisation in a suite of coastal marine invertebrates from SE Australia is robust to near-future ocean warming and acidification. *Marine Biology* 157:2061-2069.

Cain S. & Boon P.I. (1987) Cellular osmotica of plants in relation to sediment nitrogen and salt contents in mangroves and saltmarshes at Western-Port, Victoria. *Australian Journal of Marine and Freshwater Research* 38:783-794.

Callaway J.C. & Josselyn M.N. (1992) The introduction and spread of Smooth Cordgrass (*Spartina alterniflora*) in South San Francisco Bay. *Estuaries* 15:218-226.

Campbell S.J. (1999) Occurrence of Codium fragile subsp. tomentosoides (Chlorophyta: Bryopsidales) in marine embayments of southeastern Australia. *Journal of Phycology* 35:938-940.

Campbell S., Miller C., Steven A. & Stephens A. (2003) Photosynthetic responses of two temperate seagrasses across a water quality gradient using chlorophyll fluorescence. J*ournal of Experimental Marine Biology and Ecology* 291:57-78.

Campbell S.J. & Miller C.J. (2002) Shoot and abundance characteristics of the seagrass *Heterozostera tasmanica* in Westernport estuary (south-eastern Australia). *Aquatic Botany* 73:33-46.

Carlton J.T. (2000). Global change and biological invasions in the oceans. In: Carlton J.T. (ed.) *Invasive Species in a Changing World*. Island Press, Washington, pp. 31-53.

Carr G.W. (1979) *Survey of Victorian coastal saltmarsh distribution in relation to the habitat of the Orange-bellied Parrot*. Report to Kinhill Planners Pty Ltd and ICI Australia.

Chainho P., Costa J.L., Chaves M.L., Lane M.F., Dauer D.M. & Costa M.J. (2006) Seasonal and spatial patterns of distribution of subtidal benthic invertebrate communities in the Mondego River, Portugal - a poikilohaline estuary. *Hydrobiologia* 555: 59-74.

Chambers L.E. & Loyn R.H. (2006) The influence of climate variability on numbers of three waterbird species in Western Port, Victoria, 1973-2002. *International Journal of Biometeorology* 50:292-304.

Chaney A.H., Chapman B.R., Karges J.P., Nelson D.A. & Schmidt R.R. (1978) *Use of Dredged Material Islands by Colonial Seabirds and Wading Birds in Texas. Final report to Information for the Defence Community*. Texas A and I University.

Chapman V.J. (1977). Introduction. In: Chapman V.J. (ed.) Ecosystems of the world. *1 Wet coastal ecosystems*.Elsevier, Amsterdam, p. 428.

Chareonpanich C., Montani S. & Tsutsumi H. (1994) Roles of a deposit-feeding polychaete, Capitella sp.: I. On the biological and chemical changes of the experimental marine sediment systems. *Kagawa Daigaku Nogakubu Gakujutsu Hokoku* 46:21-26.

Chiaradia A., Dann P., Jessop R. & Collins P. (2002) The diet of Crested Tern (*Sterna bergii*) chicks on Phillip Island, Victoria, Australia. *Emu* 102:367-371.

Chiaradia A., Costalunga A. & Knowles K. (2003) The diet of Little Penguins (*Eudyptula minor*) at Phillip Island, Victoria, in the absence of a major prey–Pilchard (*Sardinops sagax*). *Emu* 103:43-48.

16 References

Chidgey S., Crockett P., Lewis, J. & Watson, J. (2009) *Port of Hastings. Stage 1 Development. Marine Ecosystem Preliminary Considerations. Report to AECOM and the Port of Hastings Corporation*. CEE Consultants Pty Ltd.

Chidgey S. & Crockett P. (2010) *Stony Point to Cowes vehicular ferry: marine ecosystem preliminary considerations*. CEE Consultants Pty Ltd and Meinhardt Infrastructure and Environment

Church J., White N., Hunter J., McInnes K., Cowell P. & O'Farrell S. (2008). Sea-level rise. In: Church J., White N., Hunter J., McInnes K., Cowell P. & O'Farrell S. (eds.) *Transitions: pathways towards sustainable urban development in Australia*. CSIRO Publishing.

Church J., Woodworth P.L., Aarup T. & Wilson S. (eds.) (2010) *Understanding sea-level rise and variability*. Wiley, UK.

Claridge G. & WBM Oceanics (1997) *Guidelines for managing visitation to seabird breeding colonies*. Great Barrier Reef Marine Park Authority, Queensland.

Clarke L.D. & Hannon N.J. (1967) The mangrove swamp and salt marsh communities of the Sydney district. I. Vegetation, soils and climate. *Journal of Ecology* 55:753-771.

Clarke L.D. & Hannon N.J. (1969) The mangrove swamp and saltmarsh communities of the Sydney district. II. The holocoenotic complex with particular reference to physiography. *Journal of Ecology* 57:213-234.

Clarke L.D. & Hannon N.J. (1970) The mangrove swamp and salt marsh communities of the Sydney district III. Plant growth in relation to salinity and waterlogging. *Journal of Ecology* 58:351-369.

Clarke L.D. & Hannon N.J. (1971) The mangrove swamp and salt marsh communities of the Sydney district IV. The significance of species interactions. *Journal of Ecology* 59:535-553.

Clarke P.J. (1993) Dispersal of grey mangrove (*Avicennia marina*) propagules in southeastern Australia. *Aquatic Botany* 45:195-204.

Clarke P.J. & Myerscough P.J. (1993) The intertidal distribution of the grey mangrove (*Avicennia marina*) in southeastern Australia: the effects of physical conditions, interspecific competition, and predation on propagule establishment and survival. *Australian Journal of Ecology* 18:307-315.

Cloern J.E. (2001) Our evolving conceptual model of the coastal eutrophication problem. *Marine Ecology Progress Series* 210:223-253.

Clough B.F. & Attiwill P.M. (1980) Primary productivity of *Zostera muelleri* Irmisch. ex Aschers. in Western Port Bay (Victoria, Australia). *Aquatic Botany* 9:1-13.

Cohen B.F., McArthur M.A. & Parry G.D. (2000) *Exotic marine pests in Westernport*. Marine and Freshwater Resources Institute, Queenscliff.

Coleman F. & Williams S.L. (2002) Overexploitating marine ecosystem engineers: potential consequences for biodiversity. *TREE* 17:40-44.

Coleman M.A., Kelaher B.P., Steinberg P.D. & Millar A.J.K. (2008) Absence of a large brown macroalga on urbanized rocky reefs around Sydney, Australia, and evidence for historical decline. *Journal of Phycology* 44:897-901.

Coleman N. (1976) A survey of the molluscs of Crib Point, Western Port, Victoria. *Journal of the Malacological Society of Australia* 3:239-255.

Coleman N., Cuff W., Drummond M. & Kudenov J.D. (1978) Quantitative survey of macrobenthos of Western Port, Victoria. *Australian Journal of Marine and Freshwater Research* 29:445-466.

Coleman N. & Cuff W. (1980) The abundance, distribution and diversity of the molluscs of Western Port, Victoria, Australia. *Malacologia* 20:35-62.

Coleman N. & Poore G.C. (1980) The distribution of Callianassa species (Crustacea, Decapoda) in Western Port, Victoria. *Proceedings of the Royal Society of Victoria* 91:73-78.

Collins M.I. (1921) On the mangrove and saltmarsh vegetation near Sydney, New South Wales, with special reference to Cabbage Tree Creek, Port Hacking. *Proceedings of the Linnaean Society of New South Wales* 46:376-392.

Connell, S.D. and M.A. Vanderklift (2007) Negative interactions: The influence of predators and herbivores on prey and ecological systems. In: S.D. Connell (ed.) *Marine Ecology*. Oxford University Press.

Connolly R.M. (2009). Seagrass. In: Connolly R.M. (ed.) *A Marine Climate Change Impacts and Adaptation Report Card for Australia 2009*. NCCARF Publication 05/09.

Contessa L. & Bird F.L. (2004) The impact of bait-pumping on populations of the ghost shrimp Trypaea australiensis Dana (Decapoda : Callianassidae) and the sediment environment. *Journal of Experimental Marine Biology and Ecology* 304:75-97.

Cook F.J., Hicks W., Gardner E.A., Carlin G.D. & Froggatt D.W. (2000) Export of acidity in drainage water from acid sulphate soils. *Marine Pollution Bulletin* 41:319-326.

Cook P.L., Revill A.T., Butler E.C. & Eyre B.D. (2004) Carbon and nitrogen cycling on intertidal mudflats of a temperate Australian estuary. II. Nitrogen cycling. *Marine Ecology Progress Series* 280:39-54.

Cook P.L.M., Van Oevelen J., Soetaert K. & Middelburg J.J. (2009) Benthic carbon and nitrogen cycling on intertidal mudflats of a temperate Australian estuary. IV An inverse analysis and synthesis. *Mar. Ecol. Prog. Ser.* 394:35-48.

Corfield J. (2000) The effects of acid sulphate run-off on a subtidal estuarine macrobenthic community in the Richmond River, NSW, Australia. *ICES Journal of Marine Science* 57:1517-1523.

Corrick A.H. (1981) Wetlands of Victoria II. Wetlands and waterbirds of South Gippsland. *Proceedings of the Royal Society of Victoria* 92:187-200.

Counihan R. & Molloy R. (2003) Western Port marine environment conceptual understanding. *Background material for workshop*. Coastal CRC, CSIRO Environmental Projects Office, Clayton.

Counihan R., Molloy R., Scheltinga D., Shaw R., Johnstone R., Arquit S., Lawrence P. & Fearon R. (2003) *Westerport Research Coordination Project stage 1*. Coastal CRC & CSIRO.

Cowling B. (2001) The impact of the invading saltmarsh grass *Spartina anglica* on the benthic macrofauna of a high shore intertidal mudflat. Honours thesis, Monash University.

Crinall S.M. & Hindell J.S. (2004) Assessing the use of saltmarsh flats by fish in a temperate Australian embayment. *Estuaries* 27:728-739.

Crook D.A., Macdonald J.I., O'Connor J.P. & Barry B. (2006) Use of otolith chemistry to examine patterns of diadromy in the Australian grayling *Prototroctes maraena*. *Journal of Fish Biology* 69:1330-1344.

CSIRO (2007) The air pollution model (TAPM). http://www.csiro.au/products/TAPM.html

CSIRO & BoM (2007) Climate change in Australia: technical report 2007. CSIRO and Australian Bureau of Meteorology.

CSIRO (2009) Understanding Climate Change. http://www.csiro.au/resources/Climate-is-changing.html

Cullen J.M., Montague T.L. & Hull C. (1991) Food of Little Penguins *Eudyptula minor* in Victoria: comparison of three localities between 1985 and 1988. *Emu* 91:318-341.

Curnow J. (1998) Status report. *Codium fragile* subsp. *tomentosoides*. Distribution in Western Port Bay. Internal government report. Department of Sustainability and Environment, Melbourne.

Currie D. & Crookes D. (1997) *Exotic Marine Pests in the Port of Hastings, Victoria*. Marine and Freshwater Resources Institute, Queenscliff.

Curtis W.M. & Somerville J. (1974) Boomer marsh – a preliminary botanical and historical survey. *Papers and Proceedings of the Royal Society of Tasmania* 151-157.

Dahllof, I., Blanck, H. & Hall P. (1999a) Short-term effects of tri-N-butyl-tin on marine sediment samples using nutrient fluxes as effect indicators. *Environmental Toxicology and Chemistry* 18(5): 850-857.

Dahllof, I., Blanck, H., Hall, P.O. & Molander, S. (I999b) Long-term effects of tributyltin on the function of a marine sediment. *Marine Ecology Progress Series* 188:1-11.

Dahms H.-U. & Lee J.-S. (2010) UV radiation in marine ectotherms: molecular effects and responses. *Aquatic Toxicology* 97:3-14.

Daly, H. & Fabris G. (1993) An environmental study of tributyltins in Victorian waters. Environment Protection Authority, Victoria.

Dann P. (1979) Food-robbing of Bar-tailed Godwits by Silver Gulls in Western Port Bay. *Corella* 3:84-85.

Dann P. (1987) The feeding behaviour and ecology of shorebirds. Chapter 3. In: Lane, B. (ed.) *Shorebirds of Australia*. Nelson, Melbourne.

Dann P. (1991) Feeding behaviour and diet of Double-banded Plovers *Charadrius bicinctus* in Western Port, Victoria. *Emu* 91:179-184.

Dann P. (1993) Abundance, diet and feeding behaviour of the Whimbrel *Numenius phaeopus variegatus* in Rhyll Inlet, Western Port. *Corella* 17:52-57.

Dann P. (1994) The distribution and abundance of Palearctic and Australasian waders (Charadrii) in coastal Victoria. *Corella* 18:148-154.

Dann P., Loyn R.H. & Bingham P. (1994) Ten years of waterbird counts in Western Port, Victoria. 2. Waders, gulls and terns. The Australian Bird Watcher 15:351-365.

Dann P. (1996) The distribution and abundance of Australian Fur Seals Arctocephalus pusillus and Bottlenose Dolphins Tursiops truncatus in Western Port, Victoria. Victorian Naturalist 113:306-310.

Dann P. (1999a) Feeding periods and supratidal feeding of Red-necked Stints and Curlew Sandpipers in Western Port, Victoria. *Emu* 99:218-222.

Dann P. (1999b) Foraging behaviour and diets of Red-necked Stints and Curlew Sandpipers in south-eastern Australia. *Wildlife Research* 27:61-68.

Dann P. (2000) The ecology of birds frequenting intertidal habitats in Western Port. PhD thesis, University of Melbourne.

Dann P., Jessop R. & Healy M. (2001) The distribution and abundance of Little Penguins at sea in Western Port, Victoria. *Victorian Naturalist* 118:230-235.

Dann P., Arnould J.P.Y., Jessop R. & Healy M. (2003) Distribution and abundance of seabirds in Western Port, Victoria. *Emu* 103:307-313.

Dann P. (2007) The Population Status of the Kelp Gull *Larus dominicanus* in Victoria. *Corella* 31:73-75.

Darragh T.A. (1986) The Cainozoic Trigoniidae of Australia. *Alcheringa: An Australasian Journal of Palaeontology* 10:1-34.

Davey A. & Woelkerling W.J. (1980) Studies on Australian mangrove algae. 1. Victorian communities: composition and geographic distribution. *Proceedings of the Royal Society of Victoria* 91:53-66.

Davey A. & Woelkerling W.J. (1985) Studies on australian mangrove algae .3. Victorian communities - structure and recolonisation in Western Port Bay. *Journal of Experimental Marine Biology and Ecology* 85:177-190.

DCC (2009) *Climate change risks to Australia's coast. A first pass national assessment*. Department of Climate Change, Australian Government.

DECC (2009) Identification guidelines for endangered ecological communities - coastal saltmarsh. Department of Environment and Climate Change http://www.environment.nsw.gov.au/resources/threatenedspecies/coastal_saltmarsh_A3_110108.pdf

de Jonge, V. N. & van Beusekom, J. E. (1995) Wind- and tide-induced resuspension of sediment and microphytobenthos from tidal flats in the Ems estuary. *Limnology and Oceanography* 40:166-778.

Dela-Cruz J., Ajani P., Lee R., Pritchard T. & Suthers I.M. (2002) Temporal abundance patterns of the red tide dinoflagellate *Noctiluca scintillans* along the southeast coast of Australia. *Mar. Ecol. Prog. Se*r. 236:75-88.

Dela-Cruz J., Middleton J.H. & Suthers I.M. (2003) Population growth and transport of the red tide dinoflagellate, *Noctiluca scintillans*, in the coastal waters off Sydney Australia, using cell diameter as a tracer. *Limnology and Oceanography* 48:656-674.

Dennett X. & Loyn R.H. (2009) BOCA Western Port Survey: Long-term Monitoring of Waterbird Numbers. *Victorian Naturalist* 126:99-107.

Dennison W.C. (2009) Seagrasses: biology, ecology and conservation. *Botanica Marina* 52(4):367.

Depledge M.H. & Billinghurst Z. (1999) Ecological significance of endocrine disruption in marine invertebrates. *Marine Pollution Bulletin* 39:32-38.

De Roach R., Rate A., Knott B. & Davies P. (2002) Denitrification activity in sediment surrounding polychaete (*Ceratonereis aequisetis*) burrows. . *Marine and Freshwater Research* 53:35-41.

DNRE (2000) Western Port Shorebird Site. http://www.dse.vic.gov.au/__data/assets/pdf_file/0015/100095/Western_Port_EAAF_FNS_SIS_Map.pdf

DNRE (2002) *Victoria's native vegetation management - a framework for action*. Department of Natural Resources and Environment, Victoria.

Domingues C.M., Church J.A., White N.J., Glecker P.J., Wijffels S.E., Barker P.M. & Dunn J.R. (2008) Improved estimates of upper-ocean warming and multi-decadal sea-level rise. *Nature* 453:1090-1093.

Doney S.C., Fabry V.J., Feely R.A. & Kleypas J.A. (2001) Ocean acidification: the other CO2 problem. *Annual Review of Marine Science* 1:169-192.

DPCP (2011) *Aboriginal culture and heritage. Melbourne Atlas 2006 - Melbourne's People*. Department of Planning and Community Development, Victoria.

DPI (2003) Acid sulphate soil hazards map: Guidelines for coastal Victoria. *Centre for Land Protection Research Report* No 12. Department of Primary Industries, Victoria.

DPI (2004) Declaration of *Caulerpa taxifolia* as a noxious aquatic species. http://new.dpi.vic.gov.au/fisheries/pests-weeds-diseases/noxious-aquatic-species/fn0167-declaration-of-caulerpa-taxifolia-as-a-noxious-aquatic-species.

DPI (2005) Flinders Aquaculture Fisheries Reserve Management Plan. *Fisheries Victoria Management Report* No. 32. Department of Primary Industries, Victoria.

DPI (2008) *Fisheries Victoria Commercial Fish Production Information Bulletin 2008*. Fisheries Victoria, Department of Primary Industries, Queenscliff.

DPI (2010) *Fisheries Victoria Commercial Fish Production Information Bulletin 2010*. Fisheries Victoria, Department of Primary Industries, Queenscliff.

DPI (2011a) Sites of geological and geomorphological significance - Westernport International and State. Victorian Resources Online. Map database. Accessed July 2011. http://www.dpi.vic.gov.au/dpi/vro/map_documents.nsf/pages/pp_sig_wport_int

DPI (2011b) Western Port - Tidal Watershed. No. 110. Victorian Resources Online. Accessed July 2011. http://www.dpi.vic.gov.au/dpi/vro/portregn.nsf/pages/port_lf_sig_sites_watershed

DPI (2011c) Pioneer Bay - Quaternary Stratigraphy. No. 115. Victorian Resources Online. Accessed July 2011. http://www.dpi.vic.gov.au/dpi/vro/portregn.nsf/pages/port_lf_sig_sites_115

DPI (2011d) Sandstone Island - Silurian outcrop. No. 48. Victorian Resources Online. Accessed July 2011. http://www.land.vic.gov.au/dpi/vro/portregn.nsf/pages/port_lf_sig_sites_sandstone

DPI (2011e) Reef Island and Kennedy Point. No. 141. Victorian Resources Online. Accessed July 2011. http://www.dpi.vic.gov.au/dpi/vro/portregn.nsf/pages/port_lf_sig_sites_reef

DPI (2011f) Tortoise Head. No. 210. Victorian Resources Online. Accessed July 2011. http://www.dpi.vic.gov.au/dpi/vro/portregn.nsf/pages/port_lf_sig_sites_tortoise

DPI (2011g) San Remo Peninsula - North Shore. No. 168. Victorian Resources Online. Accessed July 2011. http://www.dpi.vic.gov.au/dpi/vro/portregn.nsf/pages/port_lf_sig_sites_san_remo

DPI (2011h) Crawfish Rock. No 222. Victorian Resources Online. Accessed July 2011. http://www.dpi.vic.gov.au/dpi/vro/portregn.nsf/pages/port_lf_sig_sites_222

Dromgoole F.I. (1975) Occurrence of Codium fragile subspecies tomentosoides in New Zealand waters. *New Zealand Journal of Marine and Freshwater Research* 9:257-264.

DSE (2003) *Western Port Ramsar site: strategic management plan*. Parks Victoria report for the Department of Sustainability and Environment, Melbourne.

DSE (2003a). San Remo Marine Community. Flora and Fauna Guarantee Action Statement. Department of Sustainability and Environment, Melbourne.

DSE (2005) Melbourne 2030. Planning for sustainable growth. Department of Sustainability and Environment. http://www.dse.vic.gov.au/melbourne2030online/

DSE (2007) Models of the Catchment Analysis Tool (CAT1D version 22). Technical manual. Department of Sustainability and Environment, Melbourne.

DSE (2008a) Marine Biology Existing Conditions and Impact Assessment. *Victorian Desalination Project Environmental Effects Statement*. Technical Appendix 31. Department of Sustainbility and Environment, Melbourne.

DSE (2008b) Environmental effects of marine structures. *Victorian Desalination Project Environmental Effects Statement*.Vol. 2. Department of Sustainability and Environment, Melbourne.

DSE (2009a) *Wetland ecological vegetation classes for the Index of Wetland Condition*. Department of Sustainability and Environment, Melbourne.

DSE (2009b) Securing our natural future - Victoria's land and biodiversity white paper. Department of Sustainability and Environment, Melbourne.

DSE (2010). Seagrass and reefs program for Port Phillip Bay. Department of Sustainability and Environment, Melbourne. http://www.dse.vic.gov.au/coasts-and-marine/marine/seagrass-and-reefs-program-for-port-phillip-bay.

DSE & DPCD (2010) Future Coasts program. Department of Sustainability and Environment and Department of Planning and Community Development, Melbourne. http://www.climatechange.vic.gov.au/adapting-to-climate-change/future-coasts

Duarte C.M. (1990) Seagrass nutrient content. *Marine Ecology Progress Series* 67:201-207.

Duarte C.M., Borum J., Short F.T. & Walker D.I. (2008). Seagrass ecosystems: Their Global status and prospects. In: Duarte C.M., Borum J., Short F.T. & Walker D.I. (eds.) *Aquatic Ecosystems: Trends and Global Prospects*. Cambridge University Press, pp. 281-294.

Duke N. (1990) Phenological trends with latitude in the mangrove tree *Avicennia marina*. *Journal of Ecology* 78:113-133.

Duke N.C. (2006) Australia's Mangroves. *The Authoritative Guide to Australia's mangrove plants*. University of Queensland, Brisbane.

Duke N.C. (2008) Corrections and updates to the article by Duke et al. (2005) reporting on the unusual occurrence and cause of dieback of the common mangrove species, *Avicennia marina*, in NE Australia. *Marine Pollution Bulletin* 56:1668-1670.

16 References

Dunn R.K., Welsh D.T., Jordan M.A., Teasdale P.R. & Lemckert C.J. (2009) Influence of natural amphipod (*Victoriopisa australiensis*) (Chilton, 1923) population densities on benthic metabolism, nutrient fluxes, denitrification and DNRA in sub-tropical estuarine sediment. *Hydrobiologia* 628: 95-109.

East L.R. (1953) Swamp reclamation in Victoria *The Journal of the Institution of Engineers Australia* 7:77-91.

ECC (2000) *Marine Coastal and Estuarine Investigation. Final Report.* Environment Conservation Council.

Edgar G.J., Hammond L.S. & Watson G.F. (1993) *Consequences for commercial fisheries of loss of seagrass beds in southern Australia*. Final report to Fisheries Research Development Corporation Committee on Project 88/91. Victorian Institute of Marine Science.

Edgar G.J., Shaw C., Watson G.F. & Hammond L.S. (1994) Comparisons of species richness, size-structure and production of benthos in vegetated and unvegetated habitats in Western-port, Victoria. *Journal of Experimental Marine Biology and Ecology* 176:201-226.

Edgar G.J. & Shaw C. (1995a) The production and trophic ecology of shallow-water fish assemblages in southern Australia I. Species richness, size-structure and production of fishes in Western Port, Victoria. *Journal of Experimental Marine Biology and Ecology* 194:53-81.

Edgar G.J. & Shaw C. (1995b) The production and trophic ecology of shallow-water fish assemblages in southern Australia. II. Diets of fishes and trophic relationships between fishes and benthos at Western Port, Victoria. *Journal of Experimental Marine Biology and Ecology* 194:83-106.

Edgar G.J. & Shaw C. (1995c) The production and trophic ecology of shallow-water fish assemblages in southern Australia.III. General relationships between sediments, seagrasses, invertebrates and fishes. *Journal of Experimental Marine Biology and Ecology* 194:107-131.

Edmunds M., Roob R. and Blake S. (1999) Monitoring of Reef Biota at Phillip Island – Marine Performance Assessment Program, September 1999. *Unpublished report for Victorian Natural Resources and Environment, Australian Marine Ecology Report* No. 101, Melbourne.

Edmunds M., Hart S., Elias J. & Power B. (2004) Victorian intertidal reef monitoring program: the reef biota in central Victoria and Port Phillip Bay Marine Sanctuaries. *Parks Victoria Technical Series* No. 11. Parks Victoria, Melbourne.

Edmunds M., Mustoe S., Stewart K., Sheedy E. & Ong J. (2010) VNPA Nature Conservation Review: Marine Conservation Priorities and Issues for Victoria. *Report to Victorian National Parks Association* Report No. 405, v 10.

Eggleston D.B., Elis W.E., Etherington L.L., Dahlgren P. & Posey M.H. (1999) Organism responses to habitat fragmentation and diversity: habitat colonization by estuarine macrofauna. *Journal of Experimental Marine Biology and Ecology* 236:107-132.

Emison W.B., Beardsell, C. M., Norman F.I., Loyn R.H. & Bennett S.C. (1987) *Atlas of Victorian Birds*. Department of Conservation, Forests and Lands and RAOU, Melbourne.

Environment Australia (2004) TBT analysis protocol development and current contamination assessment. Department of Environment and Heritage, Canberra.

Environment Link & Vehicle Research and Design (2007) *Comparative assessment of the environmental performance of small engines: marine outboards and personal watercraft.* Prepared for Department of Environment and Water Resources, Canberra.

EPA (1995) T*he distribution of seagrass in Western Port, Victoria*. EPA Publication No. 490. Environment Protection Authority, Victoria.

EPA (1996) *The Western Port Marine Environment*. EPA Publication No. 493. Environment Protection Authority, Victoria.

EPA (1998a) *Protecting the Waters of Western Port and Catchment. Community Workshop Notes*. Environment Protection Authority, Victoria

EPA (1998b) *Cleaner Marinas - EPA Guidelines for Protecting Victoria's Marinas*. Environment Protection Authority, Victoria.

EPA (1999) *The Western Port catchment – pollution types, sources and remedies (draft)*. A report prepared for the Port Phillip Catchment Management Authority. Environment Protection Authority, Melbourne.

EPA (2001) *Water quality objectives for marine and estuarine waters - ecosystem protection*. Policy background paper - draft State Environment Protection Policy (Waters of Victoria). Environment Protection Authority, Victoria.

EPA (2002) *Western Port water quality: long-term trends in nutrient status and clarity, 1984-2001*. Environment Protection Authority, Victoria.

EPA (2004) Appendix 7A: A historical perspective of TBT contamination provided by the Victorian EPA. In: *TBT analysis protocol development and current contamination assessment*. Department of Environment and Heritage, Canberra.

EPA (2006) *Western Port beach water quality summer 2005-2006*. Environment Protection Authority, Victoria.

EPA (2008) *Port Phillip Bay and Western Port Marine Condition Report*. EPA report for the development of the Better Bays and Waterways Program.

EPA (2009) *Acid sulfate soil and rock*. Information bulletin. Publication 655.1. Environment Protection Authority, Victoria.

EPA (2011) *Western Port Condition Report - 2009*. Publication 1371. Environment Protection Authority, Victoria.

EPA (Qld) (2008) Coastal wetlands of south-east Queensland - survey and assessment. http://www.epa.qld.gov.au/wetlandinfo/site/mappingFandD/wetlandmapsanddata/mapsurveyandconsassess.html

Evans P.R. (1975). Exploratory investigation of feeding ecology and behaviour of shorebirds in Westernport Bay. In: Evans P.R. (ed.) *Westernport Bay Environmental Study, 1973-1974*. Ministry for Conservation, Melbourne, pp. 502-510.

eWATER (2010). http://www.ewater.com.au/products/ewater-source/for-catchments/

Eyre B.D. & Maher D. (2011) Mapping ecosystem processes and function across shallow seascapes. *Continental Shelf Researc*h 31:S162-A172.

Eyre B.D., Ferguson A.J.P., Webb A., Maher D. & Oakes J.M. (2011) Denitrification, N-fixation and nitrogen and phosphorus fluxes in different benthic habitats and their contribution to the nitrogen and phosphorus budgets of a shallow oligotrophic sub-tropical coastal system (southern Moreton Bay, Australia). *Biogeochemistry* 102:111-113.

Fabris G.J. & Longmore A.R. (2005) *An examination of key metals and their speciation and potential ecosystem risk to Victorian bays, inlets and estuaries in relation to the new ANZECC guidelines: arsenic*. Report to the Environment Protection Authority, Victoria.

Fabry V.J., Seibel B.A., Feely R.A. & Orr J.C. (2008) Impacts of ocean acidification on marine fauna and ecosystem processes. *Ices Journal of Marine Science* 65:414-432.

Fancett M.S. & Kimmerer W.J. (1985) Vertical migration of the demersal copepod Pseudodiaptomus as a means of predator avoidance. *Journal of Experimental Marine Biology and Ecology* 88:31-43.

Fancett M.S. & Jenkins G.P. (1988) Predatory impact of scyphomedusae on ichthyoplankton and other zooplankton in Port Phillip Bay. *Journal of Experimental Marine Biology and Ecology* 116:63-77.

Farrell M.J. & Ashton D.H. (1974) *Environmental factors affecting the growth and establishment of mangroves in Westernport Bay*. Unpublished report. University of Melbourne.

Ferns L.W. & Curnow J.G. (1998) *Codium fragile* spp. *tomentosoides* incursion at Western Port: the implementation of a Victorian protocol for managing exotic marine organism incursions: a case study. Department of Natural Resources and Environment, Melbourne.

Fisher P.M.J. & Scott R. (2008) Evaluating and controlling pharmaceutical emissions from dairy farms: a critical first step in developing a preventative management approach. *Journal of Cleaner Production* 16:1437-1446.

Floerl O., Inglis G.J., Dey K. & Smith A. (2009) The importance of transport hubs in stepping-stone invasions. *Journal of Applied Ecology* 46:37-45.

Forster S. & Graf G. (1995) Impact of irrigation on oxygen flux into the sediment: intermittent pumping by *Callianassa subterranea* and 'piston-pumping' by Lanice conchilega. *Marine Biology* 123:335-3469.

Fulton E.A. & Smith A.D.M. (2004) Lessons learnt from a comparison of three ecosystem models for Port Phillip Bay, Australia. *African Journal of Marine Science* 26:219-243.

Fulton E.A. (2010) Approaches to end-to-end ecosystem models. *Journal of Marine Systems* 81:171-183.

Gaughan D.J., Mitchell D.W. & Blight S.J. (2000) Impact of mortality, possibly due to herpesvirus, on pilchard *Sardinops sardax* stocks along the south coast of Western Australia in 1998-99. *Marine and Freshwater Research* 51:601-612.

Gavine F., Ingram B. & Doroudi M. (2009) Development of management strategies for herpes-like virus infection in abalone. Fisheries Research and Development Corporation. FRDC Report No. 2006/243.

Ghent M.L. (2004) The saltmarshes of the southern coast of Victoria: floristic composition, variation and distribution. Masters thesis, School of Botany, University of Melbourne.

Gilmour A.J. (1969) The ecology of the King George whiting *Sillaginodes punctatus* (Cuvier and Valenciennes) in Westernport bay, Victoria. PhD thesis, Monash University.

Gilmour P., Edmunds M., Lindsay M. & Monk J. (2006) Victorian Subtidal Reef Monitoring Program: The reef biota at Phillip Island. *Parks Victoria Technical Series* No. 30. Parks Victoria, Melbourne.

Gilmour P. & Edmunds M. (2007) Victorian intertidal reef monitoring program: the intertidal reef biota of central Victoria's marine protected areas (Volume 2). P*arks Victoria Technical Series* No. 46. Parks Victoria, Melbourne.

Glavinic A. (2010) Systematics, Phylogeny, Phylogeography and Reproduction of Neotrigonia (Bivalvia: Palaeoheterodonta). PhD thesis, Flinders University.

Goudey R. (2007) Do statistical inferences allowing three alternative decisions give a better feedback for environmentally precautionary decision making? *Journal of Environmental Management* 85:338-344.

Gray M.W., Wu R.S. & Or Y.Y. (2002) Effects of hypoxia and organic enrichment on the coastal marine environment *Marine Ecology Progress Series* 238:249-279.

Gray J.S. & Elliot M. (2009) *Ecology of Marine Sediments*. Oxford University Press.

Gullan P.K. (2008) Victorian ecosystems. Coastal saltmarsh. Internet resource. http://www.viridans.com/ECOVEG/saltmarsh.htm

Gunthorpe L. & Hamer P. (eds.) (1998) Western Port fish habitat 1998, Victoria. Assessment Report Series 22. Compiled by the Fish Habitat Assessment Group. Fisheries Victoria, Queenscliff.

Häder D.-P., Helbling E.W., Williamson C.E. & Worrest R.C. (2011) Effects of UV radiation on aquatic ecosystems and interactions with climate change. *Photochemical & Photobiological Sciences* 10:242-260.

Hallegraeff G., Beardall J., Brett S., Doblin M., Hosja W., de Salas M. & Thompson P. (2009) Phytoplankton. NCCARF.

Halpern B.S., Walbridge S., Selkoe K.A., Kappel C.V., Micheli F., D'Agrosa C., Bruno J.F., Casey K.S., Ebert C., Fox H.E., Fujita R., Heinemann D., Lenihan H., Madin E.M.P., Perry M.T., Selig E.R., Spalding M., Steneck R. & Watson R. (2008) A global map of human impact on marine ecosystems. *Science* 319:948-952.

Hamer P.A. & Jenkins G.P. (2004) High levels of spatial and temporal recruitment variability in the temperate sparid Pagrus auratus. *Marine and Freshwater Research* 55:663-673.

Hamilton A.A. (1919) An ecological study of the saltmarsh vegetation in the Port Jackson district. *Proceedings of the Linnaean Society of New South Wales* 44:463-513.

Hamilton B. (2001) The ecology and spread of Spartina in Western Port Bay. Honours thesis, University of Melbourne.

Hancock G., Olley J.M. & Wallbrink P.J. (2001) Sediment transport and accumulation in Western Port. *Report on Phase 1 of a study determining the sources of sediment to Western Port*. Technical report no 47/01. CSIRO Land and Water - Environmental Hydrology.

Hansen B., Menkhorst P. & Loyn R. (In preparation) *Western Port Welcomes Waterbirds: Improving our understanding of waterbird use of Western Port*. Arthur Rylah Institute for Environmental Research Technical Report Series, Department of Sustainability and Environment.

Harbison P. (1986a) Mangrove muds - a sink and a source for trace metals. *Marine Pollution Bulletin* 17:246-250.

Harbison P. (1986b) Diurnal variations in the chemical environment of a shallow tidal inlet, Gulf St. Vincent, South Australia: Implications for water quality and trace metal migration. *Marine Environmental Research* 20:161-195.

Harris J.E., Hinwood J.B., Marsden M.A.H. & Sternberg R.W. (1979) Water movements, sediment transport and deposition, Western Port, Victoria. *Marine Geology* 30:131-161.

Harris J.E. & Robinson J.B. (1979) Circulation in Western Port, Victoria, as deduced from salinity and reactive-silica distributions. *Marine Geology* 30:101-116.

Harris M.P. & Bode K.G. (1981) Populations of Little Penguins, Short-tailed Shearwaters and other seabirds on Phillip Island, Victoria, 1978. *Emu* 81:20-28.

Harris G., Bately G., Fox D., Hall D., Jernakoff P., Molloy R., Murray A., Newell B., Parslow J., Skyring G. & Walker S. (1996) *Port Phillip Bay Environmental Study. Final Report*. CSIRO.

Harris G. (1999) The response of Australian estuaries and coastal embayments to increased nutrient loadings and changes in hydrology. In: Smith S. & Crossland C. (eds.) *Australian estuarine systems: carbon, nitrogen and phosphorus fluxes*. LOICZ reports and studies No. 12. Texel, The Netherlands, pp. 112-124.

Harrison S., Black K., Bosserelle C. & Lee, R. (2011a) *Port Phillip and Western Port receiving water quality modelling: hydrodynamics*. ASR technical report 2007-EPA1. Published as EPA publication No. 1377.

Harrison S., Black K. & Bosserelle C. (2011b) *Port Phillip and Western Port receiving water quality modelling: Lagrangian dispersal*. ASR technical reportt 2007-EPA2. Published as EPA publication No. 1378.

Harty C. (2008) Mangrove planning and management in New Zealand and South East Australia - a reflection on approaches. *Ocean and Coastal Management* 52:278-286.

Harvell C.D., Mitchell C.E., Ward J.R., Altizer S., Dobson A.P., Ostfeld R.S. & Samuel M.D. (2002) Climate Warming and Disease Risks for Terrestrial and Marine Biota. *Science* 296:2158-2162.

Harvey A.S. & Bird F.L. (2008) Community structure of a rhodolith bed from cold-temperate waters (southern Australia). *Australian Journal of Botany* 56:437-450.

Havenhand J.N., Buttler F.R., Thorndyke M.C. & Williamson J.E. (2008) Near-future levels of ocean acidification reduce fertilization success in a sea urchin. *Current Biology* 18:R651-R652.

Hayes K., Sliwa C., Migus S., McEnnulty F. & Dunstan P. (2005) National priority pests – Part II. Ranking of Australian marine pests. CSIRO Marine Research. Report for Department of Environment and Heritage, Canberra.

Heip C. (1995) Eutrophication and zoobenthos dynamics. *Ophelia* 41:113-136.

Heislers A., Loyn R.H. & Julian M. (2003) *Wings over Western port: three decades surveying wetland birds* 1973-2003. Bird Observers Club of Victoria (BOCA).

Heislers S. & Parry D. (2007) Species diversity and composition of benthic infaunal communities found in Marine National Parks along the outer Victorian coast. *Parks Victoria Technical Paper Series* No. 53. Parks Victoria, Melbourne.

Hendreck C.D. & O'Hara T.D. (1994) *Occurrence of selected species of intertidal and shallow subtidal invertebrates at Victorian locations*. Marine Research Group of Victoria report to the Land Conservation Council.

Hewitt C. & Campbell M. (2008) *Assessment of relative contribution of vectors to the introduction and translocation of marine invasive species*. University of Tasmania report for the Department of Agriculture, Fisheries and Forestry.

Hewitt C.L., Campbell M.L., Thresher R.E., Martin R.B., Boyd S., Cohen B.F., Currie D.R., Gomon M.F., Keough M.J., Lewis J.A., Lockett M.M., Mays N., McArthur M.A., O'Hara T.D., Poore G.C.B., Ross D.J., Storey M.J., Watson J.E. & Wilson R.S. (2004) Introduced and cryptogenic species in Port Philip Bay, Victoria, Australia. *Marine Biology* 144:183-202.

Hewitt J.E. & Thrush S.F. (2009) Reconciling the influence of global climate phenomena on macrofaunal temporal dynamics at a variety of spatial scales. *Global Change Biology* 15:1911-1929.

Hindell J., Jenkins G.P. & Keough M.J. (2000a) Variability in the abundances of fish associated with seagrass habitats in relation to the diets of predatory fishes. *Marine Biology* 136:725-737.

Hindell J., Jenkins G.P. & Keough M.J. (2000b) Evaluating the impact of predation by fish on the assemblage structure of small fishes associated with seagrass, *Heterozostera tasmanica* (Martens ex Ascherson) den Hartog, and unvegetated sand habitats. *Journal of Experimental Marine Biology and Ecology* 255:153-174.

Hindell J.S., Jenkins G.P. & Keough M.J. (2002) Variability in the numbers of post-settlement King George whiting (Sillaginidae: *Sillaginodes punctata*, Cuvier) in relation to predation, habitat complexity and artificial cage structure. *Journal of Experimental Marine Biology and Ecology* 268:13-31.

Hindell J.S. & Jenkins G.P. (2004) Spatial and temporal variability in the assemblage structure of fishes associated with mangroves (*Avicennia marina*) and intertidal mudflats in temperate Australian embayments. *Marine Biology* 144:385-395.

Hindell J.S., Jenkins G.P., Connolly R.M. & Hyndes G. (2004) Assessment of the importance of different near-shore habitats to important fishery species in Victoria using standardised survey methods, and in temperate and sub-tropical Australia using stable isotope analysis. *Final Report to Fisheries Research and Development Corporation, Project No. 2001/036*.

Hindell J.S. & Jenkins G.P. (2005) Assessing patterns of fish zonation in temperate mangroves, with emphasis on evaluating sampling artefacts. *Marine Ecology Progress Series* 290:193-205.

Hindell J.S. (2006) Assessing the trophic link between seagrass habitats and piscivorous fishes. *Marine and Freshwater Research* 57:121-131.

Hindell J.S., Jenkins G.P. & Womersley B. (2008) Habitat utilisation and movement of black bream A*canthopagrus butcheri* (Sparidae) in an Australian estuary. *Marine Ecology Progress Series* 366:219-229.

Hinwood J.B. & O'Brien W.T. (1974a) *Water Quality Mathematical Model. Final Report. Volume 1: Theory*. Report for the Westernport Bay Environmental Study, Ministry for Conservation, Victoria.

Hinwood J.B. & O'Brien W.T. (1974b) *Water Quality Mathematical Model. Final Report. Volume 2: Data*. Report for the Westernport Bay Environmental Study, Ministry for Conservation, Victoria.

Hinwood J.B. (1979) Hydrodynamic and transport models of Western Port, Victoria. *Marine Geology* 30:117-130.

Hinwood J.B. & Jones J.C.E. (1979) Hydrodynamic data for Western Port, Victoria. *Marine Geology* 30:47-63.

16 References

Hobday A.J., Okey T.A., Poloczanska E.S., Kunz T.J. & Richardson A.J. (2006) Impacts of Climate Change on Australian Marine Life Part A: Executive Summary. CSIRO Marine and Atmospheric Research report to the Australian Greenhouse Office, Department of the Environment and Heritage, Canberra.

Hoedt F.E. & Dimmlich W.F. (1994) Diet of subadult Australian salmon, *Arripis truttaceus*, in Western Port, Victoria. *Australian Journal of Marine and Freshwater Research* 45:617-623.

Hoedt F.E. & Dimmlich W.F. (1995) Egg and larval abundance and spawning localities of the anchovy (*Engraulis australia*) and pilchard (*Sardinops neopilchardus*) near Phillip Island, Victoria. *Marine and Freshwater Research* 46:735-743.

Hoedt F.E., Dimmlich W.F. & Dann P. (1995) Seasonal variation in the species and size composition of the clupeoid assemblages in Western Port, Victoria. *Marine and Freshwater Research* 46:1085-1091.

Holloway M.G. & Keough M.J. (2002) Effects if an introduced polychaete, Sabella spallanzanii, on the development of epifaunal assemblages. *Marine Ecology Progress Series* 236:137-154.

Howard R.K. (1984) The trophic ecology of caridean shrimps in an eelgrass community. *Aquatic Botany* 18:155-174.

Howard R.K. & Lowe K.W. (1984) Predation by birds as a factor influencing the demography of an intertidal shrimp. *Journal of Experimental Marine Biology and Ecology* 74:35-52.

Howard R.K. & Koehn J.D. (1985) Population dynamics and feeding ecology of pipefish (Syngnathidae) associated with eelgrass beds of Western Port, Victoria. *Australian Journal of Marine and Freshwater Research*36:361-370.

Howarth R., Chan F., Conley D.J., Garnier J., Doney S.C., Marino R. & Billen G. (2011) Coupled biogeochemical cycles: eutrophication and hypoxia in temperate estuaries and coastal marine ecosystems. *Frontiers in Ecology and Environment* 9:18-26.

Hudson R.J. & Walker T.I. (1998) Distribution of the jellyfish *Catostylus mosaicus* Port Phillip Bay and Western Port during April and May 1998. Department Natural Resources and Environment, Victoria.

Hughes A.O., Prosser I.P., Wallbrink P.J. & Stevenson J. (2003) *Suspended sediments and bedload budgets for the western port Bay Basin*. Technical Report 4/03, CSIRO Land and Water.

Hurley P.J., Physick W.L. & Luhar A.K. (2005) TAPM: a practical approach to prognostic meteorological and air pollution modelling. *Environmental Modelling and Software* 20:737-752.

Hutchinson N., Hunt T. & Morris L. (2010) Seagrass and Reef Program for Port Phillip Bay: Temperate Reefs Literature Review. Fisheries Victoria, Department of Primary Industries, Queenscliff.

Huttel M. (1990) Influence of the lugworm Arenicola marina on porewater nutrient profiles of sandflat sediments. *Marine Ecology Progress Series* 62: 241-248.

HydroTechnology (1993) Groundwater nutrient and toxicant inputs to Port Phillip Bay. Port Phillip Bay Environmental Study Technical Report No. 13.

IEG (2008) *Advice on the Desalination Project Environmental Effects Statement*. Independent Experts Group report to Department of Planning and Community Development, Victoria.

IMO (2010) International Convention on the Control of Harmful Anti-fouling Systems on Ships. International Maritime Organisation. http://www.imo.org

IPCC (2007a) *Climate Change 2007: Synthesis Report. Contribution of Working Groups I, II and III to the Fourth Assessment*. Vol AR4. Intergovernmental Panel on Climate Change, Geneva.

IPCC (2007b) Summary for Policymakers. *In: Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change*. Intergovernmental Panel on Climate Change, Geneva.

IUCN (2010) *IUCN Red List of Threatened Species.* Version 2010.4. International Union for the Conservation of Nature, Gland, Switzerland. http://www.iucnredlist.org

Jackson J.B.C. (2008) Ecological extinction and evolution in the brave new ocean. *Proceedings of the National Academy of Sciences* 105:11458-11465.

Jacobs S.W.L. & Les D.H. (2009) New combinations in *Zostera* (Zosteracea). *Telopea* 12:419-423.

Jenkins G.P., Edgar G.J., May H.M.A. & Shaw C. (1993). Ecological basis for parallel declines in seagrass habitat and catches of commercial fish in Western Port Bay, Victoria. In: Jenkins G.P., Edgar G.J., May H.M.A. & Shaw C. (eds.) *Sustainable fisheries through sustaining fish habitat. Australian Society for Fish Biology Workshop, Victor Harbour, SA, 12-13 August, Bureau of Resource Sciences Proceedings*. AGPS, Canberra, pp. 124-136.

Jenkins G.P., May H.M.A., Wheatley M.J. & Holloway M.G. (1997) Comparison of fish assemblages associated with seagrass and adjacent unvegetated habitats of Port Phillip Bay and Corner Inlet, Victoria, Australia, with emphasis on commercial species. *Estuarine, Coastal and Shelf Science* 44:569-588.

Jenkins G.P., Keough M.J. & Hamer P.A. (1998) The contributions of habitat structure and larval supply to broad-scale recruitment variability in a temperate zone, seagrass-associated fish. *Journal of Experimental Marine Biology and Ecology*226:259-278.

Jenkins G.P., Black K.P. & Keough M.J. (1999) The role of passive transport and the influence of vertical migration on the pre-settlement distribution of a temperate, demersal fish: numerical model predictions compared with field sampling. *Marine Ecology Progress Series* 184:259-271.

Jenkins G.P., Black K.P. & Hamer P.A. (2000) Determination of spawning areas and larval advection pathways for King George whiting in southeastern Australia using otolith microstructure and hydrodynamic modelling. I. Victoria. *Marine Ecology Progress Series* 199:231-242.

Jenkins G.P. (2005) The influence of climate on the fishery recruitment of a temperate, seagrass associated fish, the King George whiting, Sillaginodes punctata. *Marine Ecology Progress Series* 288:263-271.

Jenkins G.P. & Hatton D. (2005) *Pop-net sampling to determine whether pale mangrove goby, Mugilogobius paludis, occurs in Corner Inlet/Nooramunga, Victoria*. Internal Report No. 52. Primary Industries Research, Victoria.

Jenkins G.P. & McKinnon L. (2006) *Channel Deepening Supplementary Environment Effects Statement - Aquaculture and Fisheries*. Primary Industries Research, Victoria.

Jenkins G.P. (2007) *Channel Deepening Supplementary Environment Effects Statement - Aquaculture and Fisheries: Supplementary report on entrance channel depth*. Primary Industries Research, Victoria.

Jenkins G.P. (2010) *Fisheries Adaptation to Climate Change – Marine Biophysical Assessment of Key Species*. Fisheries Victoria, Queenscliff.

Jenkins G.P., Conron S.D. & Morison A.K. (2010) Highly variable recruitment in an estuarine fish is determined by salinity stratification and freshwater flow: implications of a changing climate. *Marine Ecology Progress Series* 417:249-261.

Jensen A. & Healey M. (2003) Frog communities and wetland condition: relationships with grazing by domestic livestock along an Australian floodplain river. *Biological Conservation* 109:207-219.

Jones J.B., Hyatt A.D., Hine P.M., Whittington R.J., Griffin D.A. & Bax N.J. (1997) Australasian pilchard mortalities. *World Journal of Microbiology and Biotechnology* 13:383.

Jones M. & Pinn E. (2006) The impact of macroalgal mat on benthic biodiversity in Poole Harbour. *Marine Pollution Bulletin* 53:63-71.

Joye S.B., De Beer D. & Cook P.L.M. (2009). Biogeochemical dynamics of coastal tidal flats. In: Joye S.B., De Beer D. & Cook P.L.M. (eds.) *Coastal Wetlands: an Ecosystem Integrated Approach*. Elsevier, pp. 345-374.

Jung C.A., Swearer S.E. & Jenkins G.P. (2010) Changes in diversity in the fish assemblage of a southern Australian embayment: consistent spatial structuring at decadal scales. *Marine and Freshwater Research* 61:1425-1434.

Kelaher B.P., Chapman M.G. & Underwood A.J. (1998a) Changes in benthic assemblages near boardwalks in temperate urban mangrove forests. *Journal of Experimental Marine Biology and Ecology*228:291-307.

Kelaher B.P., Underwood A.J. & Chapman M.G. (1998b) Effect of boardwalks on the semaphore crab *Heloecius cordiformis* in temperate urban mangrove forests. *Journal of Experimental Marine Biology and Ecology* 227:281-300.

Kelleway J., Williams R.J. & Laegdsgaard P. (2009). Mapping, assessment and monitoring of saltmarshes. In: Kelleway J., Williams R.J. & Laegdsgaard P. (eds.) *Australian Saltmarsh Ecology*. CSIRO, Melbourne, pp. 211-229.

Kemp W.M., Boynton W.R., Adolf J.E., Boesch D.F., Boicourt W.C., Brush G., and others (2005) Eutrophication of Chesapeake Bay: historical trends and ecological interactions. *Marine Ecology Progress Series* 303:1-29.

Keough M. J. & and Ross D. J. (1999) Introduced fouling species in Port Phillip Bay. In: Hewitt C. L., Campbell M. L., Thresher R. E., and Martin R. B. (eds.). Marine Biological Invasions of Port Phillip Bay, Victoria. CSIRO Marine Research, Hobart. pp. 193-226.

Keough M.J. & Quinn G.P. (2000) Legislative vs practical protection of an intertidal shoreline: changes associated with altered coastal access in southeastern Australia. *Ecological Applications* 10:871-881.

Keough M.J. & Swearer S.E. (2007). Early life histories of marine invertebrates and fishes. In: Keough M.J. & Swearer S.E. (eds.) *Marine Ecology*. Oxford University Press, Melbourne, pp. 17-46.

Kimmerer W.J. & McKinnon A.D. (1985) A comparative study of the zooplankton in two adjacent embayments, Port Phillip and Westernport Bays, Australia. *Estuarine, Coastal and Shelf Science* 21:145-159.

Kimmerer W.J. & McKinnon A.D. (1987a) Growth, mortality, and secondary production of the copepod *Acartia tranteri* in Westernport Bay, Australia. *Limnology and Oceanography* 32:14-28.

Kimmerer W.J. & McKinnon A.D. (1987b) Zooplankton in a marine bay. I. Horizontal distributions used to estimate net population growth rates. *Marine Ecology Progress Series* 41:43-52.

Kimmerer W.J. & McKinnon A.D. (1987c) Zooplankton in a marine bay. II. Vertical migration to maintain horizontal distributions. *Marine Ecology Progress Series* 41:53-60.

Kimmerer W.J. & McKinnon A.D. (1989) Zooplankton in a marine bay. III. Evidence for influence of vertebrate predation on distributions of two common copepods. *Marine Ecology Progress Series* 53:21-35.

King A. (1992) Human activity and its effects on marine intertidal plant and animal populations: monitoring and management. MSc thesis, University of Melbourne.

King C.K., Dowse M.C., Simpson S.L. & Jolley D.F. (2004) An assessment of five Australian polychaetes and bivalves for use in whole-sediment toxicity tests: toxicity and accumulation of copper and zinc from water and sediment. *Archives of Environmental Contamination and Toxicology* 47:314-323.

King C.K., Simpson S.L., Smith S.V., Stauber J.L. & Batley G.E. (2005) Short-term accumulation of Cd and Cu from water, sediment and algae by the amphipod *Melita plumulosa* and the bivalve *Tellina deltoidalis*. *Marine Ecology Progress Series* 287:177-188.

King C.K., Gale S.A., Hyne R.V., Stauber J.L., Simpson S.L. & Hickey C.W. (2006) Sensitivities of Australian and New Zealand amphipods to copper and zinc in waters and metal-spiked sediments. *Chemosphere* 63:1466-1476.

King R.J., Adam P. & Kuo J. (1990). Seagrass, mangroves and saltmarsh plants. In: King R.J., Adam P. & Kuo J. (eds.) *Biology of marine plants*. Longman Cheshire, Melbourne.

Kinrade P. & Justus M. (2008) *Impacts of climate change on settlements in the Western Port region. Climate change risks and adaptation*. Western Port Greenhouse Alliance, Department of Sustainability and Environment, Victoria.

Kirkwood R., Hibble J. & Jerret I. (1999) An Australian Sea Lion on Phillip Island, Victoria. *Victorian Naturalist* 119:98-101.

Kirkwood R., Gales N., Lynch M. & Dann P. (2002) Satellite tracker deployments on adult, male Australian fur seals: methods and preliminary results. *Australian Mammalogy* 24:73-84.

Kirkwood R., Gales R., Terauds A., Arnould J.P.Y., Pemberton D., Shaughnessy P.D., Mitchell A.T. & Gibbens J. (2005) Pup production and population trends of the Australian fur seal *Arctocephalus pusillus doriferus*. *Marine Mammal Science* 21:260-282.

Kirkwood R., Lynch M., Gales N., Dann P. & Sumner M. (2006) At sea movements and habitat use of adult male Australian fur seals (*Arctocephalus pusillus doriferus*). *Canadian Journal of Zoology* 84:1781-1788.

Kirkwood R., Hume F. & M. H. (2008) Sea temperature variations mediate annual changes in the diet of Australian fur seals in Bass Strait. *Marine Ecology Progress Series* 369:297-309.

Kirkwood R., Pemberton D., Gales R., Hoskins A., Mitchell T., Shaughnessy P.D. & Arnould J.P.Y. (2010) Continued population recovery by Australian fur seals. *Marine and Freshwater Research* 61: 695-701.

Klumpp D.W. & Nichols P.D. (1983) Nutrition of the southern sea garfish *Hyporhamphus melanochir*: gut passage rate and daily consumption of two food types and assimilation of seagrass components. *Marine Ecology Progress Series* 12:207-16.

Knutzen J. (1981) Effects of decreased pH on marine organisms. *Marine Pollution Bulletin* 12:25-29.

Kookana R., Williams M. & Natasha Waller N. (2007) *Endocrine-disrupting chemicals in the Australian environment - Background and current research in Australia*. CSIRO - Land and Water, Canberra.

Koster W. & Dawson D. (2010) *Investigation of Australian Grayling spawning in the Yarra and Bunyip rivers*. Arthur Rylah Institute for Environmental Research. Department of Sustainability and Environment.

Kott P. (1976) The ascidian fauna of Western Port, Victoria, and a comparison with that of Port Phillip Bay. *Memoirs of the National Museum of Victoria* 37:53-96.

Kristensen E. & Blackburn T.H. (1987) The fate of organic carbon and nitrogen in experimental marine sediment systems: influence of bioturbation and anoxia. *Journal of Marine Research* 45:231-257.

Krauss K.W., Lovelock C.E., McKee K.L., Lopez-Hoffman L., Ewe S.M.L. & Sousa W.P. (2008) Environmental drivers in mangrove establishment and early development: a review. *Aquatic Botany* 89:105-127.

Kriwoken L.K. & Hedge P. (2000) Exotic species and estuaries: managing *Spartina anglica* in Tasmania, Australia. *Ocean and Coastal Management* 43:573-584.

Kuiper J., De Wilde P.A.W.J. & Wolff W. (1984) Effects of an oil spill in outdoor model tidal flat ecosystems. *Marine Pollution Bulletin* 15:102-106.

Laegdsgaard P. (2006) Ecology, disturbance and restoration of coastal saltmarsh in Australia: a review. *Wetlands Ecology and Management* 14:379-399.

Lafferty K.D., Porter J.W. & Ford S.E. (2004) Are diseases increasing in the ocean? *Annual Review of Ecology, Evolution and Systematics* 35:31-54.

Lane B. (1987) *Shorebirds in Australia*. Nelson, Melbourne.

Langdon C. & Atkinson M.J. (2005) Effect of elevated pCO2 on photosynthesis and calcification of corals and interactions with seasonal change in temperature/irradiance and nutrient enrichment. *Journal of Geophysical Research* 110:C09S07.

Larkum A.W.D., Orth R.J. & Duarte C.M. (2006) *Seagrasses: biology, ecology and conservation*. Springer, Dordecht, The Netherlands.

Last P.R., White W.T., Gledhill D.C., Hobday A.J., Brown R., Edgar G.J. & Pecl G. (2011) Long-term shifts in abundance and distribution of a temperate fish fauna: A response to climate change and fishing practices *Global Ecology and Biogeography* 20:58-72.

Lawler W. (1994) *Draft management manual for migratory shorebird feeding habitat in New South Wales estuaries*. New South Wales National Parks & Wildlife Service, Sydney.

Lee S.Y., Dunn R.J.K., Young R.A., Connolly R.M., Dale P.E.R., Dehayr R., Lemckert C.J., Powell B., Teasdale P.R. & Welsh D.T. (2006) Impact of urbanization on coastal wetland structure and function. *Austral Ecology* 31:149-163.

Lee R., Black K. & Oldman J. (2011) *Port Phillip Bay and Western Port Receiving Water Quality Modelling: Scenarios*. Joint EPA and ASR Report. Report No. 2008–EPA1. Published as EPA publication 1380.

Ling S.D. (2008) Range expansion of a habitat-modifying species leads to loss of taxonomic diversity: a new and impoverished reef state. *Oecologia* 156:883-894.

Link J.S., Fulton E.A. & Gamble R.J. (2010) The northeast US application of ATLANTIS: A full system model exploring marine ecosystem dynamics in a living marine resource management context. *Progress in Oceanography* 87:214-234.

Loebl M., van Beuskon J.E.E. & Reise K. (2006) Is spread of the neophyte *Spartina anglica* recently enhanced by increasing temperatures? *Aquatic Botany* 40:315-324.

Lohrer A.M., Thrush S.F., Hewitt J.E., Berkenbusch K., Ahrens M. & Cummings V.J. (2004) Terrestrially-derived sediment: response of marine macrobenthic communities to thin terrigeneous deposits. *Marine Ecology Progress Series* 273:121-138.

Longmore A.R. (1997) *Analysis of water quality in Western Port, 1973-97 in relation to protection of beneficial uses*. Internal report No. 4. Marine and Freshwater Resources Institute, Queenscliff.

Longmore A., Nicholson G., Lourey M., Cowdell R. & Abbott B. (1998) *Estimation of Benthic Nutrient Fluxes in Western Port - A Pilot Study*. Marine and Freshwater Institute.

Longmore A.R., Nicholson G.J. & Abbott B. (2002) *Identifying habitats important to commercial fish in Western Port*. Marine and Freshwater Resources Institute, Queenscliff.

Longmore A.R. & Nicholson G. (2008) *Nutrient Cycling (Denitrification) Monitoring Program*. Fisheries Victoria, Department of Primary Industries, Queenscliff.

Love L.D. (1981) *Mangrove swamps and salt marshes*. Cambridge University Press, Cambridge.

Lowe K.W. (1982a) The habitat requirements of Herons, Egrets, Ibises and Spoonbills of the Westernport region. *Environmental Studies Series* No. 347. Ministry for Conservation, Victoria.

Lowe K.W. (1982b) Feeding-behavior and diet of Royal Spoonbills P*latalea regia* in Westernport Bay, Victoria. *Emu* 82:163-168.

Lowe K.W. (1984) Feeding behaviour and diet of the White-faced Heron (*Ardea novaehollandiae*) in Westernport Bay, Victoria. *Corella* 7:101-109.

Loyn R. (1975) *Report on the avifauna of Westernport Bay: report to the Westernport Bay Environmental Study*. Ministry for Conservation, Melbourne.

Loyn R.H. (1978) A survey of birds in Westernport Bay, Victoria. *Emu* 78:11-19.

Loyn R.H., Dann P. & Bingham P. (1994) Ten years of waterbird counts in Western Port, Victoria, 1973-83. I: Waterfowl and large wading birds. *The Australian Bird Watcher* 15:333-350.

Loyn R.H., Dann P. & McCulloch E. (2001) Important Wader Sites in the East Asian- Australasian Flyway: 1. Western Port, Victoria, Australia. *The Stilt* 38:39-53.

16 References

Luhar A.K., Mitchell R.M., Meyer C.P., Qin Y., Campbell S., Gras J.L. & Parry D. (2008) Biomass burning emissions over northern Australia constrained by aerosol measurements: II-Model validation and impacts on air quality and radiative forcing. *Atmospheric Environment* 42:1647-1664.

MacDonald C.M. (1992). Fluctuations in seagrass habitats and commercial fish catches in Westernport Bay and the Gippsland Lakes, Victoria. In: MacDonald C.M. (ed.) *Recruitment Processes. Australian Society for Fish Biology Workshop, Hobart, 21 August 1991, Bureau of Rural Resources Proceedings No. 16*. AGPS, Canberra, pp. 192-201.

Macnae W. (1966) Mangroves in eastern and southern Australia. *Australian Journal of Botany* 14:67-104.

Macreadie P.I., Hindell J.S., Jenkins G.P., Connolly R.M. & Keough M.J. (2009) Fish Responses to Experimental Fragmentation of Seagrass Habitat. *Conservation Biology* 23:644-652.

Macreadie P.I., Hindell J.S., Keough M.J., Jenkins G.P. & Connolly R.M. (2010) Resource distribution influences positive edge effects in a seagrass fish. *Ecology* 91:2013-2021.

Macreadie V.J. (1972) Studies on the zooplankton of Westernport bay with special reference to the Copepoda. Honours Zoology, Monash University.

Marine Science and Ecology (2009) Marine Ecological Monitoring for Cold Strip Mill 1973-2009. North Arm Channel, Western Port, Victoria.

Marsden M.A.H. & Mallett C.W. (1974) Morphology and sediment distribution, Westernport bay. *Environmental Studies Series* No. 68. Ministry for Conservation, Victoria.

Marsden M.A.H. & Mallet C.W. (1975) Quaternary evolution, morphology and sediment distribution, Westernport Bay, Victoria. P*roceedings of the Royal Society of Victoria* 87:107-138.

Marsden M.A.H. & Sternberg R.W. (1976) Seabed drifter studies of bottom water movement, Westernport bay and inner Bass Strait. *Environmental Studies Series* No. 116. Ministry for Conservation, Victoria.

Marsden M.A.H. (1979a) Circulation patterns from seabed-drifter studies, Western Port and inner Bass Strait, Australia. *Marine Geology* 30:85-99.

Marsden M.A.H. (1979b) Sedimentary processes and water circulation in Western-port, Victoria, Australia - contribution to a major environmental study. Preface. *Marine Geology* 30:VI-VII.

Marsden M.A.H., Mallett C.W. & Donaldson A.K. (1979) Geological and physical setting, sediments and environments, Western Port, Victoria. *Marine Geology* 30:11-46.

Mazumder D., Saintilan N. & Williams R.J. (2006) Trophic relationships between itinerant fish and crab larvae in a temperate Australian saltmarsh. *Marine and Freshwater Research* 57:193-199.

McInnes K.L. & Hubbert G.D. (2003) A numerical modelling study of storm surges in Bass Strait. *Australian Meteorological Magazine* 52:143-156.

McInnes K.L., Abbs D.J. & Bathols J.A. (2005) Climate change in Eastern Victoria Stage 1 Report: The effect of climate change on coastal wind and weather patterns. CSIRO Consultancy Report for the Gippsland Coastal Board.

McInnes K.L., Macadam I. & O'Grady J.G. (2007) T*he effect of climate change on extreme sea levels in the Westernport region*. A project undertaken for the Australian Greenhouse Office and Department of Sustainability and Environment, Victoria. CSIRO Marine and Atmospheric Research, Melbourne.

McInnes K.L., Macadam I., Hubbert G.D. & O'Grady J.G. (2009a) A modelling approach for estimating the frequency of sea level extremes and the impact of climate change in southeast Australia. *Natural Hazards* 51:115-137.

McInnes K.L., Macadam I. & O'Grady J. (2009b) *The effect of climate change on extreme sea levels along Victoria's coast*. A project undertaken for the Department of Sustainability and Environment, Victoria as part of 'Future Coasts'. CSIRO Marine and Atmospheric Research, Melbourne.

McInnes K.L., O'Grady J.G. & Hubbert G.D. (2009c) Modelling sea level extremes from storm surges and wave setup for climate change assessments in southeastern Australia. *Journal of Coastal Research* S156:1005-1009.

McKinlay R., Plant J.A., Bell J.N.B. & Voulvoulis N. (2008) Endocrine disrupting pesticides: implications for risk assessment. *Environment International* 34:168-183.

McKinnon A.D., Kimmerer W.J. & Benzie J.A.H. (1992) Sympatric sibling species within the genus *Acartia* (Copepoda: Calanoida): a case study from Westernport and Port Phillip Bays, Australia. *Journal of Crustacean Biology* 12:239-259.

McKinnon L.J., Leporati S.C., Parry G.D. & Blake S. (2004) *Environmental characterisation of the Flinders Aquaculture Fisheries Reserve in Western Port, Victoria*. Department of Primary Industries, Queenscliff.

McNeil B.I. & Matear R.J. (2008) Southern Ocean acidification: a tipping point at 450-ppm atmospheric CO2. *Proceedings of the National Academy of Sciences of the United States of America* 105:18860-18864.

Meador J.P., Varanasi U. & Krone C.A. (1993) Differential sensitivity of marine infaunal amphipods to tributyltin. *Marine Biology* 116:231-239.

Melbourne Water (2009) *Port Phillip and Westernport DSS Decision Support System 'PortsE2' Final Report & Water Quality Loads Monitoring Program (WQMP)*. Melbourne Water, Melbourne.

Melbourne Water & EPA (2009) *Better Bays and Waterways: A Water Quality Improvement Plan for Port Phillip Bay and Western Port*. Melbourne Water and Environment Protection Authority, Victoria.

Melbourne Water (2010) Planning for sea level rise. Interim guidelines. Assessing development in areas prone to tidal inundation from sea level rise in the Port Phillip and Westernport Region. Melbourne Water.

Menkhorst P.W. (ed.) (1995) *Mammals of Victoria: distribution, ecology and conservation*. Oxford University Press, Melbourne.

Meyrick and Associates (2007) Economic Impact Study: Port of Hastings. Final Report. Meyrick and Associates in association with Econsearch. Prepared for Port of Hastings Corporation and the Victorian Department of Infrastructure.

Miles H., Widdicombe S., Spicer J.I. & Hall-Spencer J. (2007) Effects of anthropogenic seawater acidification on acid-base balance in the sea urchin *Psammechinus miliaris*. *Marine Pollution Bulletin* 54:89-96.

Miller C.J., Campbell S.J. & Scudds S. (2005) Spatial variation of Zostera tasmanica morphology and structure across an environmental gradient. *Marine Ecology Progress Series* 304:45-53.

Miller D.C., Muir C.L. & Hauser O.A. (2002) Detrimental effects of sedimentation on marine benthos: what can be learned from natural processes and rates? *Ecological Engineering* 19:211-232.

Moller L., Bilgmann K., Charlton-Robb K. & Beheregaray L. (2008). Multi-gene evidence for a new bottlenose dolphin species in southern Australia. *Molecular Phylogenetics and Evolution* 49, 674-681.

Montague T.L., Cullen J.M. & Fitzherbert K. (1986) The diet of the Short-tailed Shearwater *Puffinus tenuirostris* during its breeding season. *Emu* 86.

Montani S., Pithakpol S. & Tada K. (1998) Nutrient regeneration in coastal seas by *Noctiluca scintillans*, a red tide-causing dinoflagellate. *Journal of Marine Biotechnology* 6:224-228.

Morgan G.J. (1986) A survey of macrobenthos in the waters of Corner Inlet and the Nooramunga, southern Victoria, with an assessment of the extent of Posidonia seagrass. *Fisheries and Wildlife Paper* 31:1-49.

Morris L. & Ball D. (2006) Habitat suitability modelling of economically important fish species with commercial fisheries data. Ices J*ournal of Marine Science* 63:1590-1603.

Morris L. & Keough M.J. (2002) Organic pollution and its effects: a short-term transplant experiment to assess the ability of biological endpoints to detect change in a soft sediment environment. *Marine Ecology Progress Series* 225:109-121.

Morris L. & Keough M.J. (2003) Variation in the response of intertidal infaunal invertebrates to nutrient additions: field manipulations at two sites within Port Phillip Bay, Australia. *Marine Ecology Progress Series* 250:35-49.

Morris L., Jenkins G., Hatton D. & Smith T. (2007) Effects of nutrient additions on intertidal seagrass (*Zostera muelleri*) habitat in Western Port, Victoria, Australia. *Marine and Freshwater Research* 58:666-674.

Morrisey D. (1995). Saltmarshes. In: Morrisey D. (ed.) *Coastal marine ecology of temperate Australia*. University of New South Wales Press, Sydney, pp. 205-220.

Morrisey D., Swales A., Dittmann S., Morrison M.A., Lovelock C.E. & Beard C.M. (2010) The ecology and management of temperate mangroves. *Oceanography and Marine Biology: an annual review* 48:43-160.

Morrisey D.J., Skilleter G.A., Ellis J.I., Burns B.R., Kemp C.E. & Burt K. (2003) Differences in benthic fauna and sediment among mangrove (*Avicennia marina var. australasica*) stands of different ages in New Zealand. *Estuarine, Coastal and Shelf Science* 56:581-592.

Morton B. (1987) The Functional Morphology of *Neotrigonia margaritacea* (Bivalvia: Trigoniacea), with a Discussion of Phylogenetic Affinities. *Records of the Australian Museum* 39:339-354.

Murphy H. & Jenkins G.P. (2009) *Observational methods used in marine spatial monitoring*. Department of Primary Industries, Victoria.

Murray A.G. & Parslow J.S. (1999) Modelling of nutrient impacts in Port Phillip Bay - a semi-enclosed marine Australian ecosystem. *Marine and Freshwater Research* 50:597-611.

Nateekanjanalarp S. (1997) Ecological studies of barnacles in temperate mangrove forests. PhD thesis, University of Melbourne.

National Academy of Science (2010) *Ocean acidification: A National Strategy to Meet the Challenges of a Changing Ocean*. National Academies Press, Washington DC.

Nebel S., Porter J.L. & Kingsford R.T. (2008) Long-term trends of shorebird populations in eastern Australia and impacts of freshwater extraction. *Biological Conservation* 141:971-980.

Newell R.C., Seiderer L.J. & Hitchcock D.R. (1998) The impact of dredging works in coastal waters: A review of the sensitivity to disturbance and subsequent recovery of biological resources on the sea bed. *Oceangraphy and Marine Biology: an Annual Review* 36:127-178.

Nias D.J., McKillup S.C. & Edyvane K.S. (1993) Imposex in *Lepsiella vinosa* from Southern Australia. *Marine Pollution Bulletin* 26:380-384.

Nicholson G., Jenkins G.P., Sherwood J. & Longmore A. (2008) Physical environmental conditions, spawning and early-life stages of an estuarine fish: climate change implications for recruitment in intermittently open estuaries. *Marine and Freshwater Research* 59:735-749.

Nixon S. W., Ammerman J. W., Atkinson L. P., Berounsky V. M., Billen G., Boicourt W. C., Boynton W. R., Church T. M., DiToro D. M., Elmgren R., Garber, J. H., Giblin A. E., Jahnke R. A., Owens N. J. P., Pilson M. E. Q. and Seitzinger S. P. (1996) The fate of nitrogen and phosphorus at the land-sea margin of the North Atlantic Ocean. *Biogeochemistry* 35:141-180.

Nizzolia D., Bartolia M., Cooper M., Welsh T.D., Underwood G.J. & Viarolia P. (2007) Implications for oxygen, nutrient fluxes and denitrification rates during the early stage of sediment colonisation by the polychaete Nereis spp. in four estuaries. *Estuarine, Coastal and Shelf Science* 75:125-134.

Nordhausen W. (1988) Impact of the nemertean *Lineus viridis* on its polychaete prey on an intertidal sandflat *Hydrobiologia* 156:39-46.

Norkko A., Thrush S.F., Hewitt J.E., Cummings V.J., Norkko J., Ellis J.I., Funnell G.A., Schultz D. & MacDonald I. (2002) Smothering of estuarine sandflats by terrigenuous clay: the role of wind-wave disturbance and bioturbation in site-dependent macrofaunal recovery. *Marine Ecology Progress Series* 234:23-41.

Norman F.I. & Gottsch M.D. (1969) The Phillip Island colonies of the Tasmanian Muttonbird, with species reference to those at Cape Woolamai. *Emu* 69:137-144.

NRS (1974) Mapping of the seagrass and macrophytic algal communities of Westernport Bay. Natural Systems Research Pty Ltd. Project Report for Westernport Bay Environmental Study (1973-1974). Ministry for Conservation, Melbourne.

Nybakken J.W. (2001) *Marine biology: an ecological approach* Benjamin Cummings, San Francisco.

O'Hara T.D. & Barmby V. (2000) *Victorian Marine Species of Conservation Concern: Molluscs, Echinoderms and Decapod Crustaceans*. Department of Natural Resources and Environment, Victoria.

O'Hara T.D., Norman M.D. & Staples D.A. (2002) Baseline monitoring of *Posidonia* seagrass beds in Corner Inlet, Victoria. *Museum Victoria Science Reports* 1:1-44.

O'Hara T.D., Addison P.F.E., Gazzard R., Costa T.L. & Pocklington J.B. (2010) A rapid biodiversity assessment methodology tested on rocky shores. *Aquatic Conservation Marine and Freshwater Ecosystems* 20:452-463.

Occhipinti-Ambrogi A. & Savini D. (2003) Biological invasions as a component of global change in stressed marine ecosystems. *Marine Pollution Bulletin*.

Orbell J., Ngeh L., Bigger S., Zabinskas M., Zheng M., Healy M., Jessop R. & Dann P. (2004) Whole-Bird Models for the Magnetic Cleansing of Oiled Feathers. *Marine Pollution Bulletin* 48:336-340.

Orbell J.D., Van Dao H., Kapadia J., Ngeh L.N., Bigger S.W., Healy M., Jessop R. & Dann P. (2007) An investigation into the removal of oil from rock utilising magnetic particle technology. *Marine Pollution Bulletin* 54:1958-1961.

Orchard A.E. (ed.) (1999) *Flora and Fauna of Australia: Introduction*. Department of Environment and Heritage/CSIRO, Canberra/Melbourne.

Ormerod S.J., Dobson M., Hildrew A.G. & Townsend C.R. (2010) Multiple stressors in freshwater ecosystems. *Freshwater Biology* 55:1-4.

Orr J.C., Fabry V.J., Aumont O., Bopp L., Doney S.C., Feely R.A., Gnanadesikan A., Gruber N., Ishida A., Joos F., Key R.M., Lindsay K., Maier-Reimer E., Matear R., Monfray P., Mouchet A., Najjar R.G., Plattner G.-K., Rodgers K.B., Sabine C.L., Sarmiento J.L., Schlitzer R., Slater R.D., Totterdell I.J., Weirig M.-F., Yamanaka Y. & Yool A. (2005) Anthropogenic ocean acidification over the twenty-first century and its impact on calcifying organisms. *Nature* 437:681-686.

Orth R.J., Luckenbach M. & Moore K.A. (1994) Seed dispersal in a marine macrophyte: implications for colonisation and restoration. *Ecology* 75:1927-1939.

Otto C.J. (1992) *Literature and information review of the groundwater input of nutrients and toxicants to Port Phillip Bay*. Department of Natural Resources and Environment and CSIRO Division of Water Resources.

OzCoasts (2010a) Coastal indicators - metal contaminants. Australian online coastal information. http://www.ozcoasts.org.au/indicators/metal_contaminants.jsp

OzCoasts (2010b) Coastal indicators – imposex frequency. Australian online coastal information. http://www.ozcoasts.org.au/indicators/imposex_frequency.jsp

OzCoasts (2010c) Climate change - sea level rise maps. Australian online coastal information.http://www.ozcoasts.org.au/climate/Map_images/Melbourne/mapLevel2_East.jsp

Palacios S. L. & Zimmerman R. C. (2007) Response of eelgrass Zostera marina to CO2 enrichment: possible impacts of climate change and potential for remediation of coastal habitats. Marine Ecology Progress Series 344: 1-13.

Parks Victoria (2005) Safety and Environment Management Plan for the local port of Western Port. Parks Victoria, Melbourne.

Parks Victoria (2007a) Yaringa Marine National Park, French Island Marine National Park and Churchill Island Marine National Park Management Plan. Parks Victoria, Melbourne.

Parks Victoria (2007b) *Identification of threats to natural values in Victoria's Marine National Parks and Marine Sanctuaries*. Technical Series Publication 33. Parks Victoria, Melbourne.

Parks Victoria (2007c) Mushroom Reef Marine Sanctuary Management Plan. Parks Victoria, Melbourne.

Parry G. (1977) *Life-History Strategies of Five Species of Intertidal Limpet*. PhD Thesis, University of Melbourne.

Parry G. (1982) Reproductive effort in four species of intertidal limpets. *Marine Biology* 67: 267-282.

Parry G. & Cohen B.F. (2001) *Exotic species established in Western Port, including an assessment of the status of the exotic species Corbula gibba, Alexandrium spp, Gymnodinium spp and Undaria pinnatifida*. Report No. 45. Marine and Freshwater Resources Institute, Queenscliff.

Parry G. (2007) *The feasibility of restoration of seagrass from seeds*. Primary Industries and Research Victoria.

Patton R.T. (1942) Ecological Studies in Victoria- Part VI. Salt marsh. *Proceedings of the Royal Society of Victoria* 54:131-144.

Pauly D., Alder J., Bennett E., Christensen V., Tyedmers P. & Watson R. (2003) The future for fisheries. *Science* 302:1359-1361.

Pearson T.H. & Rosenberg R. (1978) Macrobenthic succession in relation to organic enrichment and pollution of the marine environment *Oceanography Marine Biology Annual Review* 16:229-311.

Pelegri S.P., Nielsen L.P. & Blackburn T.H. (1994) Denitrification in estuarine sediment stimulated by the irrigation of the amphipod Corophium volutator. *Marine Ecology Progress Series* 105:285-290.

Pelegri S.P. & Blackburn T.H. (1995) Effect of bioturbation by Nereis sp., Mya arenaria and *Cerastoderm*a sp. on nitrification and denitrification in estuarine sediments. *Ophelia* 42:289-299.

Peter J. (1995) Short-tailed Shearwaters at Tortoise Head. *Australasian Seabird Group Newsletter* 28:2-3.

Peterson C.H., Rice S.D., Short J.W., Esler D., Bodkin J.L., Ballachey B.E. & Irons D.B. (2003) Long-term ecosystem response to the Exxon Valdex oil spill. *Science* 302:2082-2086.

Phillips D.J.H. (1976) The common mussel *Mytilus edulis* as an indicator of pollution by zinc, cadmium, lead and copper. II. Relationship of metals in the mussel to those discharged by industry. *Marine Biology* 38:71–80.

Pidgeon I.M. (1940) The ecology of the central coast of New South Wales. III. Types of primary succession. Proceedings of the Linnaean Society of New South Wales 65:221-249.

Plattell M.E. & Freewater P. (2009) Importance of saltmarsh to fish species of a large south-eastern Australian estuary during a spring tide cycle. Marine and Freshwater Research 60:936-941.

Poore G.C.B., Rainer S.F., Spies R.B. & Ward E. (1975) The Zoobenthos Program in Port Phillip Bay, 1969-1973. *Fisheries and Wildlife Technical Paper* 7:1-78.

Poore G.C.B. & Rainer S. (1979) A three-year study of benthos of muddy environments in Port Phillip Bay, Victoria. *Estuarine and Coastal Marine Science* 9:477-497.

Poore G.C.B. (1982) Benthic communities of the Gippsland Lakes, Victoria. *Australian Journal of Marine and Freshwater Research* 33:901-915.

16 References

Port of Melbourne Authority (1992) *Victorian coastal vulnerability study*. Port of Melbourne Authority, Coastal Investigations Unit.

Przeslawski R., Davis A.R. & Benkendorff K. (2004) Effects of ultraviolet radiation and visible light on the development of encapsulated molluscan embryos. *Marine Ecology Progress Series* 268:151-160.

Przeslawski R. (2005) Combined effects of solar radiation and desiccation on the mortality and development of encapsulated embryos of rocky shore gastropods. *Marine Ecology Progress Series* 298:169-177.

Quinn G. (1985) *The ecology of two populations of the intertidal pulmonate limpet, Siphonaria diemenensis Quoy and Gaimard*, 1833. PhD thesis, University of Melbourne.

Quinn G. (1988) Ecology of the intertidal pulmonate limpet Siphonaria diemenensis (Quoy et Gaimard). II. Reproductive patterns and energetics. *Journal of Experimental Marine Biology and Ecology* 117:137-156.

Quinn D. & Lacey G. (1999) *Birds of French Island Wetlands*. Spectrum Publications, Richmond, Victoria.

Raadik T.A. & Hindell J.S. (2008) Documenting the distribution, biology and ecology of Victoria's only threatened estuarine fish species. Arthur Rylah Institute. Report for Melbourne Water, Victoria.

Raffaelli D. (1999) Nutrient enrichment and trophic organisation in an estuarine food web. *Acta Oecologia* 20:449-461.

Rahmstorf S. (2007) A semi-empirical approach to projecting future sea-level rise. *Science* 315:368–370.

Rahmstorf S. (2010) A new view on sea level rise. Has the IPCC underestimated the risk of sea level rise? *Nature Reports Climate Change* 4:44-45.

Ramskov T. & Forbes V.E. (2008) Life history and population dynamics of the opportunistic polychaete *Capitella* sp. I in relation to sediment organic matter. *Marine Ecology Progress Series* 369:181-1192.

Raulings E., Morris K., Roache M. & Boon P.I. (2010) The importance of water regimes operating at small spatial scales for the diversity and structure of wetland vegetation. *Freshwater Biology* 55:701-715.

Raven J. (ed) (2005) *Ocean acidification due to increasing atmospheric carbon dioxide*. Policy document 12/05. The Royal Society of London.

Rees C.M., Fabris G.J. & Sporcic M.I. (1998) *Toxicants in Western Port Sediments*. Marine and Freshwater Resources Institute report to EPA Victoria.

Reish D.J., Oshida P.S., Mearns A.J., Ginn T.C. & Buchman M. (1999) Effects of pollution on marine organisms. *Water Environment Research* 71:1100-1115.

Renwick L. & Kirkwood R. (2004) An extended visit by a leopard seal to Phillip Island, Victoria. *Victorian Naturalist* 121:55-59.

Richardson A.J. & Gibbons M.J. (2008) Are jellyfish increasing in response to ocean acidification? *Limnology and Oceanography* 53:2040-2045.

Richardson A.J., Bakun A., Hays G.C. & Gibbons M.J. (2009) The jellyfish joyride: causes, consequences and management responses to a more gelatinous future. *TREE* 24:312-322.

Richardson J.R. (1981) Brachiopods from carbonate sands of the Australian shelf. *Proceedings of the Royal Society of Victoria* 99:37-50.

Richardson J.R. (1997). Brachiopods (Phylum Brachiopoda). In: Shepherd, S.A. Davies, M. (eds.) *Marine Invertebrates of Southern Australia Part III*. Handbooks Committee of the South Australian Government, Adelaide, pp. 999-1027.

Risgaard-Petersen N., Dalsgaard T., Rysgaard S., Christensen P.B., Borum J., McGlathery K.J. & Nielsen L.P. (1998) Nitrogen balance of a temperate eelgrass *Zostera marina* bed. *Marine Ecology Progress Series* 174:281-291.

Roach A.C. & Wilson S.P. (2009) Ecological impacts of tributyltin on estuarine communities in the Hastings River, NSW Australia. *Marine Pollution Bulletin* 58:1780-1786.

Roberts D. (1985) *From Swampland to Farmland - A History of the Koo-Wee-Rup Flood Protection District*. Rural Water Commission of Victoria, Armadale.

Roberts D.A., Johnston E.L. & Knott N.A. (2010) Impacts of desalination plant discharges on the marine environment: A critical review of published studies *Water Research* 44:5117-5128

Robertson A.I. (1977) Ecology of juvenile King George whiting Sillaginodes punctatus (Cuvier and Valenciennes) (Pisces: Perciformes) in Western Port, Victoria. *Australian Journal of Marine and Freshwater Research* 28:35-43.

Robertson A.I. (1978) Trophic interactions among the macrofauna of an eelgrass community. PhD Thesis. University of Melbourne, Victoria.

Robertson A.I. & Howard R.K. (1978) Diel trophic interactions between vertically-migrating zooplankton and their fish predators in an eelgrass community. *Marine Biology* 48:207-213.

Robertson A.I. (1980) The structure and organization of an eelgrass fish fauna. *Oecologia* 47:76-82.

Robertson A.I. (1982) Population dynamics and feeding ecology of juvenile Australian salmon (Arripis trutta) in Western Port, Victoria. *Australian Journal of Marine and Freshwater Research* 33:369-375.

Robertson A.I. & Klumpp D.W. (1983) Feeding habits of the southern Australian garfish Hyporhamphus melanochir: a diurnal herbivore and nocturnal carnivore. *Marine Biology* 10:197-201.

Robertson A.I. (1984) Trophic interactions between the fish fauna and macrobenthos of an eelgrass community in Western Port, Victoria. *Aquatic Botany* 18:135-153.

Robinson J.B. & Harris J.E. (1974) *Preliminary study of the hydrochemistry of Westernport Bay*. Ministry for Conservation, Victoria.

Rogers K., Saintilan N. & Cahoon D. (2005a) Surface elevation dynamics in a regenerating mangrove forest at Homebush Bay, Australia. *Wetlands Ecology and Management* 13:587-598.

Rogers K., Saintilan N. & Heijnis H. (2005b) Mangrove encroachment of salt marsh in Western Port Bay, Victoria: The role of sedimentation, subsidence, and sea level rise. *Estuaries and Coasts* 28:551-559.

Rogers K., Wilton K.M. & Saintilan N. (2006) Vegetation change and surface elevation dynamics in estuarine wetlands of southeast Australia. *Estuarine, Coastal and Shelf Science* 66:559-569.

Rogers K. & Saintilan N. (2008) Relationships between surface elevation and groundwater in mangrove forests of southeast Australia. *Journal of Coastal Research* 24:63-69.

Rosenberg R. & Möllera P. (1979) Salinity stratified benthic macrofaunal communities and long-term monitoring along the west coast of Sweden. *Journal of Experimental Marine Biology and Ecology* 37:175-203.

Rosengren N.J. (1984) Sites of Geological and Geomorphological significance in the Westernport Bay catchment. *Environmental Studies Series* No. 401. Ministry for Conservation, Victoria.

Ross D.J., Johnson C.R., Hewitt C.L., Ruiz G.M. (2004) Interaction and impacts of two introduced species on a soft-sediment marine assemblage in SE Tasmania. *Marine Biology* 144: 747-756

Ross P.M. (2006) Macrofauna loss and microhabitat destruction: The impact of trampling in a temperate mangrove forest, NSW, Australia. *Wetlands Ecology and Management* 14:167-184.

Ross D.J., Keough M.J., Longmore A.R. & Knott N.A. (2007) Impacts of two introduced suspension feeders in Port Phillip Bay, Australia. *Marine Ecology Progress Series* 340:41-53.

Ross R. (2000) *Mangroves and salt marshes in Westernport Bay, Victoria*. Arthur Rylah Institute. Report for Department of Natural Resources and Environment., Victoria.

Rotherham D., Underwood A.J., Chapman M.G. & Gray C.A. (2007) A strategy for developing scientific sampling tools for fishery-independent surveys of estuarine fish in New South Wales, Australia. *ICES Journal of Marine Science* 64:1512-1516.

Ruiz J.M., Bryan G.W. & Gibbs P.E. (1994) Chronic toxicity of water tributylin (TBT) and copper to spat of the bivalve Scrobicularia plana: ecological implications *Marine Ecology Progress Series* 113:105-117.

Russell B.D. & Connell S.D. (2005) A novel interaction between nutrients and grazers alters relative dominance of marine habitats. *Marine Ecology Progress Series* 289:5-11.

Ryan K.L., Morison A.K. & Conron S. (2009) *Evaluating methods of obtaining total catch estimates for individual Victorian bay and inlet recreational fisheries*.

Saenger P., Specht M.M., Specht R.L. & Chapman V.J. (1977). Mangal and coastal salt-marsh communities in Australasia. In: Saenger P., Specht M.M., Specht R.L. & Chapman V.J. (eds.) *Wet Coastal Ecosystems*. Elsevier, Amsterdam, pp. 293-345.

Saenger P. & Snedaker S.C. (1993) Pantropical trends in mangrove above-ground biomass and annual litterfall. *Oecologia* 96:293-299.

Saintilan N. & Williams R.J. (2000) Short note: the decline of saltmarsh in southeast Australia: results of recent surveys. *Wetlands* (Australia) 18:49-59.

Saintilan N. (2009a). Distribution of Australian saltmarsh plants. In: Saintilan N. (ed.) *Australian saltmarsh ecology*. CSIRO Publishing, Melbourne, pp. 23-52.

Saintilan N. (2009b) Biogeography of Australian saltmarsh plants. *Austral Ecology* 34:929-937.

Saintilan N. (ed.) (2009c) *Australian saltmarsh ecology* CSIRO Publishing, Melbourne.

Saintilan N., Rogers K. & McKee K.L. (2009). Salt marsh-mangrove interactions in Australasia and the Americas. In: Saintilan N., Rogers K. & McKee K.L. (eds.) Coastal wetlands. *An integrated ecosystem approach*. Elsevier, Amsterdam, pp. 855-883.

Sargeant I. (1977) Westernport regional environmental study: a review of the extent and environmental effects of erosion in the Westernport catchment. *Environmental Studies Series* No. 174. Ministry for Conservation, Victoria.

Satumanatpan S. & Keough M.J. (1999) Effect of barnacles on the survival and growth of temperate mangrove seedlings. *Marine Ecology Progress Series* 181:189-199.

Satumanatpan S., Keough M.J. & Watson G.F. (1999) Role of settlement in determining the distribution and abundance of barnacles in a temperate mangrove forest. *Journal of Experimental Marine Biology and Ecology* 241:45-66.

Satumanatpan S. & Keough M.J. (2001) Role of larval supply and larval behavior in determining patterns of settlement of barnacles in a temperate mangrove forest. *Journal of Experimental Marine Biology and Ecology* 260:133-153.

Schaffelke B. & Klumpp D.W. (1998) Nutrient-limited growth of the coral reef macroalga Sargassum baccularia and experimental growth enhancement by nutrient addition in continuous flow culture. *Marine Ecology Progress Series* 164:199-211.

Schiel D.R. & Hickford M.J. (2001) Biological structure of nearshore rocky subtidal habitats in southern New Zealand. *Science for Conservation* 182: 54p.

Shapiro M.A. (ed.) (1975a) Westernport Bay Environmental Study, 1973 -1974. *Environmental Study Series* No. 502. Ministry for Conservation, Victoria.

Shapiro M.A. (ed.) (1975b) *A preliminary report on the Westernport Bay environmental study : report for the period 1973-1974 (abridged version)*. Ministry for Conservation, Victoria.

Sharpe A. & Keough M. (1998) An investigation of the indirect effects of intertidal shellfish collection. Journal of. *Experimental Marine Biology and Ecology* 223:19-38.

Shepherd K.A. & Wilson P.G. (2007) Incorporation of the Australian genera Halosarcia, Pachycornia, Sclerostegia and Tegicornia into Tecticornia (Salicornioideae, Chenopodiaceae). *Australian Systematic Botany* 20:319-331.

Shepherd S.A., McComb A.J., Bulthius D.A., Neverauskas V., Steffensen D.A. & West R. (1989). Decline of seagrasses. In: Larkum A.W.D., McComb A.J. and Shepherd S.A. (eds.) *Biology of Seagrasses*: treatise on the biology of seagrass with special reference to the Australia region. Elsevier, Amsterdam, pp. 346-387.

Shepherd S.A., Watson J.E., Womersley H.B.S. & Carey J.M. (2009) Long-term changes in macroalgal assemblages after increased sedimentation and turbidity in Western Port, Victoria, Australia. *Botanica Marina* 52:195-206.

Simpson S., Fitzpatrick R., Shand P., Angel B., Spadaro D., Merry R. & Thomas M. (2008) *Acid, metal and nutrient mobilisation following rewetting of acid sulfate soils in the Lower Murray*. Report for the Environment Protection Agency, South Australia.

Sinclair M. (1997) *Ecology of intertidal crabs, and interactions between native and introduced species of crabs*. PhD thesis, Monash University.

Skilleter G.A., Zharikov Y., Cameron B. & McPhee D.P. (2005) Effects of harvesting callianassid (ghost) shrimps on subtropical benthic communities. *Journal of Experimental Marine Biology and Ecology* 320:133-158.

Smale D., Kendrick G., Waddington K., Van Niel K., Meeuwig J. & Harvey E. (2010) Benthic assemblage composition on subtidal reefs along a latitudinal gradient in Western Australia. *Estuarine, Coastal and Shelf Science* 86:83-92.

Smith B., Coleman N. & Watson J.E. (1975) The invertebrate fauna of Westernport Bay. *Proceedings of the Royal Society of Victoria* 87:149-155.

Smith T.M. & Hindell J.S. (2005) Assessing effects of diel period, gear selectivity and predation on patterns of microhabitat use by fish in a mangrove dominated system in south-east Australia. *Marine Ecology Progress Series* 294:257-270.

Smythe G.D. (1842) Survey of Westernport. *Lands Department, Melbourne, ACT*.

Sorte C.J.B., Williams S.L. & Zerebecki R.A. (2010) Ocean warming increases threat of invasive species in a marine fouling community. *Ecology* 91(8):2198-2204.

Specht R.L. & Specht A. (1999) *Australian plant communities*. Oxford University Press, Melbourne.

Spenceley A.P. (1987) Mangroves and intertidal morphology in Westernport Bay, Victoria, Australia - comment.. *Marine Geology* 77:327-330.

Spencer-Jones D., Marsden M.A.H., Barton C.M. & Carillo-Rivera P. (1975) Geology of Westernport sunkland. *Proceedings of the Royal Society of Victoria* 87:43-67.

Spencer C., Robertson A. & Curtis A. (1998) Development and testing of a rapid appraisal wetland condition index in south-eastern Australia. *Journal of Environmental Management* 54:143-159.

Spencer J., Monamy V. & Breitfuss M. (2009). Saltmarsh as habitat for birds and other vertebrates. In: Spencer J., Monamy V. & Breitfuss M. (eds.) *Australian saltmarsh ecology*. CSIRO publishing, Melbourne, pp. 149-165.

Srinivasan M. & Swain G.W. (2007) Managing the Use of Copper-Based Antifouling Paints. *Environmental Management* 39:423-441.

Stachowicz J.J., Whitlatch R.B. & Osman R.W. (1999) Species diversity and invasion resistance in a marine ecosystem. *Science* 286:1577-1579.

Stanton J. & O'Sullivan J. (2006) *Stock and waterways: a manager's guide*. Land & Water Australia.

Stapleton K.L., Long M. & Bird F.L. (2001) Comparative feeding ecology of the ghost shrimps Biffarius arenosus and Trypaea australiensis (Decapoda: Callianassidae). *Ophelia* 55:141-150.

Stapleton K.L., Long M. & Bird F.L. (2002) Comparative feeding ecology of two spatially coexisting species of ghost shrimp; Biffarius arenosus and Trypaea australiensis (Decapoda : Callianassidae). *Ophelia* 55:141-150.

Stark J.S. (1998) Heavy metal pollution and macrobenthic assemblages in soft sediments in two Sydney estuaries, Australia. *Marine and Freshwater Research* 49:533-540.

Steer M.A. & Moltschaniwskyj N.A. (2007) The effects of egg position, egg mass size, substrate and biofouling on embryo mortality in the squid Sepioteuthis australis. *Reviews in Fish Biology and Fisheries* 17:173.

Steffan W., Burridge A.A., Hughes L., Kitching R., Lindenmayer D., Musgrave W., Stafford Smith M. & Werner P.A. (2009) *Australia's biodiversity and climate change: a strategic assessment of the vulnerability of Australia's biodiversity to climate change*. A report to the Natural Resource Management Ministerial Council commissioned by the Australian Government. CSIRO publishing, Melbourne.

Stephens A.C. (1995) Seagrasses in Western Port, Victoria. Australia. Publication No. 490. Environment Protection Authority, Victoria.

Stephens J.H.G. (1979) Some physical, chemical and bacteriological characteristics of a temperate mangrove sediment. PhD thesis, University of Melbourne.

Sternberg R.W. (1979) Bottom current measurements and circulation in Western Port, Victoria. *Marine Geology* 30:65-83.

Sternberg R.W. & Marsden M.A.H. (1979) Dynamics, sediment transport, and morphology in a tide-dominated embayment. *Earth Surface Processes* 4:117-139.

Stevens J.D. & West G.J. (1997) *Investigation of school and gummy shark nursery areas in south eastern Australia*. Fisheries Research and Development Council Final Report: Project 93/061.

Stewart J., Deletic A., Fletcher T. (2010) Application of PEST (Parameter Estimation) to the PortsE2 WaterCAST Model to Estimate Annual Flows, Loads and Uncertainty to Port Phillip Bay and Western Port. In: *Proceedings, Ozwater 2010 Convention and Exhibition*. Australian Water Association, 8-10 March 2010, Brisbane, Australia.

Stewart J. (2011) *PortsE2 model calibration and extension project*. Report to Melbourne Water.

Stewart K., Judd A. & Edmunds M. (2007) *Flinders Pier and Foreshore Coastal Management Plan: Marine Ecology Baseline Survey*. Australian Marine Ecology report to USR.

Strong D.R. & Ayres D.R. (2009). Spartina introductions and consequences in saltmarshes: arrive, survive, thrive and sometimes hydridize. In: Strong D.R. & Ayres D.R. (eds.) *Human impacts on salt marshes. A global perspective* University of California Press, Berkeley, pp. 3-22.

Study V.S. (2010) *Mangroves and coastal saltmarsh of Victoria: distribution, condition, threats and management*. Victoria University, Institute for Sustainability and Innovation.

Sumby J. (2001) Mudflat community response to biogenic habitat structuring by the exotic grass, *Spartina anglica*. Honours thesis, Monash University.

Sundback K., Enoksson V., Graneli W. & Petterson K. (1991) Influence of sublittoral microphytobenthos on the oxygen and nutrient fluxes between sediment and water: a laboratory continuous flow study. *Marine Ecology Progress Series* 74:263-279.

Sutherland D. & Dann P. (in press) Population size in burrowing seabirds: improving accuracy of population size estimates for burrowing seabirds *Ibis*.

16 References

Taylor I.R. & Bester A. (1999) The response of foraging waders to human recreation disturbance at Rhyll, Phillip Island, Victoria (abstract). *Stilt* 35:67.

Taylor S.L., Bishop M.J., Kelaher B.P. & Glasby T. M. (2010) Impacts of detritus from the invasive alga Caulerpa taxifolia on a soft sediment community. *Marine Ecology Progress Series* 420:73-81.

Thomsen M. & McGlathery K. (2006) Effects of accumulations of sediments and drift algae on recruitment of sessile organisms associated with oyster reefs. *Journal of Experimental Marine Biology and Ecology* 328:22-34.

Thomsen M.S., Adam P. & Silliman B.R. (2009). Anthropogenic threats to Australasian coastal salt marshes. In: Thomsen M.S., Adam P. & Silliman B.R. (eds.) *Human impacts on salt marshes. A global perspective*. University of California Press, Berkeley, pp. 361-390.

Thrush S.F., Lawrie S.M., Hewitt J.E. & Cummings V.J. (1999). The problem of scale: uncertainties and implications for soft-bottom marine communities and the assessment of human impacts. In: Thrush S.F., Lawrie S.M., Hewitt J.E. & Cummings V.J. (eds.) *Biogeochemical cycling and sediment ecology*. Kluwer, Dordrecht, pp. 195-210.

Thrush S.F. & Dayton P.K. (2002) Disturbance to marine benthic habitats by trawling and dredging: implications for marine biodiversity. *Annual Review of Ecology and Systematics* 33:449-473.

Thrush S.F., Hewitt J.E., Cummings V.J., Ellis J.I., Hatton C., Lohrer A.M. & Norkko A. (2004) Muddy waters: elevating sediment input to coastal and estuarine habitats. *Front. Ecol. Environ.* 2:299-306.

Thrush S.F., Halliday J., Hewitt J.E. & Lohrer A.M. (2008) The effects of habitat loss, fragmentation, and community homogenization on resilience in estuaries. *Ecological Applications* 18:12-21.

Thrush S.F., Hewitt J., Hickey C.W. & Kelly S. (2008) Multiple stressor effects identified from species abundance distributions: Interactions between urban contaminants and species habitat relationships. *Journal of Experimental Marine Biology and Ecology* 366:160-168.

Toll Western Port (2001) *Draft Environmental Management Plan For the Port of Hastings* The Toll Group.

Tsutsumi H. (1990) Population persistence of *Capitella* sp. (Polychaeta; Capitellidae) on a mud flat subject to environmental disturbance by organic enrichment. *Marine Ecology Progress Series* 63:147-156.

Tudge C.C., Poore G.C.B. & Lemaitre R. (2000) Preliminary phylogenetic analysis of generic relationships within the Callianassidae and Ctenochelidae (Decapoda: Thalassinidea: Callianassoidea). *Journal of Crustacean Biology* 20 (Special Issue 2):129-149.

Tuominen L., Mäkelä K., Lehtonen K.K., Haahti H., Hietanen S. & Kuparinen J. (1999) Nutrient fluxes, porewater profiles and denitrification in sediment Influenced by algal sedimentation and bioturbation by Monoporeia affinis. *Estuarine, Coastal and Shelf Science* 49:83-97.

Turner L., Tracey D., Tilden J. & Dennison W.C. (2004) *Where river meets sea: exploring Australia's estuaries*. CRC for Coastal Zone, Estuary and Waterway Management, Brisbane.

Udy J.W., Dennison W.C., Lee Long W.J. & McKenzie L.J. (1999) Responses of seagrass to nutrients in the Great Barrier Reef, Australia. *Marine Ecology Progress Series* 185:257-271.

Underwood G.J.C. & Kromkamp J. (1999) Primary production by phytoplankton and microphytobenthos in estuaries. *Advances in Ecological Research* 29:93-153.

UNESCO (1983). Algorithms for computation of fundamental properties of seawater. UNESCO technical papers in marine science 44.

UNESCO (2010). The international thermodynamic equation of seawater – 2010: calculation and use of thermodynamic properties. Manuals and Guides 56. Intergovernmental Oceanographic Commission of UNESCO.

Valentine J. & Johnson C. (2003) Establishment of the introduced kelp Undaria pinnatifida in Tasmania depends on the disturbance to native algal assemblages. *Journal of Experimental Marine Biology and Ecology* 295: 63-90.

Van Dao H., Ngeh L.N., Bigger S.W., Orbell J.D., Healy M., Jessop R. & Dann P. (2006) Magnetic cleansing of weathered/tarry oiled feathers – the role of pre-conditioners. *Marine Pollution Bulletin* 52:1591-1594.

van der Valk A.G. & Attiwill P.M. (1984a) Acetylene reduction in an Avicennia marina community in southern Australia. *Australian Journal of Botany* 32:157-164.

van der Valk A.G. & Attiwill P.M. (1984b) Decomposition of leaf and root litter of Avicennia marina at Westernport Bay, Victoria, Australia. *Aquatic Botany* 18:205-221.

van der Valk A.G. & Attiwill P.M. (1984c) Decomposition of leaf and root litter of Avicennia marina at Westernport Bay, Victoria, Australia. *Aquatic Botany* 18:205-221.

Victorian Coastal Council (2008) Victorian Coastal Strategy.

Victorian Government (2001) P*olicy impact assessment: Protecting Water Quality in Western Port Bay. Variations of State environment protection policy (Waters of Victoria). Schedule F8*. Victorian Government Gazette, S192.

Victorian Saltmarsh Study* (2011) *Mangroves and coastal saltmarsh of Victoria: distribution, condition, threats and management*. Institute for Sustainability and Innovation, Victoria University. *Boon P., Allen T., Brook J., Carr G., Frood D., Hoye J., Harty C., McMahon A., Mathews S., Rosengren N., Sinclair S., White M. & Yugovic J.

Voice M., Harvey N. & Walsh K. (2006) *Vulnerability to climate change of Australia's coastal zone: analysis of gaps in methods, data and systems thresholds*. Australian Greenhouse Office.

Vollebergh P.J. & Congdon R.A. (1986) Germination and growth of Ruppia polycarpa and Lepilaena cylindricarpa in ephemeral saltmarsh pools, Westernport bay, Victoria. *Aquatic Botany* 26:165-179.

Walker D.I. & McComb A.J. (1992) Seagrass degradation in Australian coastal waters. *Marine Pollution Bulletin* 25:191-195.

Walker D.I. (1999). The marine angiosperms. In: Walker D.I. (ed.) *Flora and fauna of Australia*. Introduction. Department of the Environment and Heritage/CSIRO Publishing, Canberra/Melbourne, pp. 420-426.

Walker D.I., Kendrick G.A. & McComb A.J. (2006). Decline and recovery of seagrass ecosystems - the dynamics of change. In: Walker D.I., Kendrick G.A. & McComb A.J. (eds.) *Seagrasses: Biology, ecology and their conservation*. Springer Verlag, pp. 551-566.

Wallbrink P.J. & Hancock G. (2003) *Western Port sediment study: Background and literature review*. CSIRO Land and Water Technical Report 12/03. CSIRO Land And Water.

Wallbrink P.J., Hancock G., Olley J.M., Hughes A., Prosser I.P., Hunt D., Rooney G., Coleman R. & Stevenson J. (2003) *The Western Port sediment study*. CSIRO consultancy report.

Walton W.C, MacKinnon C., Rodriguez L.F., Proctor C. & Ruiz G.M. (2002) Effect of an invasive crab upon a marine fishery: green crab, Carcinus maenas, predation upon a venerid clam, Katelysia scalarina, in Tasmania (Australia). *Journal of Experimental Marine Biology & Ecology* 272: 171-189.

Wardrop J.A., Butler A.J. & Johnson J.E. (1987) A field study of the toxicity of two oils and a dispersant to the mangrove Avicennia marina. *Marine Biology* 96:151-156.

Waring J.S., Maher W.A. & Krikowa F. (2006) Trace metal bioaccumulation in eight common coastal Australian polychaeta. *Journal of Environmental Monitoring* 8:1149-1157.

Warneke R.M. (1975) Dispersal and mortality of juvenile fur seals Arctocephalus pusillus doriferus in Bass Strait, southeastern Australia. *Rapports et Proces-verbaux des Reunions Conseil International pour l'Exploration de la Mer* 169:296-302.

Warry F.Y. & Hindell J. (2009) *Review of Victorian seagrass research, with emphasis on Port Phillip Bay*. Draft Report. Arthur Rylah Institute for Environmental Research, Victoria.

Warry F.Y. & Reich P. (2010) *Development of a methodology for fish assessment to support the Victorian Index of Estuarine Condition*. Arthur Rylah Institute for Environmental Research. Victoria.

Warwick R.M., Pearson T.H. & Ruswahyuni (1987) Detection of pollution effects on marine macrobenthos: further evaluation of the species abundance/biomass method. *Marine Biology* 95:193-200.

Watkins D. (1993) *A national plan for shorebird conservation in Australia*. Australasian Wader Studies Group of the Royal Australasian Ornithologists Union.

Watson G.F., Robertson A.I. & Littlejohn M.J. (1984) Invertebrate macrobenthos of the seagrass communities in Western Port, Victoria. *Aquatic Botany* 18:175-197.

Watson J.E. (2009) Marine ecological monitoring for Cold Strip Mill, 1973-2003. North Arm channel, Western Port, Victoria. *Marine Science and Ecology*, Melbourne.

Waycott M. & Sampson J.F. (1997) The mating system of an hydrophilous angiosperm Posidonia australis (Posidoniaceae). *American Journal of Botany* 84:621-625.

Waycott M., Duarte C.M., Carruthers T.J.B., Orth R.J., Dennison W.C., Olyarnik S., Calladine A., Fourqurean J.W., Heck K.L., Hughes A.R., Kendrick G.A., Kenworthy W.J., Short F.T. & Williams S.L. (2009) Accelerating loss of seagrasses across the globe threatens coastal ecosystems. *Proceedings of The National Academy of Sciences of the United States of America* 106:12377-12381.

Webb J.A. (2000) The effects of copper antifouling paints and marinas on the ecology of marine sessile epifaunal assemblages. PhD thesis. University of Melbourne.

Webb A.P. & Eyre B.D. (2004) Effect of natural populations of burrowing thalassinidean shrimp on sediment irrigartion, benthic metabolism, nutrient fluxes and denitrification. *Marine Ecology Progress Series* 268:205-220.

Webb J.A. & Keough M.J. (2000) Effects of two fouling marinas on the composition of fouling assemblages. *Biofouling* 16:345-360.

Webb J.A. & Keough M.J. (2002) Measurement of environmental trace-metal levels with transplanted mussels and diffusive gradients in thin films (DGT): a comparison of techniques. *Marine Pollution Bulletin* 44:222-229.

Webster I.T., Parslow J.S., Grayson R.B., Molloy R.P., Andrewartha J., Sakov P., Tan K.S., Walker S.J. & Wallace B.B. (2001) Assessing options for improving water quality and ecological function. CSIRO report for the Gippsland Coastal Board.

Weis J.S. & Weis P. (2003) Is the invasion of the common reed, Phragmites australis, into tidal marshes of the eastern US an ecological disaster? *Marine Pollution Bulletin* 46:816-820.

Welsh D.T. (2000) Nitrogen fixation in seagrass meadows: Regulation, plant-bacteria interactions and significance to primary productivity. *Ecology Letters* 3:58-71.

Wernberg T., Campbell A., Coleman M.A., Connell S.D., Kendrick G. A., Moore P. J., Russell B. D., Smale D. A. & Steinberg P. D. (2009) Macroalgae and Temperate Rocky Reefs. Marine Climate Change in Australia. Impacts and adaptation responses. 2009 Report Card. CSIRO.

Wernberg T., Thomsen M.S., Tuya F., Kendrick G.A., Staehr P.A. & Toohey B.D. (2010) Decreasing resilience of kelp beds along a latitudinal temperature gradient: potential implications for a warmer future. *Ecology Letters* 13:685-694.

Western Port Greenhouse Alliance (2008) Impacts of climate change on settlements in the Western Port region. People, property and places. Western Port Greenhouse Alliance (WPGA), Narre Warren.

Whetton P. & Power S. (2007) *Climate change in Australia*. Technical Report. CSIRO and the Bureau of Meteorology.

Widdows J. & Brinsley M. (2002) Impact of biotic and abiotic processes on sediment dynamics and the consequences to the structure and functioning of the intertidal zone. *Journal of Sea Research* 48:143-156.

Williams S.L. & Grosholz E.D. (2008) The invasive species challenge in estuarine and coastal environments: marrying management and science. *Estuaries and Coasts* 31:3-20.

Williamson R. (1996) *Spartina* in Victoria – an overview. *Australasian conference on* Spartina *control* (eds. Rash JAE, Williamson RC & Taylor SJ). Department of Conservation and Natural Resources, Yarram.

Wilson R.S., Heislers S. & Poore G.C.B. (1998) Changes in benthic communities of Port Phillip Bay, Australia, between 1969 and 1995. *Marine and Freshwater Research* 49:847-861.

Worm B., Lotze H.K., Bostrom C., Engkvist R., Labanauskas V. & Sommer U. (1999) Marine diversity shift linked to interactions among grazers, nutrients and propagule banks *Marine Ecology Progress Series* 185:309-314.

WPRPCC (1992) Western Port Bay Strategy: a strategy for the protection and development of Western Port, Victoria. Western Port Regional Planning and Coordination Committee.

Wright J. (1996) Processes affecting the distribution and abundance of the barnacle *Elminius covertus*. PhD thesis, University of Melbourne.

Yugovic J. (1984) The Grey Glasswort (Halosarcia halocnemoides) in coastal Victoria and some implications for the Orange-bellied Parrot. *The Victorian Naturalist* 101:234-239.

Yugovic J. & Mitchell S. (2006) Ecological Review of the Koo-Wee-Rup Swamp and associated grasslands. *The Victorian Naturalist* 123:323-334.

Yugovic J. (2008) F*lora and fauna of The Inlets, Koo-Wee-Rup, Victoria*. Biosis Research report to Melbourne Water.

Zuccarello G.C., Yeates P.H., Wright J.T. & Bartlett J. (2001) Population structure and physiological differentiation of haplotypes of *Caloglossa leprieurii* (Rhodophyta) in a mangrove intertidal zone. *Journal of Phycology* 37:235-244.

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